Effects of Watershed and Habitat Conditions on Stream Fishes
in the Upper Roanoke River Watershed, Virginia

by
Vann F. Stancil

Thesis submitted to the faculty of the
Virginia Polytechnic Institute and State University
In partial fulfillment of the requirements for the degree

MASTER OF SCIENCE

in Fisheries and Wildlife Science

<u>Committee</u>
Donald J. Orth, Chair
Paul L. Angermeier
John J. Ney

June 22, 2000
Blacksburg, Virginia

Key Words: Land Use, Stream Fishes, Stream Habitat, Watershed Conditions, Urbanization,
Biological Integrity

Effects of Watershed and Habitat Conditions on Stream Fishes
in the Upper Roanoke River Watershed, Virginia.

by

Vann F. Stancil

Donald J. Orth, Chair

Fisheries and Wildlife Sciences

(Abstract)

I collected fish samples and habitat data at 43 sites throughout the upper Roanoke River watershed, Virginia. Sites were separated into three watershed areas size classes: 10 – 15, 20 – 30, and 70 – 80 km^2. I correlated physical in-stream conditions with proportions of forest, disturbed, and herbaceous/agricultural land at various watershed-scales to determine factors affecting stream habitat. I grouped fishes into metrics commonly used in indexes of biotic integrity and created a multimetric index called the mean metric score to represent fish communities at sites. Fish variables and metric values were compared with stream habitat and watershed variables to determine primary influences on fish communities. I correlated land use at 24 spatial scales, which differed by buffer width and stream network area, with mean metric scores to determine zones of greatest influence on fish communities.

In-stream habitat conditions and amounts of forest, herbaceous/agricultural, and disturbed land varied greatly among sites. Habitat varied due to natural differences among sites, such as elevation and watershed area, and due to land use. Disturbed land use was greatest at lower elevations while forests were more abundant at higher elevations. Substrate size distribution was highly correlated with all three land use types at several spatial scales. Correlations between land use within various buffers and median particle size became stronger as larger proportions of watersheds were included in analysis.

Fish species richness increased from small to large sites by species addition. Species collected at small sites were also collected at large sites, but several species collected at large sites were absent elsewhere. For example, orangefin madtoms and bigeye jumprocks were only collected at three large sites. Fish distribution was a result of several factors such as watershed area, elevation, proportions of pools and of riffles, particle size, and land use within buffers and entire watersheds.

Sites with high mean metric scores were primarily limited to tributaries of the North and South Forks of the Roanoke River. Most sites with low mean metric scores were located near the cities of Roanoke and Salem. Forest and disturbed land use were highly correlated with mean metric scores. Elevation was also highly correlated with mean metric scores but herbaceous/agricultural land use was not. Correlations between percent forest within 24 buffers and mean metric scores were highest for small stream network areas and declined as more land farther from sites was included for analysis. Correlations between disturbed land use and mean metric scores were strong regardless of the area considered. Mean metric scores declined precipitously as disturbed land use within watersheds and buffers increased from 0 to 10 %, but reached a plateau at 10 to 20 % after which increases in disturbed land use did not result in lower mean metric scores.

My results suggest that species addition and ecological shifts from more generalized to more specialized species occur with increased stream size. Forested buffers are important for maintaining ecological integrity, and buffers along sites with adequate integrity should be candidates for riparian restoration. Future development should be concentrated in watersheds that are already developed and reforestation of riparian areas in developed watersheds may reduce the impacts of watershed-level disturbance.

ACKNOWLEDGMENTS

This study was made possible by funding provided by the Environmental Protection Agency and the National Science Foundation. I appreciate the commitment these organizations have to research such as mine. This study was part of a larger study examining many aspects of land use changes in the upper Roanoke River watershed. I learned a lot from the interactions of study participants from departments throughout Virginia Tech and my study benefited from their questions and suggestions.

I was blessed with excellent technicians throughout my study. Without their help, I would not have been able to sample streams as effectively as I did. In 1998, Chris Markham, Chelsea Cada, Mike Joyce, and Adrienne Weimer formed my core field crew. In 1999, I was assisted by Mary Rummel, Cliff Hutt, and Joe Grist. Matt Chan, my fellow Orthling, graciously assisted in the field during both summers. I am deeply grateful to all of them for their hard work, companionship, and cooperation. I thoroughly enjoyed our time together.

I could not have sampled large streams very effectively without the help of many volunteers. Thirty-nine people volunteered their services for one or more sites. These volunteers were Ben Orth, Brenden O'Neill, Don Orth, Craig Roghair, Tim Copeland, Lisa Hudson, Sam Orth, Kelly Smith, Ryan Sponseller, Chelsea Crenshaw, Mike Cyterski, Mike Joyce, Danny Laffoon, Kelly McCarty, Ron Small, Brett Albanese, Paul Angermeier, Ray Brooks, Chelsea Cada, Scott Chappell, Dawei Chen, John Copeland, David Crowder, Robert Dietz, Christina Faustino, Todd Fearer, Greg Galbreath, Desirae Goff, Christiana Grim, Juan Guzman-Aranda, Tim Meehan, Andrea Mummert, George Palmer, Susan Rogers, Mark Shabman, Reed Shabman, Ryan Smith, Chris Vandergoot, and Eric Warner. I appreciate their time and hard work and hope that I enhanced their appreciation of streams and stream fishes.

I could not have fully utilized the capabilities of geographical information systems without the help of Robert Dietz. He selflessly worked many hours with me to derive land use data at many scales and other information imperative for this study. I could not have done it without him. I am also indebted to the many landowners that allowed me to sample fish on their property. Nearly every landowner I spoke with was happy to let me sample on their land and was genuinely interested in the quality of their stream and the fish in it.

Science is an additive process that builds upon previous research. The Upper Roanoke River has been a popular playground of many ichthyologists and ecologists through the years. I am particularly indebted to Robert Jenkins and Noel Burkhead for synthesizing the vast array of information on Virginia fishes into one source: *Freshwater Fishes of Virginia*. I am also grateful to Roy Smogor for his work collecting fish throughout Virginia and classifying them for index of biotic integrity metrics. I used his classification system and saved a lot of time in the process. I appreciate the input that Brett Albanese, Matt Chan, Ed Pert, Kirk Krueger, and Ryan Smith gave me while discussing research issues through the years.

I was fortunate to be surrounded by a fine group of graduate students while at Virginia Tech. I appreciate their help and friendship through the years. Don Orth has been very patient and generous while I have been working with him. I have enjoyed learning from him at many levels. I thank my committee members, Paul Angermeier and John Ney, and also Tammy Newcomb, for their advice regarding my research and thesis. Rich Noble at North Carolina State University not only gave me helpful advice as my undergraduate advisor, he gave me opportunities to work on several research projects. I am forever grateful for the opportunities he gave me directly and indirectly.

I have many people to thank for guiding my path and making the thesis possible. I thank God for all the blessings in my life and for enabling me to complete this thesis. I thank Amy, my best friend and wife, for her constant love, support, and encouragement. She has always been there to congratulate me when things go well and to pick me up when things are not going as I hoped. She considered the completion of this thesis to be a team effort and I appreciate her motivation toward the end and throughout the process. I thank my parents for always pushing me to do my best and for their support during my lengthy academic journey. I thank my other family members for their support through the thesis years, even if they lacked an understanding of what I was doing in graduate school and could not tell a sucker from a minnow. Lastly, I thank all of the friends, teachers, and others I have known through the years that have in some way shaped who I am today.

TABLE OF CONTENTS

LIST OF TABLES

CHAPTER 1. Effects of Watershed Conditions on Stream Habitat and Substrate Composition

CHAPTER 2. Fish Abundance and Distribution Related to Habitat and Watershed Conditions

LIST OF FIGURES

INTRODUCTION

CHAPTER 1. Effects of Watershed Conditions on Stream Habitat and Substrate Composition

LIST OF APPENDICES

INTRODUCTION

As the human population continues to increase, more land is required to satisfy the needs of society. Housing developments, shopping malls, and roads replace forests and fields as humans spread over the American landscape. Such changes do not come without a price. These changes affect the air we breathe, the water we drink, and the landscapes we view. With planning and forethought, we can live sustainably and minimize our impacts on natural systems around us. With haphazard, uncontrolled development, the world we leave for our descendants will be far inferior to the one we inherited.

Changes in the landscape do not occur in isolation - conditions in streams and rivers reflect conditions within their watersheds. By examining streams and their biota, we can better understand the effects we have on our environment. Biotic communities within degraded watersheds consist of organisms able to persist in streams where many ecological interactions have been altered. These communities often consist of species tolerant of impaired conditions while species that are sensitive to environmental perturbations are rare or absent. Fish communities are important measures of stream and watershed conditions because they signify environmental changes and are also valued by society. Thus, fish communities are simultaneously monitors of stream quality and targets for conservation.

Fish communities are affected by habitat conditions at various spatial and temporal scales (Frissell et al. 1986, Schlosser 1995, Imhof et al. 1996). Changes in land use patterns can affect fish communities through changes in stream habitat (e.g., Gorman and Karr 1978, Angermeier and Karr 1984, Wang et al. 1997, Walser and Bart 1999), water quality (e.g., Matthews and Styron 1981, Berkman and Rabeni 1987, Ryan 1991, Smale and Rabeni 1995), and hydrology (e.g., Poff and Ward 1989, Allan and Flecker 1993, Schueler 1994, May et al 1997). These land use changes can affect various life history traits relating to feeding and reproduction by altering food availability and degrading reproductive habitats (e.g., Berkman and Rabeni 1987, Ryan 1991, Rabeni and Smale 1995, Smith 1999). Because changes in land use affect fish species differently, researchers can use various attributes of fish communities to evaluate anthropogenic impacts (Karr 1981, Karr et al. 1996). Although continued development and sprawl seem inevitable, through a better understanding of land use changes on streams and their biota, future development scenarios with minimal environment impacts can be pursued. To mitigate impacts, planners need a better understanding of what aspects of stream habitat quality are affected by disturbance from development.

This study examined the effects of land use conditions on streams and fish communities in the upper Roanoke River watershed (URRW). The goal of this project was to determine how land use affects stream habitat and fish communities in order to suggest land development scenarios that will minimize these impacts in the future. As a result of these findings, I was able to identify land use impacts on stream habitat, evaluate the response of various fishes to land use disturbances, identify streams with fish communities of high integrity, and evaluate the utility of different metrics for future indexes of biotic integrity (IBI). Examining the effects of land use at various spatial scales enabled me to suggest future development scenarios that can benefit aquatic resources, such as protecting riparian areas and concentrating development in currently degraded watersheds.

Chapter One lays the foundations for future chapters by presenting data about the in-stream conditions and watershed conditions at 43 sites that I sampled in the URRW. I quantified in-stream habitat conditions such as substrate size distribution and depth at various sites and land use conditions for watersheds upstream from these sites. Habitat conditions were then compared with land use conditions to determine the effects of land use on stream habitat. This analysis revealed pathways by which land use changes affect streams. Chapter Two introduces the fish communities of streams in the URRW. It describes trends related to stream size and the effects of various watershed and habitat variables on fishes. Chapter Three examines the fish community from a multimetric index standpoint. I used 13 metrics to look at the biological integrity of the fish communities and compared metric results with watershed and habitat variables. This analysis indicated which species were affected by various conditions and indicated the overall effect of land use changes on fish communities.

REVIEW OF LITERATURE

Disturbance by urbanization changes the landscape by replacing pervious surfaces with impervious surfaces such as roads, parking lots, and buildings. These impervious surfaces increase the rate that precipitation enters streams, causing discharge to increase at a faster rate and causing peak flows to be greater than in forested or agricultural landscapes (Klein 1979, Booth 1990, Schueler 1994). Impervious surfaces also increase the frequency of flood events because less precipitation is required to exceed bankfull discharge (Klein 1979, Schueler 1994). Channelization often accompanies the increase in impervious surfaces in urban watersheds, causing the hydrology of urban watersheds to deviate more from pre-disturbance conditions.

Water quality is often degraded in urban areas, and riparian conditions are often poor, further compounding the effects of urbanization (Klein 1979). Water quality in urbanized watersheds is impaired from both point and non-point sources. Industries produce point-source pollution while runoff from paved surfaces contains pollutants from automobiles and runoff from lawns increases nutrient input. Sedimentation, caused by streambank erosion or construction activities, also degrades streams in urban areas (Jones and Clark 1987). The width of vegetated riparian areas is often narrow along streams in developed areas and woody vegetation along streams is important for controlling water temperature, stabilizing streambanks, and providing a source of allochthonous inputs and woody debris (Castelle et al. 1994). Large woody debris contributes to habitat complexity and the creation of riffle-pool sequences, especially in small streams (Beschta and Platts 1986).

All of these changes in stream habitat quality and stability combine to make conditions for aquatic organisms much different than in less-altered systems. Reoccurring flood events can cause adult fish to be injured by bedload movement or displaced by high flows. These flood events can also eliminate year classes by depositing bedload or scouring substrate in spawning areas, and by displacing juveniles (Schlosser 1985, Harvey 1987, Smith 1999). Changes in water quality can also impact many fish species during all life stages. Sedimentation affects spawning success of fish and can lower food abundance by decreasing macroinvertebrate densities (Ryan 1991). A reduction in woody debris abundance can also reduce macroinvertebrate abundance, as well as reduce habitat complexity and refugia. These changes, brought about by changes in watershed land use at various distances from the stream, impact fish communities in many ways. Therefore, determining which factors have the most impact is difficult.

Watershed land use should be viewed from a landscape perspective with an understanding of the connnectedness between watershed conditions and stream conditions (e.g., Gregory et al. 1991, Schlosser 1991). The effects of land use practices on streams and aquatic life are affected by the spatial distribution of land use practices. Numerous studies have addressed the impacts of riparian zone conditions on in-stream habitat and fish communities (e.g., Kauffman and Krueger 1984, Wohl and Carline 1996). Previous studies have shown that entire watershed conditions (Wang et al. 1997, Allan et al. 1997), partial watershed conditions (Steedman 1988), or riparian conditions (Allan et al. 1997) can have the greatest influence on fish communities. With a better understanding of the effects of the spatial distribution of landscape disturbance on stream systems, urban planners will have the necessary tools to advocate development scenarios that minimize impacts on stream systems. For example, if narrow riparian buffers (e.g., 10 m) along streams are sufficient to maintain proper stream structure and function, riparian buffer strips should be maintained along all streams while land beyond this zone can be developed. However, if land use for the entire watershed has the greatest impact on streams and land use conditions near streams have no impact, different development scenarios should be pursued.

Fish communities are affected by conditions at multiple spatial scales ranging from the watershed to microhabitat level (Frissell et al. 1986, Allan et al. 1997), and have been effectively used to characterize the biological integrity of aquatic resources associated with changes in land use practices. The Clean Water Act of 1977 requires that the biological integrity of water be maintained, in addition to physical and chemical integrity (Hocutt 1981, Karr 1981). Other measures of water quality, such as chemical analyses to determine concentrations of nutrients and toxic chemicals, are still important and can be used in conjunction with fish and macroinvertebrate communities to better assess the quality and integrity of aquatic resources.

Biological monitoring with fish and macroinvertebrates is important because biota can reflect conditions over a longer period of time than water quality, which reflects conditions at the time of sampling. Fish communities also indicate the effects of habitat changes resulting from channelization, siltation, flow regime alteration, and dam construction. Water quality monitoring alone would not detect such changes (Karr and Chu 1997). Fish community assessments can be done quickly with limited equipment and provide information that can be easily understood by the public. By integrating changes to our aquatic resources, fish communities provide information about the effects of land use conditions occurring over multiple spatial and temporal scales.

Fish community attributes can be measured in several ways. Traditional methods to indicate the quality of a fishery, such as species abundance, fish density, biomass, and condition of individual fish, can be used effectively to relate fish community quality. Additional methods of characterizing a fish community, such as measures of diversity and the use of indicator species, are important when assessing the effects of land use on species that differ in their tolerance, longevity, and feeding and reproductive styles. Indexes of biotic integrity (Karr 1981) have been used effectively to describe the fish community of degraded streams. IBIs compares several attributes of fish communities called metrics at a disturbed site with those at reference sites which have been minimally disturbed. Scoring criteria are determined for each metric based on reference conditions, and the scores for each metric are added to produce a total score for a given site. The sites compared should be similar in size and should have had similar fish community attributes prior to anthropogenic disturbance (Karr and Chu 1997).

The original IBI was established for small, warmwater streams in the Midwest and comprised 12 metrics that reflected the species composition and richness of the fish community as well as metrics related to ecological function (Karr 1981). Community attributes vary with stream size (Vannote et al. 1980), drainage (Smogor and Angermeier 1999a), and are different for cold- and warmwater systems (Karr and Chu 1997). Therefore, different metrics and scoring criteria are required for different regions of the country as well as differences in streams on smaller scales relating to size or temperature. The IBI has been effectively modified to assess fish communities in other areas of the United States (e.g., Fausch et al. 1984, Leonard and Orth 1986, Miller et al. 1988, Bramblett and Fausch 1991, Smogor 1996) as well as Ontario (Steedman 1988), Mexico (Lyons et al. 1995), and France (Oberdorff and Hughes 1992). Reference conditions, metrics that reflect differences between reference and impacted conditions, and scoring criteria must be determined for different regions. Numerous metrics have been used in IBIs; some such as taxa richness and number of intolerant species have proven to be consistently good metrics while others are highly variable (Karr and Chu 1997).

Urban land use for streams near Toronto in southern Ontario was negatively correlated with IBI scores (Steedman 1988). IBI scores increased with increasing proportions of forest in watersheds and land use immediately upstream of sampling sites (partial watershed) was more strongly associated with IBI scores than land use for the entire watershed.

IBI scores were negatively correlated with agricultural and urban land use in Wisconsin streams (Wang et al. 1997). They found watershed land use to be a better predictor of fish community quality than riparian land use. IBI scores declined drastically between 10 and 20 % urban land use; all watersheds with more than 20 % urban land use had IBI scores indicative of degraded conditions. Relationships between agricultural land use and IBI scores indicated a threshold around 50 % but some sites with higher amounts of agriculture still maintained fish communities with high integrity. Habitat scores for urbanized streams varied greatly and were not negatively correlated with urbanization, suggesting that water quality or hydrological impacts may have more impact on the fish community than habitat quality in urban streams.

Lenat and Crawford (1994) examined the effects of land use on water quality, macroinvertebrates, and fish in three North Carolina streams by comparing a forested, agricultural, and urban stream. The agricultural and urban streams were inferior to the forest stream, but their similarity to the forested stream differed. Fish community data indicated that the agricultural stream was similar to the forested stream; the two streams had equal numbers of species but the agricultural stream had greater total abundance. The urban stream was inferior, receiving a 34 (poor) IBI score while the forested and agricultural streams received scores of 50 (good) and 48 (good), respectively. This study showed the importance of using several methods (water quality, macroinvertebrates, and fish) to assess the effects of land use.

Stream fish communities can be affected by historical land use conditions in addition to current conditions (Harding et al. 1998). Historical land use was a better predictor of current stream conditions and macroinvertebrate and fish communities than current land use. Decades are needed for streams to recover from past disturbances such as siltation (Harding et al. 1998).

Individual metrics have also been used effectively to assess changes in fish community attributes caused by anthropogenic disturbances. Berkman and Rabeni (1987) found that trophic and reproductive metrics were able to distinguish between riffle fish communities in streams impacted by siltation and unimpacted streams. Metrics used in IBIs can be used individually to assess the impact of disturbance by comparing fish assemblages in disturbed streams with those from similar streams differing only in land use practices.

Disturbance in streams can be divided into two categories: pulse disturbance and press disturbance (Yount and Niemi 1990). Pulse disturbances, such as floods, instantaneously alter lotic communities but after the disturbance, the community gradually recovers to its previous state. Press disturbances cause sustained changes in lotic communities, and the biotic community adjusts to the different conditions, but does not return to its pre-disturbance state (Yount and Niemi 1990). Fish communities usually recover from pulse disturbances within a year but recovery from press disturbances, which affect habitat quality, can take much longer

(Detenbeck et al. 1992). For example, the fish community in the South Fork of the Roanoke River recovered quickly from a pulse disturbance caused by a manure spill (Ensign et al. 1997).

Increased flood frequency due to an increase in impervious surfaces can be viewed as a press disturbance. If floods occur at a high frequency and especially during spawning season (Schlosser 1985, Harvey 1987, Smith 1999), they could depress some fish populations beyond recovery. Habitat changes from siltation, channelization, and changes in riparian conditions are press disturbances that require more time for fish communities to recover (Harding et al 1998).

Hydrological variability can affect fish community composition. Poff and Ward (1989) analyzed streamflow patterns from across the United States and predicted biological attributes for fish found in streams differing in flood frequency, flood predictability, and flow predictability. They postulated that fish communities in streams prone to more frequent flooding events, as would be expected in areas with high proportions of impervious surfaces, would be comprised of small-bodied fishes. The community would also be less speciose, and have an uneven age distribution relative to streams with less variable hydrological conditions.

In an analysis of streams in Wisconsin and Minnesota, Poff and Allan (1995) classified streams as hydrologically stable or variable using gauge data and compared fish community attributes based on presence/absence of species from existing data. Stable streams had significantly more benthic invertivores, fewer omnivores, and fewer silt-tolerant species while variable streams had fewer species typically associated with rubble, more species associated with silt, and more tolerant species.

Other studies have demonstrated the effects of flow variability on fish communities. Schlosser (1985) found that floods caused a reduction in juvenile abundance of minnow and sunfish species, thereby altering short-term community composition. Harvey (1987) found that centrarchids and cyprinids < 10 mm TL were very susceptible to displacement following flood events but that susceptibility rapidly declined with increased size. High flows can cause bedload movements that erode spawning areas or deposit bedload onto spawning areas of fishes (Smith 1999). Abundance of fluvial specialists was inversely related to proximity to a hydroelectric dam on an Alabama river, demonstrating that fluvial specialists abundance increased as the effects of variable releases dissipated (Kinsolving and Bain 1993).

Stream fishes exhibit different habitat preferences (Aadland 1993, Vadas 1994) and diversity of substrate, depth, and current velocity can directly influence species diversity

(Gorman and Karr 1978). When habitat is altered and becomes more homogeneous, species with specific habitat requirements can be extirpated. Angermeier and Karr (1984) found that woody debris promoted habitat complexity and was important for pool maintenance in Jordan Creek, Illinois. Most species showed a preference for areas with woody debris and were less abundant where woody debris was removed. Fish communities were more temporally stable in streams with complex habitat after flooding than communities in more homogeneous habitat (Pearsons et al. 1992). Habitat complexity is important because it provides refugia for fishes during harsh conditions and will therefore affect fish survival and recolonization (Schlosser 1995).

Species also vary in their tolerance of water quality and physicochemical conditions as shown with darter and minnow species from various size streams in the upper Roanoke River watershed, Virginia (Matthews and Styron 1981). Reproductive and feeding plasticity varies among species as well. Species that have specific spawning requirements, such as simple lithophilic spawners that require clean substrate, are affected by degradation more than species with less specific reproductive requirements. Likewise, species with specific feeding habits are more likely to be affected by changes in land use practices than those with wide dietary breadth. Such ecological specialization was found to be a common attribute among Virginia fishes prone to extinction (Angermeier 1995). These variations in species tolerance cause some species to be extirpated by anthropogenic degradation while others are able to maintain stable populations.

STUDY BASIN

The Roanoke River flows southeast from southcentral Virginia and meets the Dan River in the upper portion of Kerr Reservoir east of South Boston, Virginia. The Roanoke River then flows into North Carolina where it joins the Chowan River in the Albemarle Sound (Figure 1.1). It is impounded by five large reservoirs: Smith Mountain Reservoir and Leesville Reservoir in Virginia, Kerr Reservoir and Gaston Reservoir on the Virginia-North Carolina border and Roanoke Rapids Reservoir in North Carolina. The Roanoke River flows through three physiographic provinces in Virginia: Valley and Ridge, Blue Ridge, and Piedmont and through the Piedmont and Coastal Plain in North Carolina (Jenkins and Burkhead 1993).

The Roanoke River fish fauna has been considered the most speciose and distinctive of all Atlantic slope drainages south of the St. Lawrence River (Jenkins and Burkhead 1993). The Roanoke drainage contains 119 fish taxa; 82 are considered native. Six taxa are endemic to the

Roanoke drainage: Roanoke hogsucker, Hypentelium roanokense; rustyside sucker, Thoburnia hamiltoni; bigeye jumprock, Scartomyzon ariommus; orangefin madtom, Noturus gilberti; an undescribed spotted madtom tentatively considered a subspecies of N. insignis, and riverweed darter, Etheostoma podostemone. The Roanoke drainage has a rich diversity of Catostomidae; 14 species are found in the drainage, three of them are endemic (Jenkins and Burkhead 1993).

STUDY WATERSHED

This study took place in the upper Roanoke River watershed (URRW) which covers 1,500 km^2 in Montgomery, Roanoke, Botetourt, and Floyd counties, Virginia (Dickson 1979). All of the streams in the URRW, except Back Creek, are upstream of Niagara Dam and all streams are upstream of Smith Mountain Reservoir. The URRW is in the Valley and Ridge and Blue Ridge physiographic provinces (Jenkins and Burkhead 1993). I divided the URRW into five smaller systems: South Fork, North Fork, mainstem Roanoke River with associated tributaries, Tinker Creek including Glade Creek, and Back Creek (Figure 1.1).

Urbanization has increased in the URRW, especially in the cities of Roanoke, Salem, and the surrounding areas. Streams impacted the most by urbanization (such as Peters, Mudlick, and Wolf Creeks) flow directly into the mainstem of the Roanoke River. Deer Branch and Lick Run, tributaries of Tinker Creek near the city of Roanoke, are also heavily impacted by urbanization. Urbanization has also impacted portions of Back and Glade Creeks.

The entire URRW is nearly 70 % forested with agricultural and herbaceous land covering over 16 % of the land and developed land over 11.5 %. Land cover in the North Fork is primarily forested with agriculture, primarily livestock grazing, occurring in lower elevations near streams. South Fork tributaries have the highest gradients and elevation within the URRW and are primarily forested. Mainstem tributaries, with the exception of Mason Creek, have small watersheds less than 25 km^2 and low gradients. The Tinker system is comprised of low gradient tributaries of Tinker Creek and Glade Creek and drains land impacted by development and agriculture. Much of the Back Creek watershed is forested but development continues to increase.

With an estimated 59 species from nine families (44 considered native) found in the URRW, the area is relatively rich in fish diversity (Sheldon 1988). Four species endemic to the Roanoke drainage, including the orangefin madtom, which is considered threatened by the state of Virginia, are found in the URRW. In addition, the Roanoke logperch, <u>Percina</u> <u>rex</u>, a federally endangered species found only in the Roanoke and Chowan drainages, persists in the URRW. American eels, <u>Anguilla</u> <u>rostrata,</u> and Roanoke bass, <u>Ambloplites</u> <u>cavifrons,</u> were historically present in the URRW. Numerous dams along the Roanoke River have eliminated American eels from the area while competition and hybridization with rock bass, <u>Ambloplites</u> <u>rupestris,</u> have eliminated Roanoke bass from the URRW.

CHAPTER 1. Effects of Watershed Conditions on Stream Habitat and Substrate Composition

Conditions in streams reflect natural conditions and human alterations within their watersheds. Conversion of forests to agriculture or development can change streams in several ways such as altered channel form, increased sedimentation, reduced abundance of woody debris, increased hydrologic variability, and increased temperature variability. All of these changes affect fish communities. Through a better understanding of the relationships between land use conditions and in-stream conditions, we can determine pathways through which watershed level changes are manifested in streams. By understanding these pathways, we can determine how land use conditions will affect stream communities and be better armed to mitigate these impacts.

OBJECTIVES

The goal of the first chapter was to determine how land use throughout watersheds affects in-stream conditions. The first objective was to summarize values for habitat and watershed variables to better understand the variability of conditions among sites. This was important for reporting the range of watershed and habitat conditions and for assessing differences related to stream size. The second objective was to correlate habitat variables with watershed and land use variables. I did this to understand which variables were correlated at sites and to identify possible causal relationships. By identifying relationships between watershed and in-stream conditions, I hoped to determine possible pathways through which watershed conditions influenced fish communities via in-stream conditions. I expected land use to influence substrate size and also expected to see trends relating in-stream and land use conditions to factors such as site elevation and watershed area. A sub-objective of the second objective was to determine how land use at various spatial scales affected substrate size. I assessed this objective by correlating land use within 24 buffers of various sizes with median particle size. This analysis was important because land use was heterogeneously distributed within watersheds.

METHODS

Site selection

I selected sites representing various stream sizes and watershed scale land uses. I divided sites into three distinct size classes based on watershed area to mitigate the effects of increased fish species richness usually observed with increased stream size (e.g., Jenkins and Freeman 1972, Hambrick 1973). Several small streams in the upper Roanoke River watershed have watershed areas of 10 - 15 km^2 when they enter other tributaries or the mainstem. Many other streams had maximum watershed areas of 20 - 30 km^2 and 70 - 80 km^2, so I selected sampling sites that would fit into one of these three watershed size classes. I chose general areas for sampling by delineating watersheds using ArcView GIS or using USGS 7.5 minute topographical maps and a dot grid to estimate watershed area.

I located sites that were optimistic representations of surrounding stream reaches. If there was a large pool resulting from natural stream functions in the area, I included it because it might provide unique habitat unavailable nearby and contain fishes absent in adjacent stream reaches. Exact sampling sites were demarcated after securing landowner permission. I determined mean stream width at sites by measuring wetted width at 10 points (15 for large sites) spaced approximately 10 m apart. This distance was estimated by walking within streams the number of steps that equaled 10 m. I then determined site length by multiplying this pre-sample mean width by 35. Sampling a distance of 35 times the mean stream width made site length a function of width and ensured that several riffle-pool sequences were included (Lyons 1992). This sampling distance is sufficient to assess number of species and relative abundance in montane Virginia streams (Angermeier and Smogor 1995). I selected natural breaks, such as downstream ends of riffles or cascades, for upper and lower site boundaries. These breaks should hamper fish movement while sampling.

Habitat sampling

My field crew and I collected habitat data during late Spring and early Summer at 30 sites in 1998 and 13 sites in 1999. We assessed habitat at all 43 sites, but did not recollect habitat data at seven sites when their fish communities were resampled in 1999. My field crew and I collected habitat data at each site after collecting and processing fish. Habitat measurements were substrate particle size, depth, canopy closure, stream width, and habitat type (pool, riffle, run, etc.), and land use along sites.

We developed a measuring tape for each site using mean widths determined when first demarcating sites. Flagging was spaced along the measuring tape at 10% of the mean width. The first flagging was located 5 % of the mean width from the water's edge and subsequent flagging points were spaced 10 % of the mean width apart (Figure 1.2). For example, if the mean stream width was five meters, the first flagging would be 0.25 m from the water's edge on the right bank and subsequent flagging would occur at 0.75 m, 1.25 m, 1.75 m, etc. After wrapping flagging around the tape and stapling it at 15 to 20 places, we positioned the tape across the stream channel perpendicular to flow. Measurements were taken at each of these points along the transect until reaching the water's edge on the left bank. Transects were spaced two mean stream widths (MSWs) apart based on recommendations by Simonson et al. (1994) and began at MSW 1 (downstream end of site was MSW 0). We used another tape that was two MSWs long to space the transects by following the stream thalweg upstream. We determined stream width at each transect using the measuring tape and determined depth, canopy closure, and substrate particle size at each flagging point. Measuring tape flagging was always moved between sites because mean stream widths varied among sites and measurement points had to be adjusted accordingly.

We characterized substrate using a method similar to the pebble count method (Kondolf and Li 1992, Harrelson et al. 1994). We selected the first substrate particle touched with an index finger from directly below each transect flagging point and determined particle size using a gravelometer. The gravelometer was a thin, square plexiglass sheet with nine square holes representing distinct substrate size classes. Gravelometers perform the same function for large particles that sieves perform for small ones and particles belong to the size class corresponding to the smallest hole they will pass through. Size classes increased in a geometric manner with each size class being 1.41 times larger than the preceding class. Each square hole accommodated a range of particle sizes and this range increased as the hole size increased for larger particles. The gravelometer measured the intermediate, or B, axis of a substrate particle, which is not the widest or narrowest dimension of a particle, but the one in between. I used the scale provided by Harrelson et al. (1994) to convert particle sizes into meaningful substrate terms such as gravel and cobble (Table 1.1). The gravelometer had holes for particles from sizes four to 12 which corresponded to fine gravel and medium cobble. We sized finer particles, (i.e.,, silt, sand, and very fine gravel) based on feel and appearance and measured the intermediate axis of particles too large to fit through any gravelometer holes using a meter stick. Occasionally we encountered bedrock substrate and recorded it as such without a size classification.

We measured depth to the nearest centimeter and determined canopy closure by counting the number of flags covered directly overhead (by tree limbs, bridges, tall grass, etc.), regardless of vertical distance above the stream. We classified transect habitat type as wholly or partially riffle, run, pool, cascade, backwater, or island by recording the number of flags included in each habitat type. By identifying habitat type based on flag position, an approximation of proportion of each habitat type was possible and substrate and depth characteristics could be matched with habitat type. Habitat types were determined visually using traits such as flow rate, depth, and water surface slope. Unless a transect had two or more distinct habitat types, the entire cross-section was classified as one habitat type. I avoided dividing a transect into multiple habitat types because it promoted microhabitat identification rather than considering habitat type on a larger spatial and temporal scale, such as appearance during different discharges.

Site land use was visually estimated at every four MSWs beginning at MSW 1. This constituted land use within five meters of each channel bank and was classified as woody vegetation, impervious surfaces, weeds or grass, yard, pasture, bare ground, or row crop agriculture in 10 % increments (e.g., 60 % woody vegetation, 40 % pasture). We determined discharge during sampling by measuring depth and flow rate at several points along a channel cross-section. We determined flow rate at depths a distance of 60 % below the water surface using a Marsh-McBirney Flo-Mate Model 2000 Portable Flowmeter.

Substrate variables for each site were determined by ranking size classes for all particles from smallest to largest and identifying D10 (10th percentile), D50, and D90 particle size classes. I excluded substrate classified as bedrock from these calculations. I also determined D50 values for pools and riffles. I converted gravelometer size classes to mm measurements by taking the geometric mean, which is the square root of the product of the smallest and largest possible measurements for each size class (Table 1.1). I then square root transformed these values to linearize the data. The proportion of silt and sand, which corresponded to size codes one and two, represented fine sediment. I calculated standard deviation of substrate using values that had been converted to mm sizes and then square root transformed. This technique minimized the effect of large particles on standard deviation values.

Depth measurements were ranked in ascending order to determine median depth. The maximum depth recorded during habitat sampling was also used because it indicated deep habitat availability at each site. I also calculated standard deviation of depth to represent depth variability at each site. Canopy closure represented stream shading and riparian conditions for each site and I determined it by dividing the number of covered flags by the total number of flags at all transects. I determined proportions of riffles and of pools for each site by counting the number of flags representing riffles or pools and dividing by the total number of flag measurements. Mean stream width was the average of all transect widths determined during habitat sampling. Pre-sample widths were only used to determine site length and transect measurement distances; they were replaced by mean transect widths for analysis.

Site land use conditions were originally classified as woody vegetation, impervious surfaces, weeds or grass, yard, pasture, bare ground, or row crop agriculture for each side of the channel. Percentage values for each transect were averaged to represent the entire site. I combined yard values with grass or weeds values for analysis because they were often hard to differentiate. Bare ground and row crop agriculture were excluded from analysis because they never represented a large portion of land use conditions at sites and were often nonexistent.

Watershed level analysis

Robert Dietz of the Civil and Environmental Engineering Department at Virginia Tech and I did all watershed level analysis using ArcView GIS Version 3.1. Land use for the URRW was a clipped portion of the draft land cover grid of Virginia produced through the Virginia Gap Analysis Project (VAGAP). VAGAP interpreted fourteen LANDSAT Thematic Mapper scenes and used several methods to create the grid. The grid had a pixel resolution of 30 X 30 m and was georeferenced in UTM zone 17 coordinates with North American Datum (NAD) 1927 as the datum. Land use was characterized into 28 different types. Stream channel networks were obtained from six digital line graph (DLG) files based on 1:100,000 scale USGS topographic quadrangles. The URRW was covered by 24 individual digital elevation models (DEMs) of 30 m^2 resolution at a scale of 1:24,000. The 24 DEMs were joined in a seamless mosaic to provide continuously varying elevation data (Dietz 2000).

I determined site location coordinates from 7.5 minute USGS topographic quadrangle maps or a GPS unit and we added them to an ArcView GIS field. We determined watershed areas by demarcating watershed boundaries upstream of sites and determined elevation at each site with a DEM. We buffered each site with a 60 m radius circle and chose the lowest elevation within the area to represent site elevation. We used this method to reduce variation in elevation values because elevation was averaged for each 30 m^2 cell and point estimates usually overestimated stream channel elevation.

We simplified land use from 23 different types to five categories for analysis: forest, herbaceous/agriculture (herb/ag), disturbed, mixed/unknown, and open water (Table 1.2). Land classified as mixed contained more than one land use and precluded classification as only one type. Ground truthing of mixed areas revealed that such areas are often low intensity developed areas containing impervious surfaces as well as trees and lawns. These mixed areas varied in amount of different land uses; therefore it was not feasible to exactly determine their composition. As a result, we eliminated open water, which constituted a small percentage of watershed land use, and mixed/unknown as land use categories. We determined percent land use for forest, herb/ag, and disturbed categories by dividing land areas of each category by the total land area of the three.

To assess the effects of land use within areas closer to sites, we developed other variables called partial watershed buffers that were corridors of various widths around stream channels and included varying proportions of stream channel networks upstream of sites. These buffers excluded land use far away from channels and excluded varying amounts of land along channels far upstream from sites to determine zones of greatest influence on stream habitat and fish communities (Figure 1.3). The first step in this process was averaging watershed areas for each of the three watershed size classes. I determined a circle size that would encompass an area 5, 10, 20, 40, or 60 % of the mean watershed area for each size class. In addition to circle areas, I included entire stream networks upstream of sites, as buffer areas for spatial scale analysis. Circles were positioned upstream of each site so they would encompass the largest watershed area possible and still include the site. Streams were then buffered by 30, 60, 90, and 150 meters on each side and land use determined for the buffer area that was within both the circle and watershed. This technique produced 24 different combinations of partial buffer areas ranging from a 5 % circle with a 30 m buffer (5 %, 30 m buffer) to the entire stream network with a 150 m buffer. The amount of land area included for each site depended on the length of stream encompassed by the circle; this varied with channel density and watershed shape.

Statistical analysis

To determine if watershed area had an effect on habitat variables, I compared 29 habitat variables segregated by watershed size class using Friedman's Test. This is a nonparametric test similar to an ANOVA which tests the three groups, in this case habitat variables for each watershed size class, for differences (SAS Institute Inc. 1999). To determine if stream conditions were significantly different during 1998 and 1999, I compared data from these years using two-tailed t-tests. To investigate land use changes with buffer widths and stream network areas, I averaged land use across all 43 sites for 24 buffer combinations. I determined land use percentages for different buffer widths, such as land use from 60 to 90 m, by excluding land use data from narrower buffers. This method indicated how land use changed with varying distances from stream channels by excluding data from smaller buffers widths.

I correlated watershed land use variables with buffer variables of the same land use type and buffer variables with corresponding site land use variables to determine if conditions close to sites were generally indicative of conditions elsewhere in the watersheds. To determine possible land use causes for stream habitat conditions, I correlated habitat variables with land use variables at three spatial scales: sites, entire watershed, and 60 %, 60 m buffers. To assess the influence of land use at various spatial scales on substrate size, I correlated D50 values with partial buffer land use at 24 spatial scales. I used Spearman's rank order correlation coefficient (r_s) for all correlations. I chose D50 values because the median particle size is often used for substrate assessments and will decline with sedimentation. Additionally, preliminary analysis showed strong correlations between D50 values and land use variables. Significance for all statistical comparisons was determined at $\alpha = 0.05$.

RESULTS

Habitat and watershed data summary

My field crew and I sampled all medium and large streams and most small streams in the URRW (Appendix A). In 1998 and 1999, we sampled 43 different sites on second through fourth order streams: 22 small, 13 medium, and eight large sites (Figure 1.4). Site widths ranged from 2.8 to 11.0 m (mean = 5.1 m) and site lengths ranged from 104 to 353 m (mean = 184 m). Watershed areas ranged from 9.9 to 83.9 km^2 and means for the three watershed size classes were 13.2, 28.6, and 73.1 km^2. Site elevation ranged from 276 m (Lower Glade Creek) to 766 m (Upper Bottom Creek) and averaged 421 m. Mean widths were significantly different among watershed size classes ($P < 0.001$, Friedman's Test, Table 1.3). Correspondingly, site length and area sampled, which are proportional to mean width, were significantly different among watershed size classes.

Mean widths determined during habitat sampling were generally close to pre-sample mean widths. Pre-sample widths only averaged 0.22 m less than sample widths and for seven of 43 sites the two means were identical. Mean site length for all sites was 36 MSWs and site lengths met or exceeded the goal of 35 MSWs for 29 of 43 sites (67.4 %). Additionally, only one site length was less than 30 MSWs and 38 sites were 30-39 MSWs in length (88.4 %). A scarcity of natural breaks sometimes caused sites to be longer than mean widths necessitated; this provided insurance against inappropriately short sites.

Habitat conditions varied considerably among sites in terms of substrate, depth, and land use conditions (Appendix B). Canopy closure averaged 58 % but ranged from 2 % for a pasture site to 93 % for a forested site. D50 substrate size ranged from sand to small cobble, while the percent of silt and sand ranged from < 1 to 50 % of substrate particles. Median depth ranged from 4 to 28 cm and maximum measured depth ranged from 26 to 97 cm. Proportions of riffles and pools were as high as 45 and 85 %, respectively. Most sites had some bedrock substrate and while this was usually a small amount, bedrock represented > 30 % of the substrate at three sites. All substrate size variables except riffle D50 showed small, consistent increases in mean particle size from small to large watersheds (Table 1.3). Discharge at sites ranged from 0.005 to 0.263 m^3/s (discharge at Lower Glade Creek was 0.521 m^3/s following a rain) and averaged 0.094 m^3/s. Mean discharge was higher at small sites (0.072 m^3/s) than medium sites (0.065 m^3/s) because they were sampled earlier in the summer. Discharge was positively correlated with watershed area ($r_s = 0.277$, $P = 0.072$), but the relationship was not significant at $\alpha = 0.05$.

Woody vegetation and weeds/grass/yard were the two most common land use conditions at sites. Woody vegetation was present at all sites; it averaged 43 % and ranged from 1 to 99 %. Impervious surfaces were as high as 38 % while pasture reached 98 % and the combination of weeds, grass, and yard reached 87 %. The proportion of impervious surfaces increased from small to large sites and was the only variable not directly linked to stream size that was significantly different among watershed size classes (Table 1.3). This increase is probably because many large sites were paralleled by roads while smaller sites were more often crossed by roads than paralleled.

The summers of 1998 and 1999 were drier than normal and 1999 was the drier of the two years. As a result, stream widths were narrower than under normal rainfall conditions. Mean width (P = 0.027, two-tailed t-test) and area sampled (P = 0.023) were significantly greater for small sites in 1998 than 1999. Site lengths were significantly greater for small (P = 0.019) and medium sites (P = 0.021) in 1998 than 1999. There were no significant differences in discharge between years for any watershed size and no significant differences at $\alpha = 0.05$ for any large watershed comparisons.

Over 61 % of the entire URRW is classified as forest, making it by far the most common land use in the study basin (Table 1.2). Well-forested watersheds are limited to portions of the South and North Forks of the Roanoke River, Mason Creek, and Back Creek (Figure 1.5). Herb/ag land use affects over 14 % of the URRW. Agricultural land use is greatest in Tinker Creek, Glade Creek, North Fork, and Elliott Creek watersheds. Twelve percent of the URRW is classified as disturbed; 84 % of this disturbed land, or 10 % of the URRW, is low or high intensity development. Most development occurs in the cities of Roanoke and Salem and surrounding areas and all of the most disturbed watersheds are found in these areas. Land classified as mixed/unknown, which has characteristics of wooded, disturbed, and herbaceous or agricultural land, covers nearly 12 % of the basin. If mixed/unknown land is proportionally distributed among the three primary land use types, 69.9 % of the land is the URRW is forest, 13.8 % is disturbed, and 16.3 % is herb/ag.

Sites varied considerably in entire watershed land use (Appendix C). Watershed forest varied the most among sites while many sites had very little or no disturbed or herb/ag land. Eighteen watersheds were at least 80 % forested; three of these were at least 96 % forested. Five watersheds were over 40 % disturbed, while 15 were less than 1 % disturbed. Watershed herb/ag land ranged from 1.3 to 65.5 %. Lick Run had the least amount of forested land (4.4 %) and the most disturbed land (63.5 %). Although on average there was more forested land and less disturbed land in watersheds of large sites, there were no significant differences among watershed size classes for any watershed or buffer level land use variables (Table 1.3).

Land classified as disturbed or herb/ag was more common in buffers than in entire watersheds and conversely, a smaller proportion of buffer land was forested. Percent forest was less for buffers than entire watersheds primarily because areas close to streams are more suitable for houses, agriculture, and development due to lower gradients and proximity to water. Generally, as stream network area increased by including land farther upstream of sites, percent forest increased and percent herb/ag and disturbed land decreased. Averaging land use values across all sites for buffer areas revealed how land use varied with buffer width and stream network area (Figure 1.6). Average percent forest increased with stream network area and generally declined as buffer width increased. The amount of forest in partial buffers varied the most, ranging from 44 to 66 %. Percent herb/ag and disturbed land was highest for small stream network areas (5 % areas) and wide buffers (150 m). Percent of herb/ag land only varied from 23 to 31 %, while disturbed land use ranged from 11 to 25 % for different scales.

Land use varied considerably within watersheds as well. Forests covered over 80 % of the entire watershed of UTWORC, Horners Branch, and Lower Mason Creek. However, disturbed land use within the smallest buffers (5 %, 30 m) for these sites exceeded 80 %. Although these sites are extreme cases, many other sites had forested headwaters but were affected by urbanization or agriculture in areas downstream near my sites.

Averaging land use values for all sites masked trends for sites with extreme land use conditions, such as highly urbanized and well-forested streams. Comparisons of land use at various spatial scales for 12 urbanized and 14 forested sites (Figure 1.7) showed that land use trends at these sites were similar to averages for all sites (Figure 1.5). However, these comparisons showed that percent forest and percent disturbed changed across spatial scales more for urbanized sites than for means for all 43 sites. Mean percent disturbed land for these urban watersheds was only 30 % when all channels were buffered, but increased to 70 % for the 5 % areas closest to sites. For forested sites, percent forest was nearly 91 % for all channels but declined to 68 % when only land closest to sites was included. Percent forest for different network areas was greatest within 30 meters of streams; this mirrored comparisons using all 43 sites. Mean disturbed land use at well-forested sites was greatest within the 30 m buffer for the 5 % stream network area and declined for wider buffers and larger network areas.

Watershed and habitat relationships

Land use for entire watersheds was correlated with the same land use for partial buffers (i.e., percent watershed forested vs. percent buffer forested) for all three land use types ($r_s > 0.65$, $P < 0.0001$ for all). This connection between different spatial scales was much weaker between partial buffer and site land uses. Only 60 %, 60 m buffer forest and woody vegetation (r_s = 0.308) and buffer disturbed and weed/grass/yard (r_s = 0.344) were significantly correlated at α = 0.05. Buffer disturbed and impervious surfaces were not significantly correlated.

Sedimentation is expected to be a function of watershed land use activities such as agriculture and construction. I compared land use at three spatial scales with six substrate variables to determine which scale had the most influence on substrate size. None of the four site land uses were significantly correlated with any substrate variables (Table 1.4). However, the D50 substrate size was significantly correlated with all six watershed and 60 %, 60 m buffer land use variables ($|r_s| > 0.45$, $P \leq 0.002$ for all). Substrate variables were most often correlated with forest variables (11 of 12 comparisons). Watershed forest was the only variable significantly correlated with all substrate variables ($|r_s| \geq 0.33$, $P \leq 0.03$ for all). Substrate size increased as percent forest increased, but substrate size decreased as the percentage of herb/ag and disturbed land increased.

I also correlated depth and proportions of pools and riffles with watershed area, site elevation, and watershed land use. Median depth was significantly correlated with all three watershed land use variables (Table 1.4). It was positively correlated with herb/ag and disturbed land and negatively correlated with forested land. Proportion of riffles was positively correlated with site elevation ($r_s = 0.340$, $P = 0.026$). Proportion of riffles increased ($r_s = 0.363$, $P = 0.017$) and proportion of pools decreased ($r_s = -0.378$, $P = 0.012$) with increased percent buffer as forest. These two habitat type variables showed opposite trends with disturbed buffer land use. Elevation was negatively correlated with disturbed land use at the watershed ($r_s = -0.666$) and buffer ($r_s = -0.800$) level and was positively correlated with percent buffer as forest ($r_s = 0.541$, $P \leq 0.0002$ for all).

Using the D50 to assess the influence of land use at various spatial scales revealed that correlations between land use and substrate size strengthened as a greater area of land upstream of sites was included (Figure 1.8). Correlations with D50 values were stronger for forest and disturbed land use within entire watersheds than for any buffers (r_s = 0.698 for forest, -0.597 for disturbed). As percent forest increased, D50 values increased, while increases in percent herb/ag or disturbed land use resulted in lower D50 values. Spearman's coefficients (r_s) were consistently stronger at all spatial scales for comparisons of percent forest than for herb/ag or disturbed land with D50 values at the same scale. Buffer width had very little influence on relationships between percent forest and D50 values. For percent herb/ag land, $|r_s|$ values for a given stream network area declined for wider buffers, but showed the opposite trend for percent disturbed land.

DISCUSSION

Although discharge and watershed area were positively correlated, some streams have notable spring inputs which augment their flow (e.g., Mill, Elliott, and Glade Creeks) while others flow underground (e.g., Dry Branch, Brake Branch, Mason Creek, and Bradshaw Creek). Mean discharge was higher for small sites than medium ones primarily because small sites were sampled first each summer and discharge declined over the summer as sampling progressed. With the exception of Lower Glade Creek, which was sampled after a rain, discharge measurements were representative of flows for the time period surrounding sampling.

Dry conditions during both summers caused discharges, mean widths, and site lengths to be below normal. Water levels in Carvins Cove Reservoir in the Tinker Creek watershed decreased to more than nine meters below conservation pool and discharge for the Roanoke River at Roanoke was below historic low levels in the summer of 1999. Widths were smaller for sites sampled in 1999 than in 1998 resulting in smaller areas being sampled. Narrower widths were mostly due to drier conditions but inherent differences in stream width at the sites may have also had an effect. These dry conditions affected sampling by allowing shorter sites than mean widths would necessitate under normal discharges.

Land use conditions along sampling sites had little effect on substrate size distribution at my sites. Conversely, watershed and buffer level land uses were usually significantly correlated with substrate conditions. These data indicated that buffer conditions have effects on substrate size and suggest that improvements along stream buffers may reduce sedimentation and cause median particle sizes to increase.

I included six variables that assessed different aspects of the substrate size at sites because of the well-known influence of substrate and sediment on fish communities (e.g., Berkman and Rabeni 1987, Ryan 1991, Rabeni and Smale 1995). Additionally, watershed conditions are expected to have a direct influence on substrate in streams (e.g., Richards and Host 1994, Harding et al. 1998, Walser and Bart 1999). Of my six substrate variables (D50, D90, standard deviation of substrate, riffle D50, pool D50, and percent of silt and sand), D50 was most strongly correlated with watershed land use. It is also the best single indicator of substrate conditions because it also reflects conditions indicated by other variables. For example, D50 values decrease with increased sedimentation, as represented by percent of silt and sand, and are higher for sites with many large particles, which D90 represents. Four of these variables are good indicators of sedimentation: D50, riffle D50, pool D50, and percent of silt and sand. All variables except percent of silt and sand reflect the abundance of large substrate. However, the presence of large substrate is determined more by geological factors than changes in land use. Correlations between D90 values and land use may be more spurious that causal due to the influence of elevation, slope, and geology on both substrate size and land use. For future studies, I recommend median particle size, D50, for substrate assessments because it reflects sedimentation, and also indicates the amount of large particles, which is important for assessing streambed roughness, habitat complexity, and refugia.

Several significant correlations among variables complicate identifying single factors that influence in-stream habitat conditions the most. Percent disturbed land at entire watershed and buffer scales increased significantly at lower elevations. This is to be expected because urban areas are often established on low slope areas near water resources. The proportion of riffles also declined significantly as elevation decreased and disturbed buffer land use increased. Therefore these are three correlated variables with several possible causal relationships. Site elevation is an independent variable and the proportion of riffles is likely related to elevation changes, but not disturbance. Most likely, riffles are scarcer in disturbed areas because elevation (and slope) decreases. Therefore, disturbed buffer land use and proportion of riffles are likely correlated without a causal relationship, unless elevation is considered.

Comparisons with land use at 24 spatial scales revealed that median sediment size is influenced by factors from far upstream. Percent forest had more influence on D50 than herb/ag and disturbed land because the latter land uses cause increased sedimentation while percent forest accounts for both land uses simultaneously. Sedimentation, which reduces median particle size, is one of the most pervasive problems affecting aquatic ecosystems today (e.g., Berkman and Rabeni 1987, Ryan 1991, Bruton 1995, Rabeni and Smale 1995). Many recent studies have shown strong links between watershed scale land use practices, such as agriculture, logging, and urbanization, and in-stream sedimentation (e.g., Richards and Host 1994, Wohl and Carline 1996, May et al. 1997, Harding et al. 1998).

Sedimentation affects fish directly by clogging gills, smothering eggs and larvae, and affecting vision dependent behavior (Rabeni and Smale 1995). Fish are also impacted by habitat alterations that occur over time due to sedimentation. Interstitial spaces needed for egg deposition and important for aquatic macroinvertebrate communities become clogged by fine sediment (Ryan 1991). Fishes that require clean substrate for spawning or feed primarily on benthic insects suffer the most from fine sediment in streams (Berkman and Rabeni 1987, Rabeni and Smale 1995).

Protecting riparian areas is often seen as a vital measure to reduce sediment impacts in streams (e.g., Osborne and Kavicic 1993, Castelle et al. 1994, Davies and Nelson 1994, Rabeni and Smale 1995). Current restoration activities call for revegetating riparian areas, especially in areas impacted by livestock grazing (Wohl and Carline 1996, Gutzwiller et al. 1997, Kauffman 1997). My results indicate that land use activities have strong effects on sediment far downstream and merely revegetating riparian areas along streams in distances measured in hundreds of meters will have minimal effects. Intact riparian corridors throughout stream channel networks are needed to reduce sedimentation occurring throughout watersheds (Rabeni and Smale 1995). Although adequate buffer widths are usually discussed, buffer continuity and quality are also important considerations (Gregory et al. 1991, Naiman et al. 1993) and my data suggest that continuity of protective buffers throughout watersheds directly affects stream substrate size.

My results suggest that sedimentation impacts from livestock grazing and other agricultural practices can be reduced by riparian rehabilitation activities within a narrow buffer around streams. In my analysis, as wider buffers were considered, correlations of herb/ag land use with D50 values weakened. This suggests that herb/ag land closest to stream channels has the most impact, and this influence dissipates as land use farther from stream channels is included for analysis. Therefore, merely reducing grazing impacts within a 30 m or less area should make a noticeable difference, if impacts throughout drainage networks are addressed.

Sedimentation from disturbed land use, such as urbanization, cannot be mitigated as easily. Disturbed land use appears to affect sedimentation even when it occurs far from stream channels because the amount of disturbed land within the entire watershed was a better predictor of sedimentation than disturbed land use at any of the buffer scales. Construction activities related to roads and buildings are likely responsible for sedimentation impacts in developed watersheds (Jones and Clark 1987). Such activities take place throughout a watershed, not just in riparian zones. Wide, forested buffers along streams will likely intercept a large amount of sediment, and along with proper sediment control measures during construction, will reduce sedimentation problems in urbanizing watersheds. Because sources of sedimentation will vary as construction occurs in different parts of a watershed, a more complete buffer may be necessary in urbanizing areas than in agricultural areas where impacts are more visible, confined, and constant from year to year.

Current land uses are not the only sources of sedimentation in disturbed watersheds. Land that is currently being developed was either in forest, agriculture, or some state of disturbance prior to recent land use changes (Moglen 2000). These previous land use conditions can affect stream habitat conditions for decades, even if conditions are improved by reforestation (Harding et al. 1998). As forests and agricultural areas in the URRW are developed, stream conditions will reflect both current and past land use conditions. Stream substrate conditions in developed watersheds could be linked more to historical land uses, such as agriculture, because of the long amount of time required for streams to recover from extended degradation (Harding et al. 1998).

Recent studies, including mine, have demonstrated the importance of including land use within several spatial scales when examining land use effects on streams and their biota (Steedman et al. 1988, Richards and Host 1994, Hunsaker and Levine 1995, Allan et al. 1997, Wang et al. 1997, Harding et al. 1998). This is especially important when land use is not homogeneously distributed within watersheds. Results from previous studies have varied in their conclusions regarding the scale of land use that has the most influence on streams and their inhabitants. Therefore, when possible, obtaining land use data for several spatial scales is an important step toward assessing the influence of watershed conditions. Areas that encompass varying amounts of stream networks and differ in width within these areas can be used to examine portions of watersheds closest to streams.

In addition to entire watershed land use, I examined land use within 24 buffers that differed in stream network area and buffer width. I have a few recommendations for future studies based on my study. First, I recommend examining land use within different areas, especially if land use is heterogeneously distributed, as it was in many watersheds in the URRW. I also recommend concentrating partial watershed areas around stream channels and varying them by stream network area and buffer width because of the importance of riparian areas (Gregory et al. 1991, Naiman et al. 1993). The 150 m buffer category is arguably too wide because it is much wider than restoration areas near stream channels. If riparian conditions have been improved over large areas and scientists wish to assess their benefits, buffer widths corresponding to the width of these restored areas should be considered. The number of stream network area scales can also be reduced; perhaps only areas corresponding to watershed areas of 10, 40, and 60 % would be sufficient. This number should increase if land use is patchily distributed within watersheds (i.e., disturbed land concentrated near sites), but fewer network areas could be used if land use is homogeneously distributed.

Table 1.1.-Substrate size classes determined with a gravelometer and used to assess substrate size distribution. Calculated size is the geometric mean for each size class (square root of the product of the smallest and largest possible measurements for each size class). Size range is the minimum and maximum size for each size class. Substrate type identified based on Harrelson et al. (1994).

Size Class	Calculated Size (mm)	Size Range (mm)	Substrate Type
1	0.03	0.01-0.1	silt
2	0.4	0.1-2	sand
3	3.2	2-5	very fine gravel
4	7	5-9	fine gravel
5	11	9-13	medium gravel
6	16	13-19	medium gravel
7	22	19-27	coarse gravel
8	32	27-38	coarse gravel
9	45	38-53	very course gravel
10	63	53-75	very course gravel
11	90	75-107	small cobble
12	127	107-151	medium cobble
13	179	151-213	large cobble
14	254	213-302	very large cobble
15	359	302-427	small boulder
16	507	427-603	small boulder
17	718	603-853	medium boulder
18	1015	853-1207	medium boulder
19	1435	1207-1707	large boulder
20	2030	1707-2414	large boulder
21	2871	2414-3414	very large boulder

Table 1.2.-Twenty-eight original classifications and simplified categories (underlined) for land use for the entire upper Roanoke River watershed.

Classification	Area (m^2)	Percent
Forest		
dry deciduous	410,754,391	27.74 %
montane oak dominated	219,235,715	14.81 %
mesic deciduous	165,888,619	11.20 %
piedmont/coastal plain forest complex	30,654,641	2.07 %
montane yellow pine	27,341,185	1.85 %
montane dry oak dominated	22,284,230	1.50 %
montane mesic conifer	15,555,711	1.05 %
mixed central hardwood	11,928,443	0.81 %
red cedar woodland	2,081,588	0.14 %
submontane yellow pine	344,384	0.02 %
red spruce/fraser fir	0	0 %
riparian forest	0	0 %
submontane oak dominated	0	0 %
tupelo/red maple wet forest	0	0 %
Virginia deciduous forest complex	0	0 %
Forest Total	906,068,907	61.19 %
Disturbed		
high intensity developed	76,795,757	5.19 %
residential/low intensity developed	73,108,245	4.94 %
non-vegetated (mines, barrens, etc.)	27,426,607	1.85 %
recent clear cut	919,855	0.06 %
Disturbed Total	178,250,463	12.04 %
Herbaceous/agriculture		
high herbaceous/field crop	162,810,746	11.00 %
mixed herbaceous	45,101,671	3.05 %
sparse herbaceous/row crop	2,904,332	0.20 %
pasture/low herbaceous	651,901	0.04 %
Herbaceous/agricultural total	211,468,650	14.28 %
Others		
mixed class/unknown	174,355,242	11.77 %
open water	10,627,338	0.72 %
herbaceous wetland	0	0 %
coastal wet shrub	0	0 %
forested wetland	0	0 %

Table 1.3.-Mean values for 29 habitat and watershed level variables separated by watershed size class. Significant differences among size classes at $\alpha = 0.05$ using Friedman's Test signified by an asterisk. Values for substrate were determined by taking the square root (SQRT) of particle sizes in mm.

	Watershed Class			
Variable	Small	Medium	Large	*P*-value
N	22	13	8	
site length (m)	144.6	176.1	304.5	NA
mean width (m)	4.13	4.84	8.34	< 0.001*
area sampled (m^2)	610	876	2588	NA
MSWs sampled	35.8	36.6	36.8	NA
canopy closure	58.6%	58.5%	57.8%	0.886
median depth	11.5	13.0	14.8	0.413
max. observed depth (cm)	51.2	53.6	58.1	0.463
standard deviation of depth	10.81	11.56	12.79	0.337
proportion of riffles	0.238	0.249	0.235	0.859
proportion of pools	0.244	0.295	0.210	0.682
discharge (m^3/s)	0.072	0.065	0.169	0.086
D50 (mm, SQRT)	4.16	4.88	5.36	0.457
D90 (mm, SQRT)	10.69	12.14	12.70	0.407
standard deviation of substrate (mm, SQRT)	8.68	9.35	10.46	0.320
riffle D50 (mm, SQRT)	5.94	7.69	6.86	0.138
pool D50 (mm, SQRT)	3.01	3.92	4.04	0.799
proportion of silt and sand	0.22	0.20	0.17	0.509
proportion woody vegetation - site	0.47	0.33	0.47	0.394
proportion impervious surfaces - site	0.02	0.10	0.11	0.026*
proportion pasture - site	0.15	0.17	0.11	0.971
proportion weed/grass/yard - site	0.35	0.39	0.31	0.638
watershed area (km^2)	13.2	28.6	73.1	< 0.001*
site elevation (m)	411	452	396	0.610
% forest - entire watershed	66.0%	66.9%	72.1%	0.730
% herb/ag - entire watershed	20.3%	22.4%	21.6%	0.560
% disturbed - entire watershed	13.7%	10.7%	6.3%	0.810
% forest - 60 %, 60 m buffer	54.8%	59.6%	67.1%	0.754
% herb/ag - 60 %, 60 m buffer	26.1%	25.5%	17.8%	0.560
% disturbed - 60 %, 60 m buffer	19.0%	14.9%	15.0%	0.837

Table 1.4.-Comparisons of watershed and land use variables with habitat variables using Spearman's rank order correlation coefficient (r_s). Only correlations significant at $\alpha = 0.05$ are shown. For each significant correlation, r_s values are on top and *P*-values are below.

	Site Land Use Level						Entire Watershed Level			60 %, 60 m Buffer Level		
Habitat Variable	Woody Vegetation	Impervious Surfaces	Pasture	Weed/yard /grass	Watershed Area	Site Elev.	Forest	Herb/ag	Disturbed	Forest	Herb/ag	Disturbed
D50							0.698	-0.478	-0.597	0.587	-0.457	-0.463
							<0.0001	0.001	<0.0001	<0.0001	0.002	0.002
D90							0.343		-0.330	0.341		
							0.024		0.031	0.025		
st. dev. substrate							0.330		-0.312	0.413		-0.307
							0.031		0.042	0.006		0.045
riffle D50							0.372		-0.389	0.356	-0.367	
							0.015		0.011	0.021	0.017	
pool D50							0.337					
							0.029					
% of silt & sand							-0.561	0.473	0.369	-0.487	0.478	
							<0.0001	0.001	0.015	0.001	0.001	
median depth							-0.434	0.346	0.330	-0.385		
							0.004	0.023	0.031	0.011		
max. depth	0.341											
	0.025											
st. dev. depth						-0.379			0.306			
						0.012			0.046			
prop. riffles						0.340			-0.308	0.363		-0.333
						0.026			0.045	0.017		0.029
prop. pools		-0.315								-0.378		-0.318
		0.040								0.012		0.038

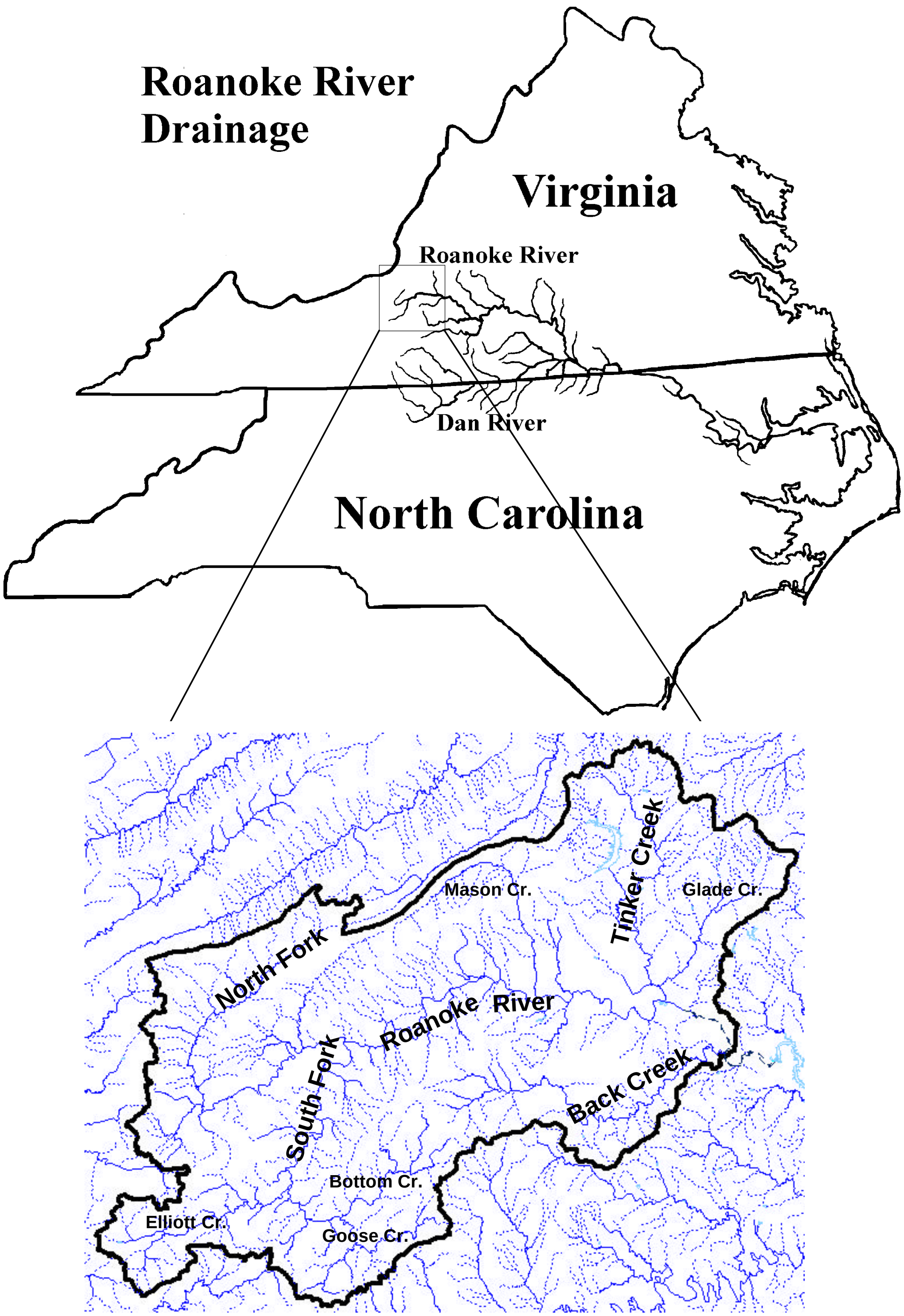


Figure 1.1 Maps of the Roanoke River drainage and upper Roanoke River watershed showing the main tributaries.

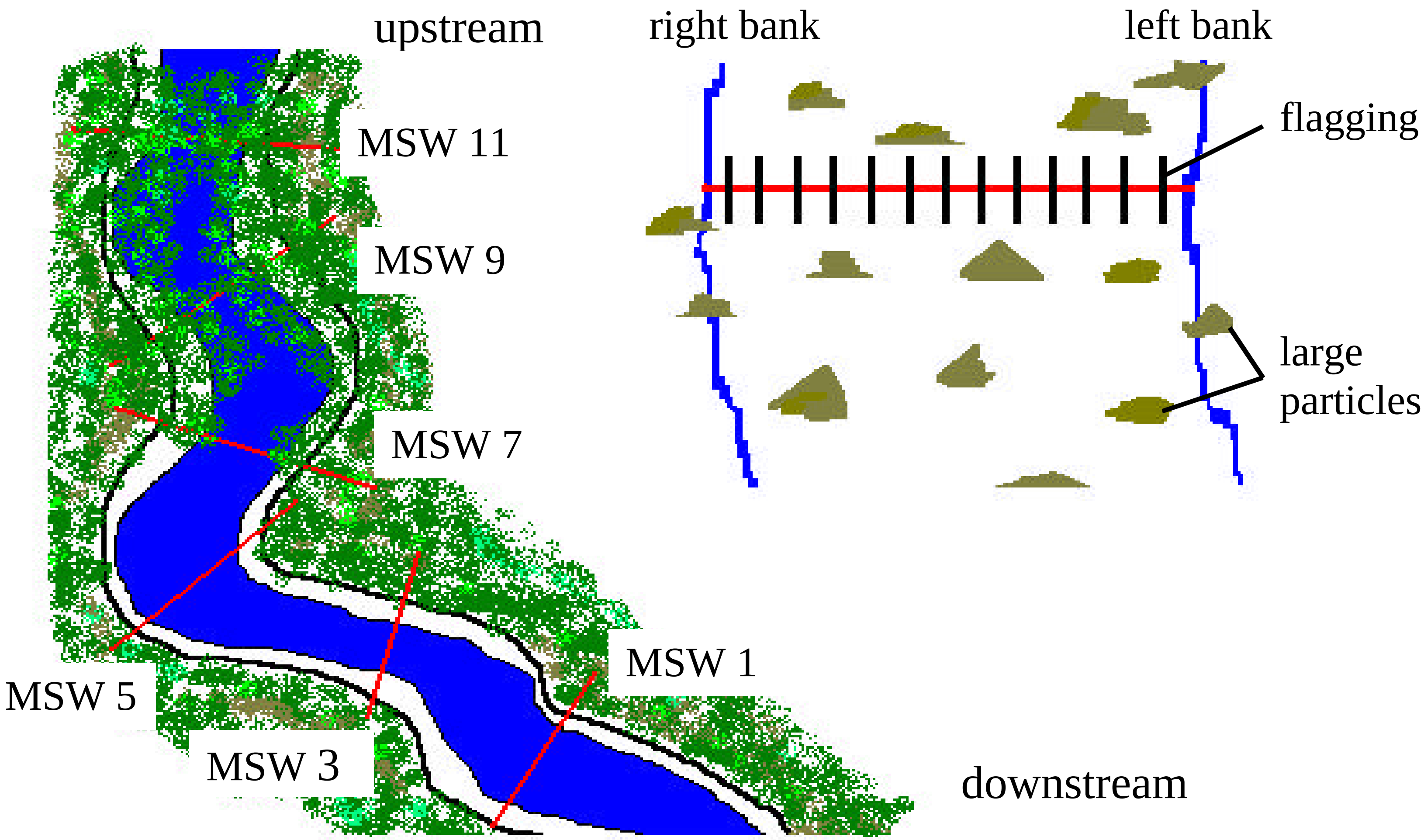


Figure 1.2.-Habitat sampling scheme. Cross-section transects began at MSW 1 and were spaced every two mean stream widths apart by following the thalweg upstream. Flagging on the cross channel tape was placed a distance of 10 % of the mean stream width apart. The first flagging was placed a distance of 5 % of the mean stream width from the water's edge on the right bank. Substrate, depth, and canopy closure were determined at each flagging point.

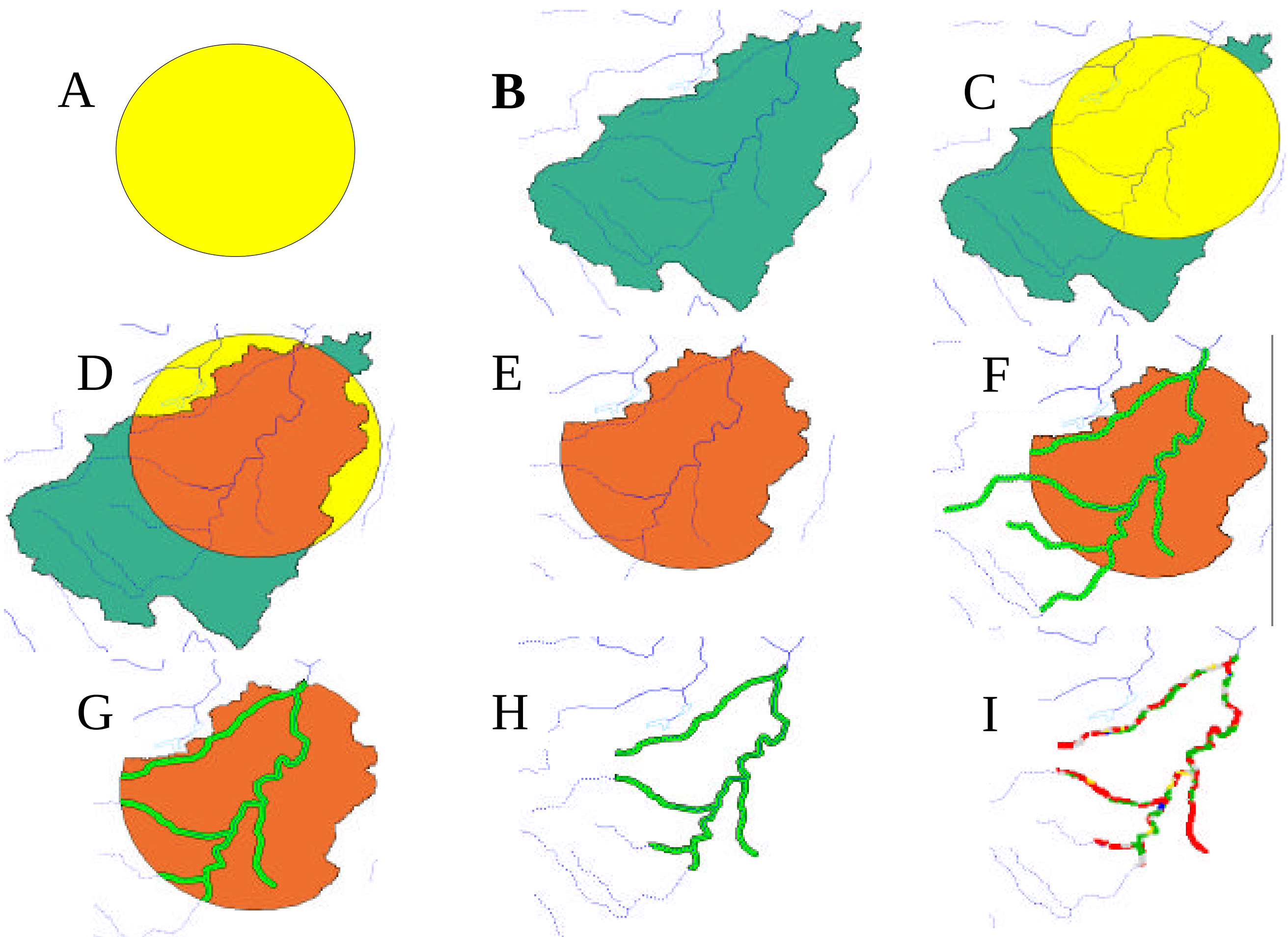


Figure 1.3.-Steps for creating partial watershed buffers. A circle is created with an area that is 5, 10, 20, 40, or 60 % of the mean watershed area for each watershed size class (A). This circle is positioned within a watershed (B) so that it encompasses the maximum amount of the watershed while still including the site (C). Areas outside of the circle (D) and the watershed (E) are excluded. The stream network is buffered (F) and the network outside of the circle is excluded (G). The remaining area (H) is the buffer to which land use data is added (I).

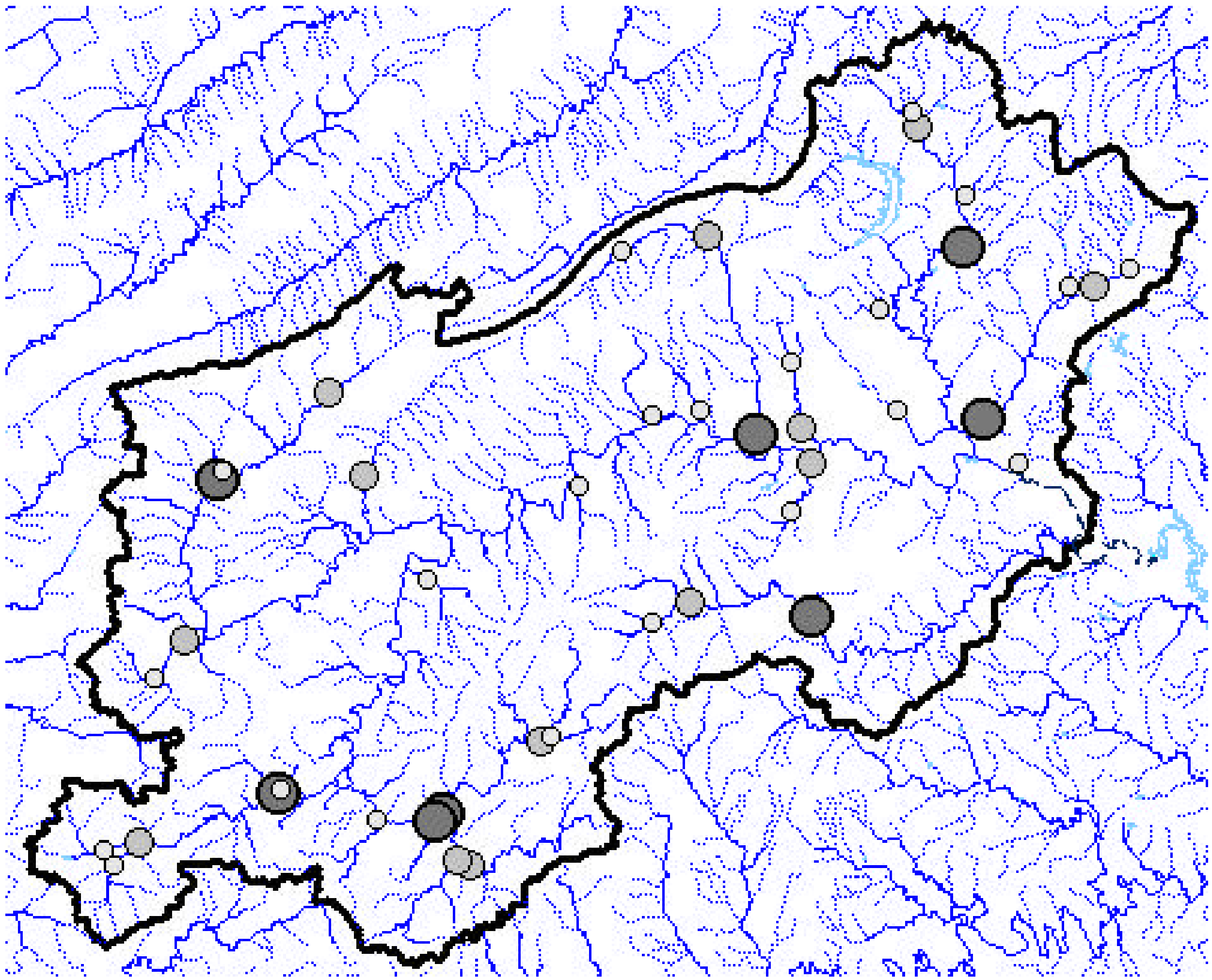


Figure 1.4.-Location of 43 sampling sites in the URRW. Small, light gray circles indicate sites on small streams, medium, gray circles indicate medium sites and large, dark gray circles indicate large sites. For both years combined, there were 22 small, 13 medium, and eight large sites.

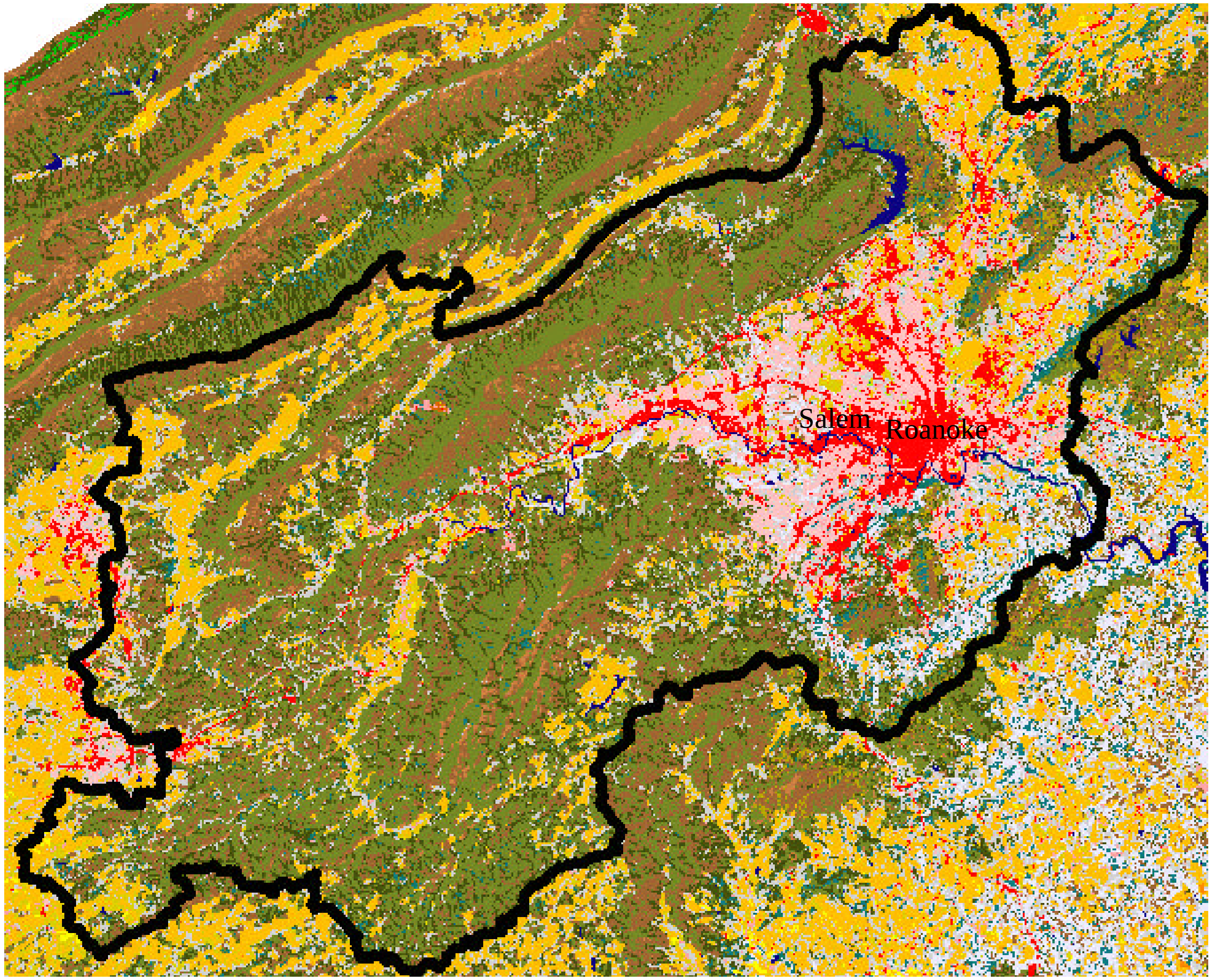


Figure 1.5.-Land use for the upper Roanoke River watershed. Green and brown areas are forests. Red is high intensity development and pink is low intensity development. Yellow indicates herb/ag land and gray areas are mixed/unknown.

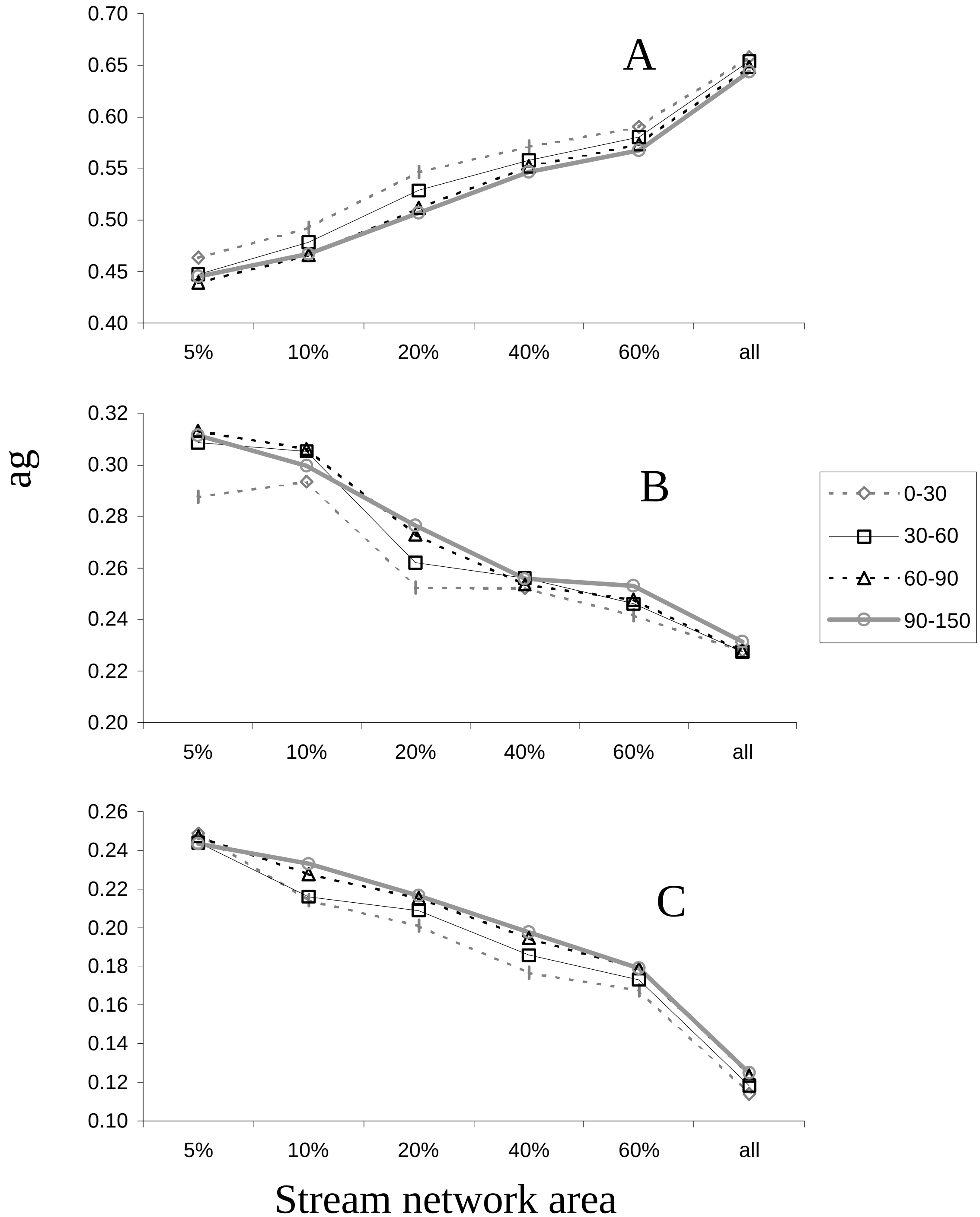


Figure 1.6.-Mean land use percentages for 43 sites at different buffer widths and stream network areas for A) forest, B) herb/ag, and C) disturbed. Buffer widths exclude land use in narrower buffers (i.e., 30-60 m instead of 0-60 m). Dependent axis scales have been reduced to only include the range of average land use variables. Mean percentages for entire watersheds were 67.4 % forest, 21.2 % herb/ag and 11.4 % disturbed.

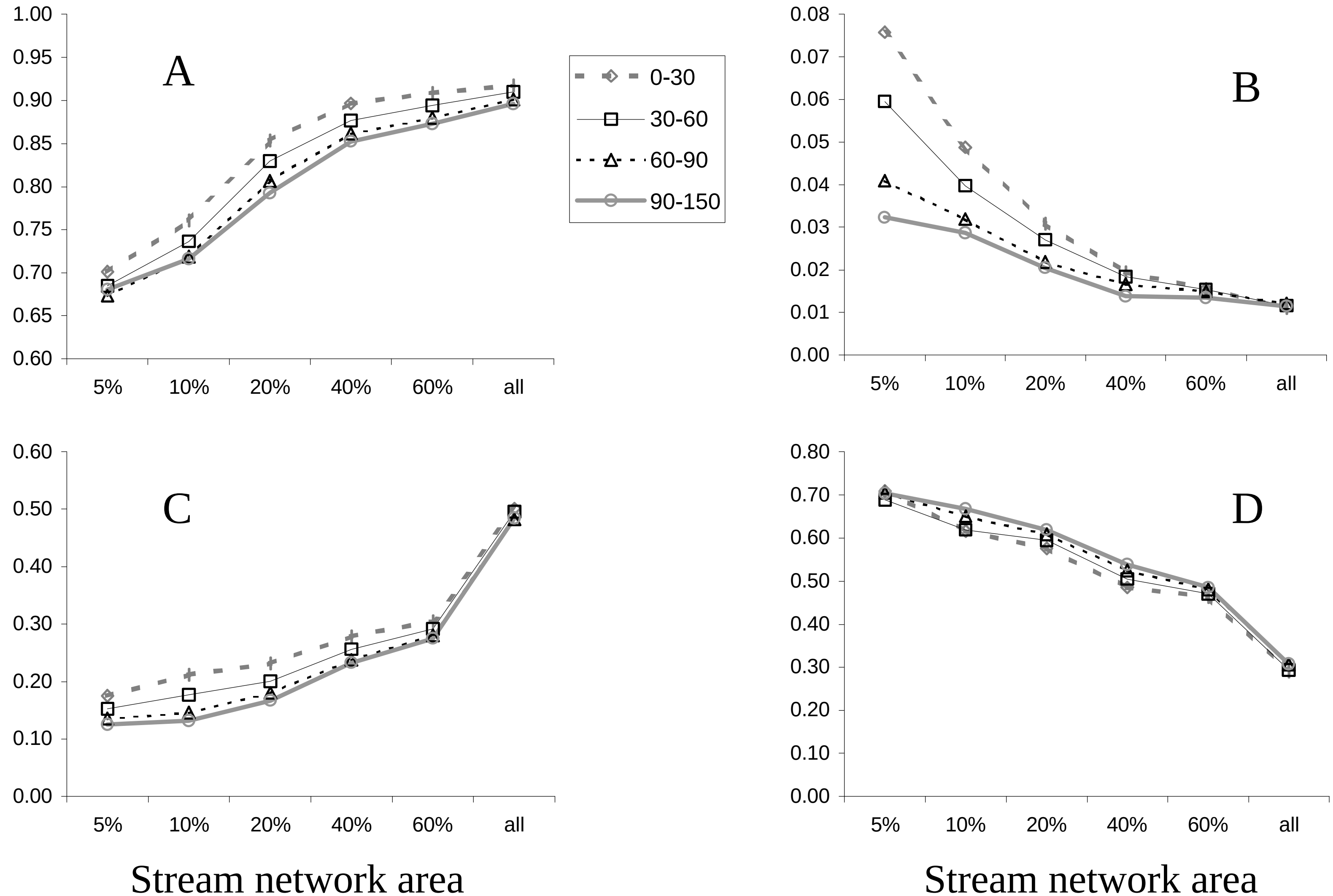


Figure 1.7.-Average amounts of forested and disturbed land for 14 sites that are well-forested (A and B) and 12 disturbed sites (C and D). Buffer widths exclude land use in narrower buffers (i.e. 60-90 m instead of 0-90 m). Dependent axis scales have been reduced to only include the range of values.

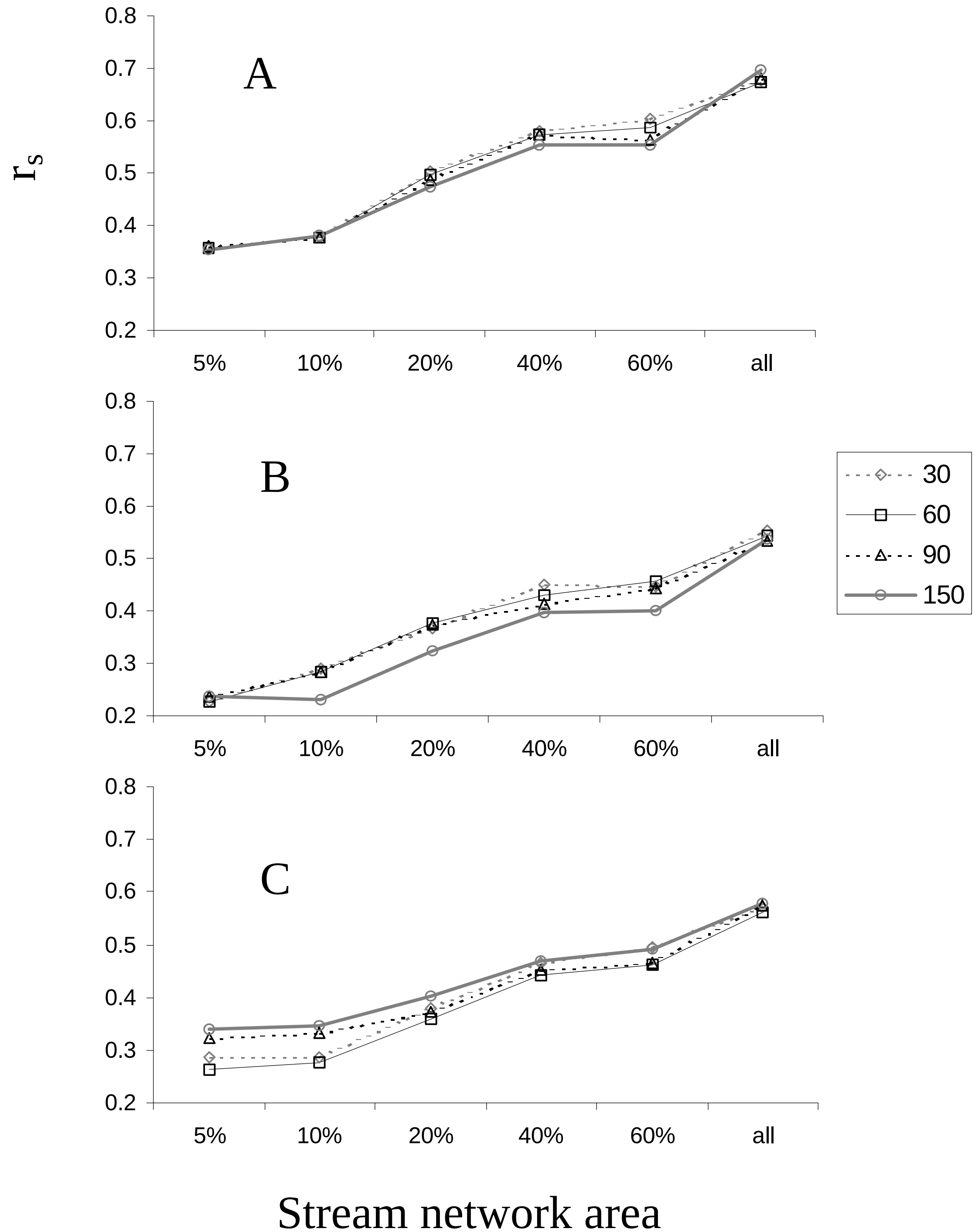


Figure 1.8.-Absolute values of Spearman's rank order correlation coefficients ($|r_s|$) comparing land use at 24 spatial scales with D50 values for 43 sites. Lines represent buffers from 30 to 150 m wide for different land uses: A) forest, B) herb/ag, and C) disturbed. Stream network area represents circles corresponding to 5 to 60 % of mean watershed areas and "all" represents entire stream network areas. For entire watershed correlations, r_s values are 0.698 for forest, -0.478 for herb/ag, and -0.597 for disturbed. Spearman coefficients < |0.30| are not significant at $\alpha = 0.05$.

CHAPTER 2. Fish Abundance and Distribution Related to Habitat and Watershed Conditions

Fish communities are simultaneously indicators of degradation and valued resources. Monitoring fish communities is important because fish communities reflect stream and watershed conditions by responding to degradation differently. The URRW has a rich diversity of stream fishes and has been thoroughly sampled in the past. Understanding the current distribution and abundance of various species is important for conservation efforts in the area as well as monitoring the effects of land use changes. By determining how different species respond to watershed and habitat conditions, species sensitive to degradation and factors that impact them can be identified.

OBJECTIVES

The goal of the second chapter was to summarize fish abundance and distribution at sites throughout the URRW and determine what factors affect their abundance and distribution. The first objective was to describe the fish community of URRW tributaries. I did this by reporting presence/absence data and density values for species. I also examined trends in species density from small to large sites. The second objective was to assess relationships of fishes with habitat and watershed variables to determine which factors may be responsible for their distribution and abundance. I expected fishes to respond to both natural and anthropogenic differences among sites. The third objective was to compare my collections with historical collections from URRW streams. I chose two sources of historical data which reported thorough sampling in Mason Creek (Jenkins and Freeman 1972) and in Back Creek (Hambrick 1973).

METHODS

Fish sampling

Several days after establishing a sampling site, my field crew and I collected fish with one upstream pass using a gas powered Smith-Root Model 15-D POW backpack electrofishing unit operating with DC current. We used one electrofishing unit for small and medium sites and two for large sites. For small and medium sites, at least three netters, in addition to the person electrofishing, collected fish. At least six people, in addition to the two people electrofishing, collected fish at large sites. For small and medium sites, we processed fish after sampling the entire site, but for large sites we processed fish at the middle and end of the transect.

We identified all fish to species level and counted them. The first 100 individuals of each species were measured to the nearest 5 mm total length (TL) class and examined for external anomalies. We identified external anomalies and recorded them for each individual according to Rankin (1989). Black spot, leech, and anchor worm anomalies could be light or heavy depending on the number of infestations. Leeches and anchor worms were considered light unless there were more than five of them on an individual. Black spot was considered light unless black spots covered the entire fish's body at a density equal to or greater than eye diameter (Rankin 1989). After processing, we released most fish, but kept some for other purposes.

I used minimum length limits to reduce the number of young of year fish included in analysis so that fish susceptibility to sampling would be similar throughout sampling seasons and species abundance would not be influenced by the high variability of young of year abundance (Angermeier and Karr 1986, Karr et al. 1987). Young of year fish were identified in the field and excluded from species abundance counts or excluded after analysis of length frequency data. I established length limits through length frequency analysis of my samples and personal observations. Minimum length was 30 mm TL for all species except the following: all percids, mottled sculpin (Cottus bairdi), and blacknose dace (Rhinichthys atratulus): 25 mm; torrent sucker (Thoburnia rhothoeca): 35 mm; Hypentelium sp.: 50 mm; white sucker (Catostomus commersoni): 55 mm; Micropterus sp.: 65 mm; and Moxostoma sp.: 80 mm TL. This list is not exhaustive because I did not encounter small individuals of other species that may have required minimum length limits greater than 30 mm TL.

Statistical analysis and comparisons

I transformed raw fish data into density values for some analyses. Density was expressed as number of fish per 100 m^2 and determined by dividing the number of fish of each species by the area sampled. Area sampled was the product of mean width determined during habitat sampling and site length determined when sites were demarcated prior to sampling.

Because water levels were much lower in 1999, I compared fish samples for each year to determine if fish collections were significantly different between 1998 and 1999. I compared the similarity of fish communities at sites sampled twice using percentage similarity. This test compares two communities by taking the sum of the minimum proportional abundance values of each species collected in either sample (Krebs 1989). Values range from zero to one, with a value of one indicating identical communities. I also compared number of species and total individuals collected at all sites grouped by watershed size using two sample, two-tailed t-tests. All sites were included except resampled sites, Dry Branch, and Upper Wilson Creek samples from 1999. These sites were excluded because Dry Branch was completely dry in 1999 and Upper Wilson Creek was sampled late in 1999.

To determine if density of fishes varied among streams of different sizes, I grouped sites by watershed size class and used the Kruskal-Wallis test, a nonparametric test which uses Wilcoxon scores and analyzes ranks of data for more than two samples (SAS Insititute, Inc. 1999). The null hypothesis for this test was that species density was equal for all watershed size classes while the alternative hypothesis was density differed among size classes. I grouped sites by watershed size class regardless of year collected and excluded species collected only once or twice from analysis. I used averaged density values for sites sampled twice and determined significance at $\alpha = 0.05$. To determine which watershed size class density values differed significantly from each other, I used the Tukey studentized range test. I related native species richness to stream width using simple linear regression to derive an equation to explain the relationship between native species richness and stream width. I set the intercept at 0 for this equation and excluded sites sampled twice because mean stream width was only determined during initial sampling and may have changed between years.

To further determine what factors affected species, I divided values for habitat and land use variables into two groups for each species. Variable values for one group represented all sites where a given species was collected while the other group was comprised of variable values for sites where the species was not collected. I designed this analysis to indicate variables that may play a large role in determining species presence/absence. I compared these 29 different variables from the two groups using a two-tailed t-test and quantified significance at $\alpha = 0.05$. For a species to be included in this analysis, at least 100 individuals must have been collected and the species must have been collected at 10 – 33 sites. Eleven species qualified. I excluded species collected at less than 10 sites because they were collected too infrequently and excluded species collected at more than 33 sites because their abundance and distribution indicated general ecological requirements.

I descriptively compared my fish collection data with two sources of historical data that reported fish collections throughout the 1900s and concentrated in the late 1960s and early 1970s. Jenkins and Freeman (1972) reported fish collections at seven sites along Mason Creek and Hambrick (1973) reported fish collections within the Back Creek watershed. Because both streams were well-sampled in the past, I was able to assess temporal changes in their fish communities during a time that development increased in both watersheds. I did not compare the samples statistically because I did not sample the exact sites that were sampled historically and actual numbers of fish were not reported for the historical data. Such historical fish community changes should indicate potential changes at current sites in the future as land conditions are altered.

RESULTS

Fish collections

My field crew and I collected 50 fish samples from 43 different sites during 1998 (Appendix D) and 1999 (Appendix E). Samples in 1998 were collected from 21 May to 13 August while 1999 samples were collected from 2 June to 11 August (with an additional sample on 16 September). In 1998, we collected samples at 30 different sites: 17 small, nine medium, and four large sites. In 1999, we collected 20 samples - 13 new sites and seven sites that were also sampled in 1998. We collected 10 samples at small sites (five were repeat samples), six samples at medium sites (two repeat samples) and four samples at large sites. Dry Branch, a small site, was sampled twice but was completely dry in 1998. Upper Wilson Creek, another small site, was resampled in September 1999, a month after we completed summer sampling. In summary, we sampled 22 small, 13 medium, and eight large sites and sampled seven of these twice.

During the two years, we collected 49 different species (35 native) from 9 families for a total of 54,809 individuals (Table 2.1). Species occurrence varied greatly among taxa; two species, fantail darter and bluehead chub, were collected in every sample (except Dry Branch when it was dry) while 10 species were only collected at one site. Species richness increased with stream width and watershed area, with small sites averaging 10.8 species, medium sites 13.8 species, and large sites 23.3 species. Species richness per sample ranged from five to 31 while total abundance ranged from 78 to 3,566 individuals. Abundance also varied greatly among species; 13,691 fantail darters and 13,269 mountain redbelly dace composed 49.2 % of the total collected. Conversely, only single individuals represented three species.

There was a general trend of species addition from small to large sites because all species found at small sites were also present at large sites. Conversely, many species were only collected at large sites. In addition to fantail darters and bluehead chubs, mountain redbelly dace, central stonerollers, crescent shiners, white suckers, and blacknose dace were found at 37 or more sites. With a few exceptions, these species were present at even the most degraded sites. Other species such as mottled sculpin, rosyside dace, and torrent sucker were abundant at some small sites but were not as ubiquitous and were much less tolerant of degraded conditions.

Although we collected most of the species found in past URRW samples, we failed to collect a few noteworthy native species: satinfin shiner, Cyprinella analostana; bull chub, Nocomis raneyi; quillback, Carpiodes cyprinus; shorthead redhorse, Moxostoma macrolepidotum; white catfish, Ameiurus catus; and shield darter, Percina peltata. These species appear to inhabit larger streams than I sampled (Jenkins and Burkhead 1993). Roanoke logperch are usually found at sites larger than I sampled but we did collect one at Lower Glade Creek. American eels and Roanoke bass, two species considered extinct in the URRW, were never collected.

None of the four species endemic to the Roanoke River, Roanoke hogsucker, riverweed darter, orangefin madtom, and bigeye jumprock, were very abundant at my sites. Roanoke hogsuckers were collected at 16 sites but were never abundant (Figure 2.1). Although collected at more sites than black jumprocks, Roanoke hogsuckers were less abundant and were only the fourth most abundant catostomids. However, they were more common than their congenerics, northern hogsuckers. Riverweed darters were more abundant than other endemic species and were the second most abundant darters. They were also collected at 16 sites and were collected at all large sites including Mason and Tinker Creeks, which were degraded and far less speciose than other large sites.

Orangefin madtoms and bigeye jumprocks were only collected at three large sites. Seventeen orangefin madtoms were collected at Lower Goose Creek but both species were rare or absent at all other sites. Only eleven bigeye jumprocks were collected at three sites. All four endemic species were collected together at Lower Goose and Lower Bottom Creeks. Four orangefin madtoms were collected in Elliott Creek, approximately six kilometers above its confluence with the South Fork. This find is noteworthy because a 1991 manure spill in the Elliott Creek headwaters eliminated fish downstream into the South Fork (Ensign et al. 1997). Sites within this affected area had fish communities similar to nearby streams and did not show any signs of this past impairment.

A total of 28,028 fish (51.1 % of all fish collected) were inspected for external anomalies and parasites. We recorded a total of 4,379 occurrences of 15 different external anomalies (Table 2.2). Light black spot infestation, the most common anomaly, occurred in 43 of 49 samples. A total of 3,294 fish (11.8 % of inspected fish) were infested with light black spot, thereby accounting for 75.2 % of all anomalies. Other external anomalies in decreasing order were light leech infestations, eroded fins, heavy black spot, lesions, deformities, light anchor worm, tumors, bloating, heavy leeches, swirled scales, fungus, popeye disease, blindness, and heavy anchorworm infestations. Light leech infestations, the second most common anomaly, only accounted for 2.7 % of anomalies and occurred on 118 inspected individuals in 20 samples.

Occurrence of anomalies varied among species (Table 2.3). Considering only species for which 100 or more individuals were measured, white shiners had the highest percentage of all anomalies with 83.2 % (390 of 469). Excluding light black spot infestations, redbreast sunfish had the highest percentage of anomalies with 7.5 % (25 of 335). Mottled sculpins were at the other extreme - no external anomalies were detected on 1,728 individuals.

Comparisons

Although discharge in 1999 was significantly lower than in 1998 for sites sampled twice (P = 0.021 with paired t-test), the fish communities at resampled sites were generally similar to one another between years (percentage similarity > 0.530 for all, Table 2.4). Excluding Purgatory Creek, all percentage similarity values were at least 0.80. The value was low for Purgartory Creek because proportional abundance of mountain redbelly dace and mottled sculpins changed drastically from 1998 to 1999. The species present were never the same between years for any of the five resampled sites. Fifteen times a species was present at a site one year but absent the other year. However, for these 15 cases no more than four individuals of these species were ever collected at the sites and six times the species were only represented by one individual. Mountain redbelly dace were more abundant at all sites in 1999. When I compared all samples (excluding repeated samples) from each year separated by watershed size class, there were no significant differences between years for number of species collected or total individuals ($P \geq 0.20$ for each watershed size class using two-tailed t-tests). Samples from 1998 were not significantly different from ones from 1999; therefore, all samples were combined for subsequent analysis.

Density values for 15 species differed significantly among watershed size classes using the Kruskal-Wallis test (Table 2.5). Fourteen of these species increased in density from small to large sites. Blacknose dace was the only species with significantly different densities that declined from small to large sites. The Tukey studentized range indicated which groups of density values were significantly different and generally corroborated Kruskal-Wallis test results. However, the Tukey test indicated that mountain redbelly dace were significantly more dense at medium sites than small sites, but the Kruskal-Wallis test did not indicate significant differences among all three groups. Several species were rarely collected at small and medium sites, but were noticeably more common at large sites (Roanoke darter, swallowtail shiner, cutlips minnow, and golden redhorse). Additionally, several native species (spottail shiner, mimic shiner, silver redhorse, v-lip redhorse, bigeye jumprock, orangefin madtom, and Roanoke logperch) were only collected at large sizes. These results corroborated results from correspondence analysis where many species were associated with large watersheds and only blacknose dace was associated with small watersheds. The linear equation comparing native species richness with stream width at 43 sites predicted an increase of 2.29 native species per meter of stream width (y = 2.29x, R^2 = 0.39, intercept set at 0, Figure 2.2). When Middle North Fork, a narrow site with high species richness and Lower Tinker Creek, a wide site with few species, were removed from analysis, the R^2 increased to 0.68 and the equation was slightly modified (y = 2.36x, intercept set a 0). This second equation, which excluded anomalous sites, predicted that native species richness would increase by more than two species with every one meter increase in stream width. Conversely, a reduction in stream width by one meter would reduce native species richness by more than two species based on this equation.

Comparisons of variables grouped by species presence or absence indicated many differences in conditions between sites where species were collected and sites where they were not (Table 2.6). Twenty-two of 29 variables were significantly different at $\alpha = 0.05$ for one or more of the 11 species. At least four variables were significantly different for each species, except torrent sucker, which had only two significantly different variables. Watershed areas differed significantly for presence/absence of eight species and mean widths differed significantly for seven species. In all cases, sites where species were collected were larger (i.e., wider streams and larger watershed areas) than sites where they were absent. There were more significant differences for land use variables in 20 %, 60 m buffers than for other groups of land use variables. Except for black jumprock comparisons, all significant differences for land uses at entire watershed scales were also significant at both buffer scales. Likewise, land use variables significantly different at 60 %, 60 m buffers differed significantly at 20 %, 60 m buffers.

Bluntnose minnows and redbreast sunfish were the only species that were more common at disturbed sites than forested sites (Table 2.6). Roanoke hogsuckers and mottled sculpins were more common at forested sites. Only four species, rosyside dace, white shiner, Roanoke hogsucker, and black jumprock, showed any significant difference for substrate variables and substrate was larger for sites where these species were present.

Comparing my samples with historical samples from Back Creek (Table 2.7, Hambrick 1973) and Mason Creek (Table 2.8, Jenkins and Freeman 1972) revealed mixed results. Species richness was generally higher in my Back Creek samples than historic samples at nearby sites and we found species at sites farther upstream than they had been collected previously. For example, we collected 14 species at Middle Back Creek but historically only six species were collected at a nearby site. We collected 25 species at Lower Back Creek but only 16 species were collected at the nearest upstream site and 19 species at the nearest downstream site historically. We also collected all species that were considered common historically. I was unable to detect any changes in the fish community in the past 30 years that could be attributed to land use.

My samples were very similar to historic samples in Mason Creek at the upper and middle portions of the stream but were less speciose than historic samples closer to the mouth. My field crew and I only collected 17 species at Lower Mason Creek. Historically, 31 species were collected near the mouth and 24 species were collected upstream of my site. Historically, bigeye jumprocks were collected in some samples near the mouth and Roanoke logperch were collected in some samples near the mouth and at a site upstream of my site. I failed to collect either in my Mason Creek samples. However, bigeye jumprocks and Roanoke logperch were not collected during all sampling events at each site (each site was sampled at least 3 times). In the 1940s, Roanoke logperch were collected at a site upstream from my sample site, but they were not collected in the only sample there in 1969. Additionally, rosefin shiners were considered abundant and swallowtail shiners common by Jenkins and Freeman (1972) along lower parts of Mason Creek that would include my middle and lower sites. I failed to collect any rosefin shiners and collected only one swallowtail shiner, which was present at my middle site. Four species were added to the species list of Back Creek: brown trout, rainbow trout, creek chub, and yellow bullhead. Only Roanoke hogsucker was added to the list of Mason Creek fishes.

DISCUSSION

Fantail darters and bluehead chubs were collected at every site and did not decline at degraded sites. Although they were not collected at every site, mountain redbelly dace, crescent shiners, and white suckers were well distributed at all watershed sizes and present at the most degraded sites such as Lick Run. All five species were tolerant of conditions at sites affected by development, agriculture, and sedimentation. Abundance and ubiquity of bluehead chubs likely enhanced cyprinid populations, which use chub mounds for spawning (Smith 1999). Without chub mounds for spawning, mountain redbelly dace and crescent shiners may have been less abundant or absent at heavily impacted sites.

A relatively large number of fish had black spot disease, which is caused by the larval stage of trematode parasites (Rankin 1989). Although most fish had very few of these small, black cysts, some were heavily infested. Black spot disease often occurred at sites where fish were not afflicted by other anomalies, or anomalies were very rare. Black spot trematodes have a complex life cycle that includes snails and piscivorous birds (Rankin 1989). These intermediate hosts may be absent or rare at heavily impacted streams and as a result black spot could be rarer than at less disturbed sites. Sanders et al. (1999) recommend excluding black spot infestations in IBI applications because they do not reflect degradation similarly to other external anomalies. My results support their recommendation.

My results showed that species were differentially affected by external anomalies. Some fishes, such as white shiners and redbreast sunfish, often had anomalies while others, such as mottled sculpin, never had any external anomalies. For parasites (i.e., black spot infestations, leeches, and anchor worms), such differences are likely due to life history strategies of the parasites and their hosts. Also, some species may survive longer with external anomalies while others may die more easily and therefore are rarely encountered with anomalies.

Studies such as mine that examine external anomalies to assess the health of individual fish calculate the proportion of external anomalies at a site without regard for the fish assemblage and compare it with other sites that may have different fish assemblages. Such studies assume that all fishes are equally susceptible to external anomalies and also assume that survival with external anomalies is equal among species. Streams containing species highly susceptibility to external anomalies that are better able to survive with anomalies will likely have higher proportions of external anomalies than streams only containing species rarely found with them. The effect of differential susceptibility and survival on anomaly assessments can be offset by only including species that are present at all sites or are shown to be similarly affected by anomalies. External anomalies are important indicators of stream integrity and future studies should consider issues of differential susceptibility and survival to determine if current methods of assessing external anomalies can be improved. Reducing the number of species examined, such as only species found at most sites, may be the first step toward reducing the variability of external anomalies caused by differential susceptibility and survival.

Species abundance and occurrence differed at sites sampled in both years, but the differences were not significant. The differences could be a result of drier conditions in 1999 or the inherent biases of electrofishing. With only one pass depletion, assuming that all species present are collected is presumptuous and likely incorrect. Discharge was significantly lower at all five sites in 1999 and some species showed consistent trends between years (Table 2.4). Creek chubs were more common at sites in 1999 while Roanoke hogsuckers were collected at more sites in 1998. Creek chubs are considered tolerant species (Smogor 1996) and occurred more often at small sites while Roanoke hogsucker density increased significantly with increased stream size. Mountain redbelly dace abundance increased at all five sites from 1998 to 1999 samples, possibly because they tolerate physicochemical changes such as dissolved oxygen, temperature, and pH better than other cyprinids of the URRW (Matthews and Styron 1981). In small headwater streams, physicochemical conditions have a large effect on fish distribution and abundance, especially during droughts (Smale and Rabeni 1995). Regardless of these differences, these sites were not significantly different between years and my samples are representative of the sites, at least for summer base flow conditions.

Physicochemical conditions in small streams play a large role in determining fish community structure (Matthews and Styron 1981, Schlosser 1990, Smale and Rabeni 1995). Such factors may have more influence than land use conditions on fish communities in small streams. Tolerance of physicochemical variability is a possible explanation for increased abundance of mountain redbelly dace at resampled sites in 1999. Also, the diversity of fish and similarity of the community in 1998 and 1999 at Mill Creek is likely due to spring inputs that regulate temperature and discharge. Biologists sampling fish communities should consider the role of physicochemical conditions when identifying streams to sample (Schlosser 1990, Smale and Rabeni 1995). Fish communities in small streams may not reflect watershed land use conditions as much as fishes in larger streams due to the influence of temperature, dissolved oxygen, and pH. Therefore, future studies should monitor these conditions or avoid sampling sites where they will play a large role in influencing fish communities if linking land use with fish communities is the research objective.

Fish assemblages changed from small to large sites, usually by the addition of species. Several species were only collected at large sites and many species showed increased abundance; therefore, separating sites by watershed area was an important step. Longitudinal additional of species has been well documented for streams in North America (e.g., Sheldon 1968, Whiteside and McNatt 1972, Evans and Noble 1979) and for Back Creek (Hambrick 1973) and Mason Creek (Jenkins and Freeman 1972) in the URRW. Most rare and specialized fishes of the URRW were only collected at large sites or the majority of individuals were collected at large sites. For example, orangefin madtoms, considered threatened by the state of Virginia, and Roanoke logperch, which are federally endangered, were only collected at large sites. Bigeye jumprocks, which have a strong preference for deep, unsilted habitat that results in a patchy distribution (Jenkins and Burkhead 1993), were only collected at large sites. These species are also found in streams larger than I sampled, but their presence at my sites indicated suitable conditions. All three species are considered endangered, threatened, or of special concern due to destruction, alteration, or reduction of their habitat and range (Williams et al. 1989). Identifying streams where these species live is important for future planning because such areas should be protected from development that could cause them to become rarer (Sheldon 1988, Angermeier 1995).

Increases in impervious surfaces affect stream hydrology by increasing the frequency of bankfull discharges and reducing base flow (Klein 1979). Reduced base flows can cause summer and early autumn flows to be less than under pristine watershed conditions. Many species increased in abundance with increased stream width, which is indicative of higher discharge and deeper, more diverse habitat (Gorman and Karr 1978). If base flows are reduced, species richness will decline and the range of species that require wider, deeper streams will be reduced. The linear equation comparing native species richness with mean width predicted an increase of 2.4 native species per meter increase in mean width. Conversely, a one meter reduction in stream width would result in the loss of two or more species based on the linear equation. Such stream size reductions resulting from increases in impervious surfaces could drastically reduce the distribution of species affected by stream size.

Stream reaches downstream of areas containing orangefin madtoms, Roanoke logperch, and bigeye jumprocks will likely contain them as well because they are influenced by the same land use conditions as those reaches upstream. For example, the South Fork of the Roanoke River is known to contain these species (Jenkins and Burkhead 1993). Forested landscapes in Bottom and Goose Creek watersheds (> 85 % forest for each) not only resulted in suitable conditions for several of these species at my sites, they contribute to good conditions downstream in the South Fork. Conversely, none of these species were collected at my sites on Mason Creek, Tinker Creek, Back Creek or the North Fork. Evidently, conditions in these watersheds will have to improve for these species to live there.

Comparisons of my samples with historical Back Creek samples indicated that the fish community has either increased in species richness and species are found farther upstream than in the past, or previous samples lacked the intensity to collect all species present. Additional reasons for differences include sampling methods (seining versus electrofishing) and differences in the habitat quality and diversity of sampled sites. Normal, annual fluctuations in populations could also account for the differences in the samples. Comparisons of samples from the upper and middle portions of Mason Creek indicated that the fish communities today are very similar to what they were 30 years ago. However, comparisons of my Lower Mason Creek site with historical data indicated that some species are currently less common. The upper and middle sites are primarily forested and have little disturbance, but development increases considerably upstream of my site near Salem.

The absence of rosefin shiners and swallowtail shiners from the Lower Mason Creek site is disconcerting because they were considered abundant and common, respectively, at adjacent sites in the past (Jenkins and Freeman 1972). Their absence could be related to low discharge at the site due to extremely dry conditions during the summer of 1999. It is possible that these pool-dwelling species moved downstream to deeper water. In 1998, I collected Notropis sp., including swallowtail shiners, at three of four large sites, but I failed to collect any Notropis sp. at four large sites in 1999. To determine if this difference was related to lower discharge in 1999, I resampled Lower Glade and Lower Back Creeks with seines in an attempt to verify the presence of Notropis sp. at these sites. Notropis sp. were collected at both sites, reducing the likelihood that discharge had an effect on their distribution. These findings fail to support the theory that rosefin shiners and swallowtail shiners moved elsewhere due to drought conditions and that their absence from Mason Creek is only temporary.

Rosefin shiners and swallowtail shiners occurred most often at my large sites and are also found in larger streams (Jenkins and Burkhead 1993). Intolerance of physicochemical changes, such as dissolved oxygen, temperature, and pH, may have affected rosefin shiners, given low water levels in 1999 (Matthews and Styron 1981). However, Jenkins and Freeman (1972) attributed their collecting success to low water levels during most samples, which weakens the low water level theory. In conclusion, although both species appear to prefer larger streams that I sampled, they were more abundant during past samples in Mason Creek than during my 1999 sample and the cause of their apparent reduction is unknown.

Although rosefin shiners occur in several drainages and ecoregions and are currently somewhat common throughout their ranges (Jenkins and Burkhead 1993), abundance and range reduction could signal the extirpation of these species from a portion of their range. Such small incremental range reductions are usually precursors of more wide spread extirpations, which can eventually culminate with species extinction (Angermeier 1995). Although, ecological plasticity of rosefin shiners in terms of trophic ecology and habitat preference (Surat et al. 1982) reduces the likelihood of extirpation, continued monitoring of their status in Mason Creek is warranted.

Jenkins and Freeman (1972) ended their discussion of the fishes of Mason Creek by concluding that despite urbanization in the lower portion of the drainage, the fish community was relatively similar to that described by David Starr Jordan in 1889. My results suggest that the fish community has changed noticeably since collections in the middle and late 1900s. Differences in Lower Mason Creek fish collections should be seen as warning signs of possible small scale extirpations and range reductions that can result from urbanization. Studies comparing current and historical fish community data have linked land use changes with reduced species richness. Species richness in Tuckahoe Creek near Richmond, Virginia declined from 32 to 23 species over 32 years as development increased in the watershed. Abundance of species declined at all sites as well (Weaver and Garman 1994). Three species were apparently extirpated from the Roaring River, Tennessee based on sampling in 1972 and 1986; two of these were especially sensitive to siltation. Watershed and instream conditions deteriorated over that time period due to several anthropogenic impacts such as dredging and construction activities (Crumby et al. 1990). These studies show that deteriorating watershed conditions can cause species, especially ecological specialists that are sensitive to perturbations, to be extirpated. Such historical losses indicate the likelihood of future losses caused by continued landscape alteration.

The issue of concentrated or dispersed development on a large spatial scale is an important consideration for rare species as well as other fish community members. The entire URRW is nearly 14 % disturbed but disturbed land is patchily distributed because it is concentrated around the urban centers of Salem and Roanoke. Herbaceous and agriculture land constitutes over 16 % of the land but is likewise patchily distributed. As a result, some watersheds are well-forested while others are impacted by land use changes. If land use was equally distributed throughout the URRW so that every watershed contained 14 % disturbed and 16 % herb/ag land, stream habitat quality and fish communities would be more similar to each other. As a result, there would be no streams impacted so much that only a few species could live there. Unfortunately, fish that require pristine conditions would likely decline because there would not be any minimally impacted watersheds either.

Protecting areas inhabited by rare species will not only protect rare species, it will protect fish assemblages and ecological processes necessary for continued stream integrity. Such aquatic communities provide examples of proper stream structure and function that resource managers can strive to replicate in degraded streams nearby. Urban sprawl spreads disturbance over a large land area and resembles dispersed development more than concentrated development. As a result, fewer areas represent least disturbed conditions and species that require such conditions will likely decline. However, the impacts of urban sprawl and dispersed development can be mitigated by considering development within a watershed context. If a watershed is already heavily impacted by urbanization or other degradation, future development in that watershed is unlikely to affect rare species because previous impacts probably eliminated them. However, sprawl that spreads into a watershed that is somewhat pristine will likely impact and possibly eliminate rare species.

Another consideration for placement of development involves the size of the impacted stream and its position within the watershed. In the URRW, species thought to be declining are not currently found in small and medium streams and likely were absent from these streams historically as well. Additionally, small streams far exceed larger streams and rivers in terms of cumulative length; therefore, fewer opportunities exist to preserve large streams and rivers void of anthropogenic impacts. With these facts in mind, disturbance could be concentrated on small streams. Unfortunately, small streams form the headwaters of large streams and land use practices have far reaching downstream effects. However, streams such as Peters, Mudlick, and Wolf Creeks, which flow directly into the mainstem of the Roanoke River, contribute a relatively small amount of water and sediment compared to mainstem tributaries farther upstream. In other words, the downstream effect of land use conditions within these watersheds is minimal because they flow directly into a much larger stream. Headwater tributaries of streams large enough to contain rare species should not be developed with this same logic because of the connectedness and far reaching downstream impacts of land use activities.

In summary, concentrating future development in already impacted watersheds is better than developing somewhat pristine watersheds because rare species are probably already absent from impacted watersheds but might be eliminated if development occurs in watersheds that are still somewhat pristine. It is better to have some streams that are minimally impacted because some species are very sensitive to impacts and cannot survive even small amounts of disturbance. Such species are already absent from impacted streams so further development will not impact them. Also, small streams are less likely to contain rare species so development in these watersheds will have less impact than development near larger streams. However, one must remember stream network connectivity and the far-reaching downstream impacts of land use changes that may affect large streams.

Table 2.1.-All fishes collected at 43 sites, their occurrence at different size sites and all sites combined, and their total abundance.

Common Name	Scientific Name	Number of Occurrences				Total
		Small Sites	Medium Sites	Large Sites	Total	Collected
rainbow trout	Oncorhynchus mykiss	2	5	4	11	33
brown trout	Salmo trutta	2	3	4	9	56
brook trout	Salvelinus fontinalis	1	0	1	2	2
chain pickerel	Esox niger	1	2	0	3	8
central stoneroller	Campostoma anomalum	20	13	8	40	2350
rosyside dace	Clinostomus funduloides	14	11	8	32	3316
common carp	Cyprinus carpio	0	1	0	1	2
cutlips minnow	Exoglossum maxilingua	1	1	4	6	95
white shiner	Luxilus albeolus	4	3	7	14	661
crescent shiner	Luxilus cerasinus	19	13	8	39	3008
rosefin shiner	Lythrurus ardens	1	2	3	6	76
bluehead chub	Nocomis leptocephalus	23	13	8	43	5634
spottail shiner	Notropis hudsonius	0	0	1	1	185
swallowtail shiner	Notropis procne	0	1	3	4	77
mimic shiner	Notropis volucellus	0	0	1	1	1
mtn. redbelly dace	Phoxinus oreas	22	13	7	42	13269
bluntnose minnow	Pimephales notatus	3	3	4	10	491
fathead minnow	Pimephales promelas	0	1	0	1	4
blacknose dace	Rhinichthys a. atratulus	21	10	6	37	2811
longnose dace	Rhinichthys cataractae	1	0	1	2	22
creek chub	Semotilus atromaculatus	4	3	1	8	36
white sucker	Catostomus commersoni	20	13	8	41	1105
northern hogsucker	Hypentelium nigricans	2	2	6	10	38
Roanoke hogsucker	Hypentelium roanokense	4	6	6	16	264
silver redhorse	Moxostoma anisurum	0	0	2	2	10
golden redhorse	Moxostoma erythrurum	1	1	3	5	33
v-lip redhorse	Moxostoma pappillosum	0	0	1	1	5
bigeye jumprock	Scartomyzon ariommus	0	0	3	3	11
black jumprock	Scartomyzon cervinus	2	2	7	11	322
torrent sucker	Thoburnia rhothoeca	12	8	6	26	2568
yellow bullhead	Ameiurus natalis	0	0	1	1	2
brown bullhead	Ameiurus nebulosus	0	1	0	1	6
orangefin madtom	Noturus gilberti	0	0	3	3	22
margined madtom	Noturus insignis	4	11	7	22	657
eastern mosquitofish	Gambusia holbrooki	1	0	0	1	2
mottled sculpin	Cottus bairdi	8	4	4	16	2781
rock bass	Ambloplites rupestris	2	4	6	12	120
redbreast sunfish	Lepomis auritus	5	6	6	17	344
green sunfish	Lepomis cyanellus	2	1	2	5	14
pumpkinseed	Lepomis gibbosus	1	1	0	2	11
bluegill sunfish	Lepomis macrochirus	6	3	2	11	65
smallmouth bass	Micropterus dolomieu	1	2	6	9	53
spotted bass	Micropterus punctulatus	1	0	0	1	1
largemouth bass	Micropterus salmoides	3	2	1	6	12
fantail darter	Etheostoma flabellare	22	13	8	43	13691
johnny darter	Etheostoma nigrum	2	2	3	7	71
riverweed darter	Etheostoma podostemone	4	4	8	16	402
Roanoke logperch	Percina rex	0	0	1	1	1
Roanoke darter	Percina roanoka	0	1	7	8	61

Table 2.2.-Anomalies for all individuals inspected. BL = light black spot, CL = light leeches, E = eroded fin, BH = heavy black spot, L = lesion, D = deformity, AL = light anchor worm, T = tumor, BO = bloating, W = swirled scales, CH = heavy leeches, F = fungus, Y = popeye disease, N = blind, and AH = heavy anchor worm. Numbers shown are individual fish.

Species	Number Measured	BL	CL	E	BH	L	D	AL	T	BO	W	CH	F	Y	N	AH	Total
rainbow trout	33	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1
brown trout	56	0	0	1	0	0	0	0	0	0	0	0	0	0	0	0	1
brook trout	2	0	0	0	0	0	1	0	0	0	0	0	0	0	0	0	1
chain pickerel	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
central stoneroller	1700	736	29	16	8	3	2	0	0	0	0	0	0	0	0	0	794
rosyside dace	1786	4	0	8	0	0	0	0	0	0	0	0	0	0	0	0	12
common carp	2	0	0	2	0	1	0	0	0	0	0	0	0	0	0	0	3
cutlips minnow	95	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8
white shiner	469	373	1	1	11	3	1	0	0	0	0	0	0	0	0	0	390
crescent shiner	1772	758	1	14	6	2	2	0	0	1	0	0	0	0	0	0	784
rosefin shiner	70	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2
bluehead chub	3370	1042	49	16	3	9	6	8	6	0	0	0	0	1	2	0	1142
spottail shiner	97	25	0	0	0	2	0	0	0	0	0	0	0	0	0	0	27
swallowtail shiner	77	22	0	0	0	9	0	0	0	0	0	0	0	0	0	0	31
mimic shiner	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
mtn. redbelly dace	4423	84	2	20	1	2	1	6	2	1	0	0	1	1	0	0	121
bluntnose minnow	389	126	0	1	1	2	0	0	0	0	0	0	0	0	0	0	130
fathead minnow	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
blacknose dace	1871	356	2	8	16	1	3	1	0	3	0	0	0	0	0	0	390
longnose dace	22	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	4
creek chub	36	14	0	1	1	1	0	0	0	0	0	0	0	0	0	0	17
white sucker	1101	39	0	6	0	11	1	2	4	0	2	0	1	0	0	0	66
northern hogsucker	38	3	0	0	0	2	0	1	0	0	0	0	0	0	0	1	6
Roanoke hogsucker	264	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8
silver redhorse	10	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1
golden redhorse	33	12	0	0	0	0	0	0	0	0	1	0	0	0	0	0	13
v-lip redhorse	5	3	0	0	0	0	0	0	0	0	1	0	0	0	0	0	4
bigeye jumprock	11	3	0	0	0	0	0	0	0	0	0	0	0	0	0	0	3
black jumprock	322	77	0	0	0	11	0	0	2	0	0	0	0	0	0	0	90
torrent sucker	1514	76	2	1	0	3	2	0	1	0	0	0	0	0	0	0	85
yellow bullhead	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
brown bullhead	6	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
orangefin madtom	22	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
margined madtom	656	0	0	1	0	1	0	0	0	0	0	0	0	0	0	0	2
eastern mosquitofish	2	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
mottled sculpin	1728	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
rock bass	120	9	1	2	0	1	0	0	0	0	0	0	0	0	0	0	13
redbreast sunfish	335	1	17	2	0	1	1	0	0	0	0	4	0	0	0	0	26
green sunfish	14	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
pumpkinseed	11	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
bluegill sunfish	65	1	6	0	0	1	0	0	0	0	0	0	0	0	0	0	8
smallmouth bass	53	2	2	1	0	2	1	3	0	0	0	0	0	0	0	0	11
spotted bass	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
largemouth bass	12	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
fantail darter	4886	125	6	4	21	0	1	0	0	0	0	0	0	0	0	0	157
johnny darter	71	0	0	3	0	0	0	0	0	0	0	0	0	0	0	0	3
riverweed darter	402	10	0	2	6	0	0	0	0	0	0	0	0	0	0	0	18
Roanoke logperch	1	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Roanoke darter	60	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Total	28028	3924	118	111	74	68	22	21	15	5	4	4	2	2	2	1	4372

Table 2.3.-Anomaly summary for species where 100 or more individuals were collected. BL = light black spot, CL = light leeches, E = eroded fin, BH = heavy black spot, L = lesion, D = deformity, AL = light anchor worm, T = tumor, BO = bloating, W = swirled scales, CH = heavy leeches, F = fungus, Y = popeye disease, N = blind, and AH = heavy anchor worm. Numbers shown for each anomaly are individual fish.

	External Anomalies															Total	All anomalies	Fish	% w/ any	% w/ any
Species	BL	E	T	L	D	BH	CL	CH	AL	AH	Y	F	N	W	BO	anomalies	except BL	measured	anomaly	except BL
fantail darter	125	4	0	0	1	21	6	0	0	0	0	0	0	0	0	157	32	4886	3.2%	0.7%
mtn. redbelly dace	84	20	2	2	1	1	2	0	6	0	1	1	0	0	1	121	37	4423	2.7%	0.8%
bluehead chub	1042	16	6	9	6	3	49	0	8	0	1	1	2	0	0	1143	101	3370	33.9%	3.0%
blacknose dace	356	8	0	1	3	16	2	0	1	0	0	0	0	0	3	390	34	1871	20.8%	1.8%
rosyside dace	4	8	0	0	0	0	0	0	0	0	0	0	0	0	0	12	8	1786	0.7%	0.4%
crescent shiner	758	14	0	2	2	6	1	0	0	0	0	0	0	0	1	784	26	1772	44.2%	1.5%
mottled sculpin	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	1728	0.0%	0.0%
central stoneroller	736	16	0	3	2	8	29	0	0	0	0	0	0	0	0	794	58	1700	46.7%	3.4%
torrent sucker	76	1	1	3	2	0	2	0	0	0	0	0	0	0	0	85	9	1514	5.6%	0.6%
white sucker	39	6	4	11	1	0	0	0	2	0	0	0	0	2	0	65	26	1101	5.9%	2.4%
margined madtom	0	1	0	1	0	0	0	0	0	0	0	0	0	0	0	2	2	656	0.3%	0.3%
white shiner	373	1	0	3	1	11	1	0	0	0	0	0	0	0	0	390	17	469	83.2%	3.6%
riverweed darter	10	2	0	0	0	6	0	0	0	0	0	0	0	0	0	18	8	402	4.5%	2.0%
bluntnose minnow	126	1	0	2	0	1	0	0	0	0	0	0	0	0	0	130	4	389	33.4%	1.0%
redbreast sunfish	1	2	0	1	1	0	17	4	0	0	0	0	0	0	0	26	25	335	7.8%	7.5%
black jumprock	77	0	2	11	0	0	0	0	0	0	0	0	0	0	0	90	13	322	28.0%	4.0%
Roanoke hogsucker	8	0	0	0	0	0	0	0	0	0	0	0	0	0	0	8	0	264	3.0%	0.0%
rock bass	9	2	0	1	0	0	1	0	0	0	0	0	0	0	0	13	4	120	10.8%	3.3%

Table 2.4.-Comparison of fish community at five resampled sites. Similarity of samples between years determined using percentage similarity (Krebs 1989). Numbers shown are counts for each sample.

	Purgatory Creek			Mill Creek			Wolf Creek			Middle Back Creek			Lick Fork		
Species	1998	1999	mean	1998	1999	mean	1998	1999	mean	1998	1999	mean	1998	1999	mean
rainbow trout				4	8	6							9	1	5
brown trout				7	6	6.5				4		2	8	6	7
brook trout				1		0.5									
central stoneroller	1	4	2.5	8	9	8.5	14	17	15.5	12	39	25.5	2		1
rosyside dace	20	32	26	66	151	108.5		4	2		3	1.5	156	152	154
cutlips minnow	1		0.5												
white shiner	16	5	10.5	1	6	3.5	2		1						
crescent shiner	10	20	15	1	5	3	3	4	3.5	127	169	148	7	8	7.5
bluehead chub	41	27	34	34	31	32.5	42	31	36.5	193	175	184	120	81	100.5
mtn. redbelly dace	39	270	154.5	11	25	18	145	248	196.5	420	758	589	83	143	113
bluntnose minnow							2		1						
blacknose dace	19	67	43	11	6	8.5	68	112	90	53	83	68	14	21	17.5
creek chub							1	9	5		1	0.5		2	1
white sucker	12	4	8	4		2	41	24	32.5	51	90	70.5	14	12	13
northern hogsucker				1	1	1									
Roanoke hogsucker	2		1							18	25	21.5	1		0.5
black jumprock				2	2	2				8	7	7.5			
torrent sucker	10	19	14.5	8	41	24.5							48	19	33.5
margined madtom	7	2	4.5							23	22	22.5	67	56	61.5
mottled sculpin	60	2	31	495	355	425							106	108	107
largemouth bass											1	0.5			
fantail darter	55	62	58.5	118	128	123	77	148	112.5	272	162	217	88	80	84
riverweed darter					1	0.5									
percentage similarity			0.530			0.799			0.864			0.815			0.876
total abundance	293	514	403.5	772	775	773.5	395	597	496	1181	1535	1358	723	689	706
number of species	14	12	14	16	15	17	10	9	11	11	13	14	14	13	15
discharge (m^3/s)	0.116	0.026		0.263	0.078		0.018	0.005		0.125	0.006		0.119	0.013	

Table 2.5.-Mean density values separated by watershed size class for species collected at three or more sites. Significance using the Kruskal-Wallis test determined at α = 0.05. The Tukey studentized range test was used to determine differences among the three groups. Groups with different letters are significantly different. General trends of increased or decreased density from small to large sites are indicated for densities significantly different with the Kruskal-Wallis test.

	Mean Density Values				
Species	Small	Medium	Large	*P*-value	Trend
rainbow trout	0.06	0.07	0.04	0.084	
brown trout	0.06	0.06	0.16	0.096	
chain pickerel	0.00	0.07	-	0.314	
central stoneroller	4.03	6.26	5.54	0.082	
rosyside dace	5.03	9.64	6.10	0.256	
cutlips minnow	<0.01 A	0.04 A	0.48 B	0.003	increase
white shiner	0.09 A	0.58 A	3.79 B	<0.001	increase
crescent shiner	3.71	8.73	6.52	0.007	
rosefin shiner	0.01	0.16	0.26	0.079	
bluehead chub	10.34	17.87	10.52	0.275	
swallowtail shiner	-	0.01	0.31	0.009	increase
mountain redbelly dace	31.87	49.76 A	13.47 B	0.053	
bluntnose minnow	0.43	1.07	1.61	0.113	
blacknose dace	12.92	5.55	1.10	0.013	decrease
creek chub	0.15	0.04	<0.01	0.808	
white sucker	1.96	2.87	1.75	0.943	
nothern hogsucker	0.02	0.06	0.12	0.002	increase
Roanoke hogsucker	0.20	0.60	0.81	0.014	increase
golden redhorse	0.01	0.14	0.15	0.048	increase
bigeye jumprock	- A	- A	0.06 B	0.001	increase
black jumprock	0.03 A	0.35 A	1.48 B	<0.0001	increase
torrent sucker	5.85	9.05	5.81	0.470	
orangefin madtom	- A	- A	0.10 B	0.001	increase
margined madtom	0.12 A	1.88 B	1.62 B	<0.0001	increase
mottled sculpin	9.72	8.17	3.09	0.832	
rock bass	0.07	0.32	0.45	0.004	increase
redbreast sunfish	0.80	0.84	0.85	0.063	
green sunfish	0.04	0.01	0.03	0.289	
bluegill sunfish	0.37	0.10	0.01	0.865	
smallmouth bass	0.01 A	0.11	0.28 B	<0.001	increase
largemouth bass	0.06	0.04	<0.01	0.962	
fantail darter	42.38	49.90	13.06	0.063	
johnny darter	0.05	0.22	0.20	0.199	
riverweed darter	0.79	0.68	1.34	0.001	increase
Roanoke darter	- A	0.02 A	0.35 B	<0.0001	increase
total density / 100 m	131.2	175.5	82.4		

Table 2.6.-*P*-values for two-tailed t-tests comparing groups where each species was present and absent. Asterisks signify unequal variances, and "+" indicates that values for the group of variables where a species was collected were greater than where it was not collected. N present is the number of sites where the species was collected. Significant at $\alpha = 0.05$ is a count of significant tests.

Variable	rosyside dace	white shiner	bluntnose minnow	Roanoke hogsucker	black jumprock	torrent sucker	margined madtom	mottled sculpin	rock bass	redbreast sunfish	riverweed darter	Significant at $\alpha = 0.05$
N present	32	14	10	16	11	26	22	16	12	17	16	
N absent	11	29	33	27	32	17	21	27	31	26	27	
mean width (m)	<0.0001* +	0.0012 +		0.024 +	0.003 +		0.007* +		0.009 +		0.006* +	7
canopy closure												0
median depth							0.047* +		0.001 +	0.004* +	0.027 +	4
max. obs. depth (cm)	0.0004 +											1
st. dev. depth	0.0003* +		0.011 +				0.049* +		0.003 +	0.005 +		5
prop. riffles								0.014 +		0.002 -		2
prop. pools			0.021 +			0.007 -		0.0008* -		0.002* +		4
discharge (cfs)						0.021* +					0.033* +	2
D50					0.042 +							1
D90		0.026 +			0.038* +							2
st dev. substrate				0.028 +								1
riffle D50	0.0096 +	0.016 +										2
pool D50												0
% of silt & sand												0
% woody vegetation	0.010 +											1
% impervious surfaces							0.005* +					1
% pasture												0
% weed/grass/yard	0.0015 -							0.033 -				2
watershed area (km2)	0.0003* +	0.0086 +		0.0066 +	0.002* +		0.0005* +		0.006* +	0.014* +	0.001* +	8
site elevation (m)			0.0002* -	0.0007 +				0.0004* +		0.015* -		4
watershed forest					0.012* +							1
watershed herb/ag												0
watershed disturbed			0.015 +	0.0012* -	0.0043* -			0.0024* -				4
forest 60 %, 60 m			0.014 -	0.001 +				0.0004 +		0.036 -		4
herb/ag 60 %, 60 m												0
disturbed 60 %, 60 m			0.0035 +	<0.0001* -				<0.0001* -				3
forest 20 %, 60 m			0.034 -	0.001 +				0.0002 +		0.048 -		4
herb/ag 20 %, 60 m												0
disturbed 20 %, 60 m			0.0016 +	0.0003* -			0.036 -	<0.0001* -		0.046 +		5
Significant at $\alpha = 0.05$	8	4	6	7	5	2	5	7	4	8	4	60

Table 2.7.–Historical comparison of Back Creek fishes. Samples at sites 7, 8, and 11-14 were summarized by Hambrick (1973). A = abundant, C = common, U = uncommon, R = rare. Entries for my sites are number of individuals for each species. Lower Back Creek site was located halfway between Sites 7 and 8. Middle Back Creek was at the same location as Site 12; Site 11 is included for comparison. Upper Back Creek was located between Sites 13 and 14, closest to Site 14. Middle Back Creek was sampled twice (VS019 and VS041).

Species	Site 7	Lower Back Creek	Site 8	Site 11	Middle Back Creek	Middle Back Creek	Site 12	Site 13	Upper Back Creek	Site 14
largemouth bass						1				
Roanoke darter		2								
white shiner		1								
silver redhorse	U									
v-lip redhorse	U	5								
smallmouth bass	U	9								
shield darter	U									
swallowtail shiner	C	71	C							
satinfin shiner	U									
pumpkinseed	U		U							
riverweed darter	U	50	U							
central stoneroller	U	248	U	C	12	39				
rosyside dace		1		U		3				
common carp				U						
johnny darter		26	U	C						
margined madtom	U	89	U	C	23	22				
channel catfish				U						
white sucker		79	C	U	51	90				
northern hogsucker	U	7	U	C						
black jumprock	C	82	U	U	8	7				
golden redhorse		12		U						
redbreast sunfish	U	67	U	U						
bluegill sunfish				U						
American eel				U						
rosefin shiner	C	47	A	C						
Roanoke hogsucker	U	45	U	C	18	25	U			
bluehead chub	A	612	A	A	193	175	U	C	94	
crescent shiner	C	739	A	A	127	169	A	U	8	
mountain redbelly dace	U	328	A	A	420	758	A	A	130	
blacknose dace		18		U	53	83	U	U	38	C
fantail darter	C	372	C	A	272	162	U	C	491	C
yellow bullhead		2								
rainbow trout		1								
brown trout		1			4					
creek chub						1				
Total species	19	25	16	20	11	13	6	5	5	2

Table 2.8.–Historical comparison of Mason Creek fishes. Samples at sites 1, 2, and 4-7 were summarized by Jenkins and Freeman (1972). A = abundant, C = common, U = uncommon, R = rare. Entries for my sites are number of individuals for each species. Lower Mason Creek site was located between Sites 1 and 2, closest to Site 1. Middle Mason Creek site was between Sites 4 and 5, closest to Site 5. Upper Mason Creek site was located halfway between Sites 6 and 7.

Species	Site 1	Lower Mason Creek	Site 2	Site 4	Middle Mason Creek	Site 5	Site 6	Upper Mason Creek	Site 7
bigeye jumprock	R								
brown bullhead	R								
satinfin shiner	U								
spottail shiner	C								
Roanoke logperch	R		R						
largemouth bass	R		R		4				
golden redhorse	R		U						
torrent sucker	U		R						
riverweed darter	A	27	C						
black jumprock	R	23							
Roanoke darter	U	3	C						
cutlips minnow	R	26	U						
smallmouth bass	R	10	U						
mimic shiner	U			R					
bluntnose minnow	C	149	C	U	10				
white shiner	C	246	C	C					
bluegill sunfish	U		U	U	3				
margined madtom	U			R	36				
rock bass	U	37		R					
rosefin shiner	A		A	A		R			
johnny darter	C	13	A	C	20	C			
redbreast sunfish	C	64	C	C	18	R			
swallowtail shiner	C		C	C	1	C			
crescent shiner	C	68	C	C	43	A	U	1	
bluehead chub	C	144	C	C	24	R	R	27	
white sucker	C	12	C	U	15	C	U	2	
pumpkinseed	U		C			C	R		
northern hogsucker	R		R				R		
chain pickerel			C		2	C	C	1	
central stoneroller	C	155	C	U	167	U	U	105	U
fantail darter	C	153	C	U	206	A	U	130	C
mountain redbelly dace	R		U		70	A	U	473	A
rosyside dace		1		U	90	C	U	101	A
blacknose dace								67	C
creek chub								12	C
Roanoke hogsucker		1							
Total species	31	17	24	16	15	13	10	10	6

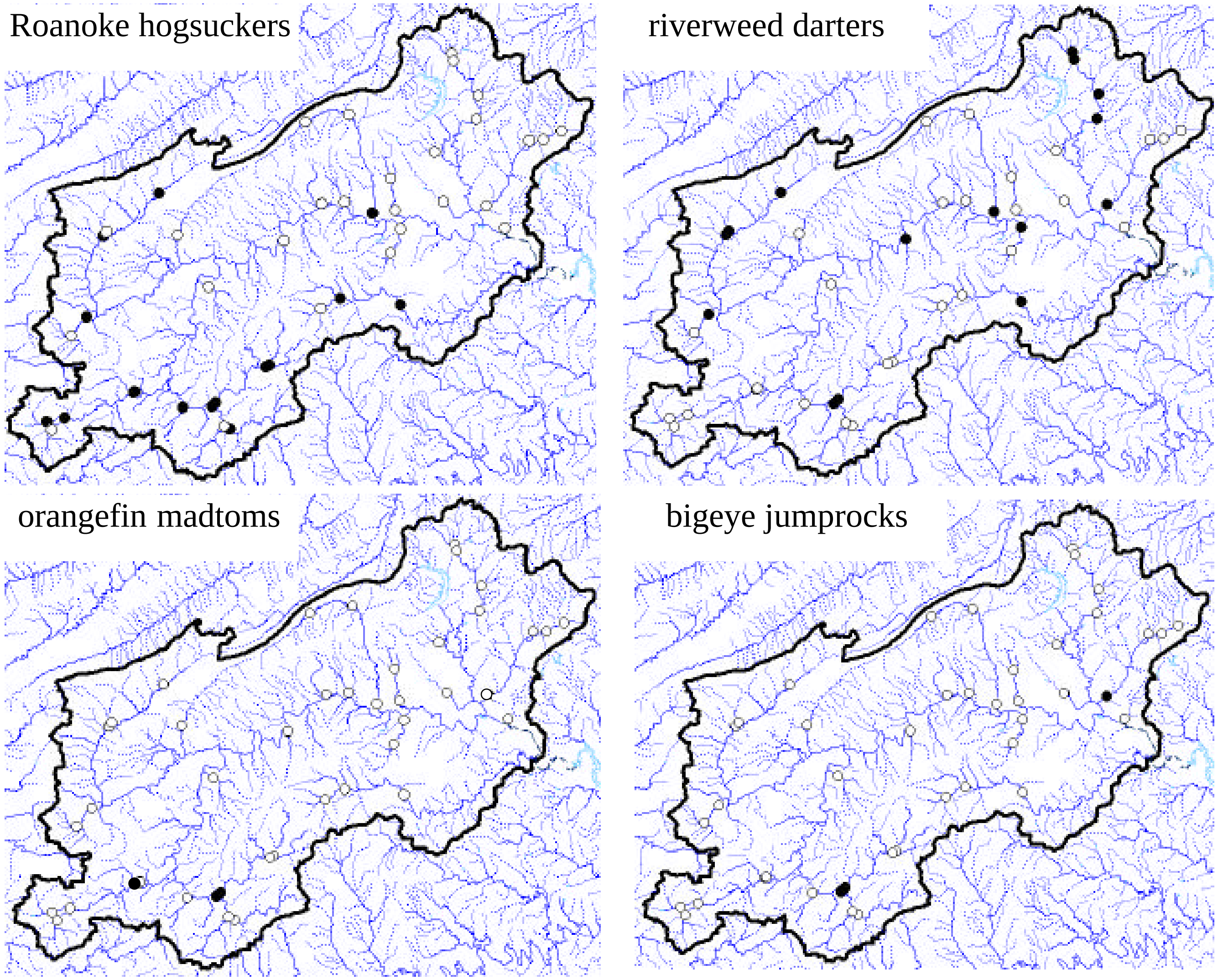
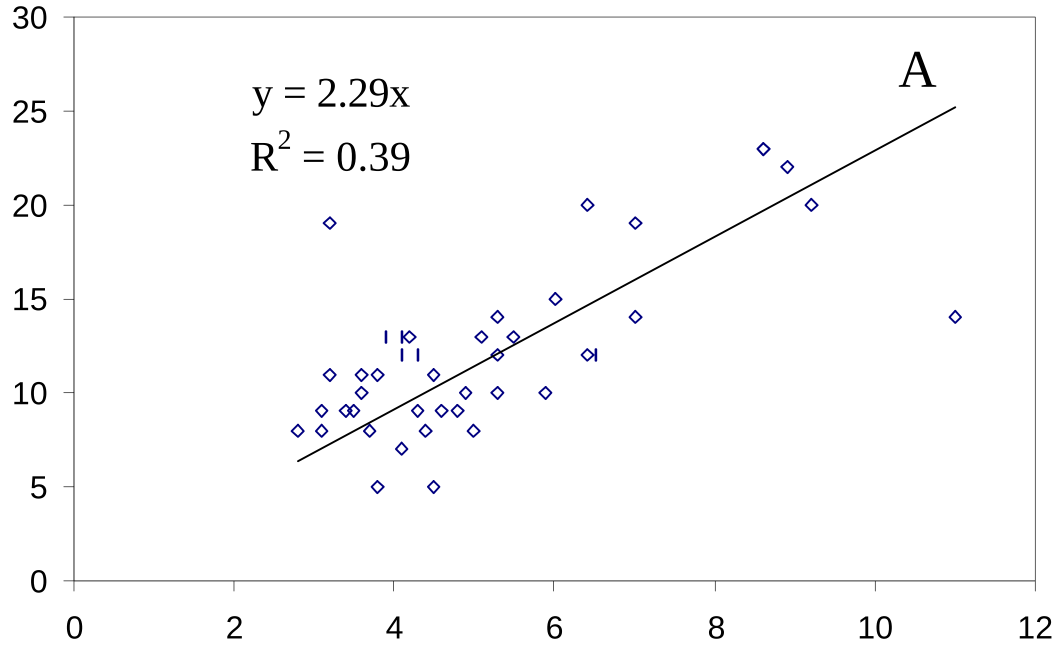


Figure 2.1.-Distribution of Roanoke hogsuckers, riverweed darters, orangefin madtoms, and bigeye jumprocks in the URRW. Sites where these species were collected are indicated by black circles. White circles indicate that the species was not collected at that site.



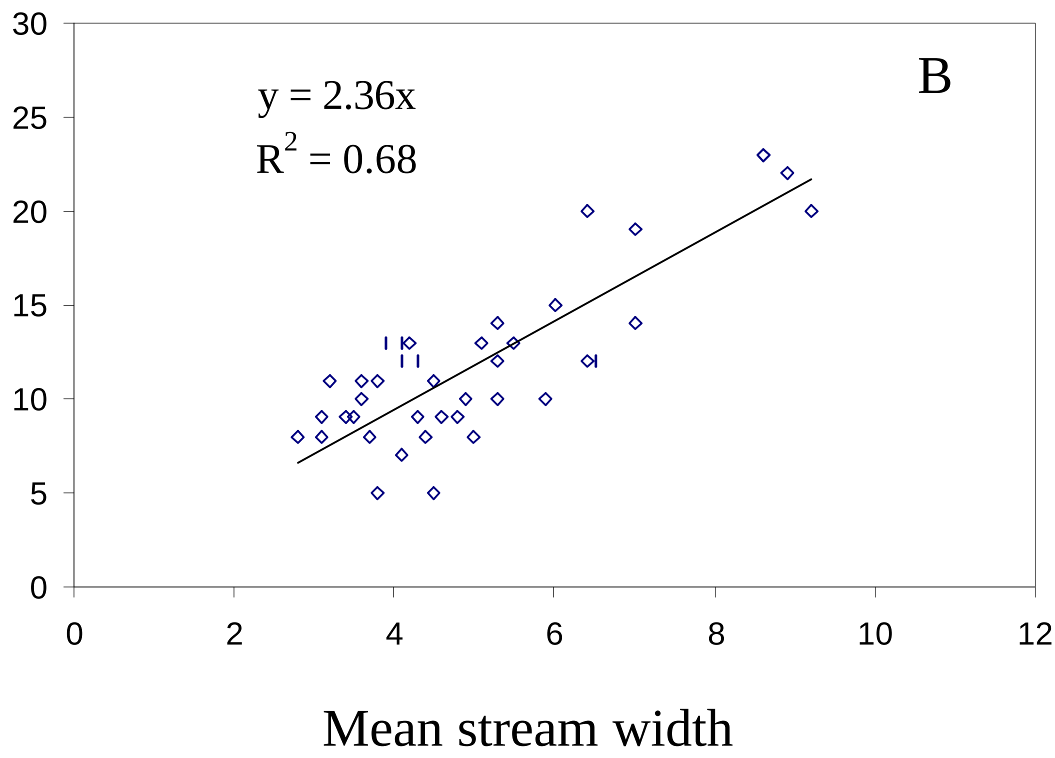


Figure 2.2.-Simple linear regression comparing native species richness and mean stream width. All sites are included in chart A while Middle North Fork and Lower Tinker Creek are excluded from chart B. Regressions for both charts were forced through zero.

CHAPTER 3. Comparing Fish Community Biological Integrity with Watershed and Habitat Conditions Using a Multimetric Index

Fish communities can be used effectively to convey the health or integrity of a stream reach. Multimetric indexes combine many different features of fish communities to produce one number that represents the biological integrity of the entire fish community from both a taxonomic structure and ecological function standpoint. This value can easily be used to compare fish community biological integrity with land use and habitat conditions. It is especially useful for identifying which landscape levels have the most influence on fish communities. This is important to identify the necessary spatial extent of future restoration and protective measures at watershed and riparian levels.

OBJECTIVES

The goal of this chapter was to assess the biological integrity of fish communities in the URRW and determine ways that habitat conditions and land use at different scales affect integrity. The first objective was to identify metrics commonly used in indexes of biotic integrity that would be appropriate to assess fish communities at my sites. This was accomplished by reviewing studies that have used an IBI, especially studies from Virginia. The second objective was to evaluate fish community integrity at sites using a multimetric index. My index, the mean metric score, summarized 13 metrics my taking the average of standardized metric scores determined for each metric at each site. Another facet of this objective was to determine if metric values changed from small to large sites in consistent, significant manners. The third objective was to determine which species were consistently correlated with mean metric scores using density values for different species. The fourth objective was to determine which in-stream and watershed level variables were correlated with mean metric scores and with values for individual metrics. This objective included correlating mean metrics scores with land use at 24 different spatial scales.

METHODS

Fish community organization

I used metrics that are commonly used in indexes of biotic integrity (IBI) to examine fish community structure and function at my sites (Table 3.1). Fishes were classified into different functional guilds based primarily on Smogor (1996) but also on Jenkins and Burkhead (1993) and Smith (1999). Thirteen metrics were selected *a priori* based on studies that analyzed the utility of different metrics for Virginia streams (Smogor 1996, Smogor and Angermeier 1999a, 1999b). I chose five taxonomic, three trophic, two reproductive, two tolerance and one individual health metric. A multimetric index should comprise three to five taxa richness, two or three tolerance, two to four trophic structure, one or two individual health, and two or three other ecological attribute metrics (Karr and Chu 1997). The 13 metrics (their abbreviations in parentheses when introduced), expected response to anthropogenic disturbance, and the number of species included in each metric can be found in Table 3.1 and classification of species for each metric is in Table 3.2. Values for species count metrics were determined by counting the number of species collected at each site, regardless of abundance. Values for proportional abundance metrics ranged from 0 to 1 and were calculated using the proportion of the total number of fish collected that met metric requirements.

Number of native species (natives) and number of introduced or nonnative species (nonnatives) are metrics commonly used in IBIs (Karr and Chu 1997). I determined native or nonnative status for the Roanoke River system from Jenkins and Burkhead (1993) with one exception. Brook trout are considered probably native to the Roanoke River drainage but were considered introduced in my study because captured individuals had an appearance and size typical of stocked fish and were found in areas atypical of wild brook trout.

Three metrics reflected the number of species from four different families: Percidae and Cottidae combined, Catostomidae, and Cyprinidae. All darter and sculpin species (darters or sculpins) and all sucker species (suckers) found in the URRW are native. Two minnows (common carp and bluntnose minnow) found in the URRW are introduced and tolerant of degradation. Native cyprinids will be more affected by anthropogenic degradation than these tolerant, nonnative species (Jenkins and Burkhead 1993). Therefore, only native cyprinids were considered for the minnows metric (minnows).

Centrarchidae was the only family represented by more than three species that was not depicted by a metric. This metric was excluded because five of seven species collected were nonnative. Although most IBIs consider reduced sunfish species richness to be indicative of disturbance, Smogor and Angermeier (1999b) found more sunfish species in urbanized streams than in pristine streams.

I included three functional metrics that described trophic preferences of species. Trophic classification of all species was based on Smogor and Angermeier (1999b) except for orangefin madtom and Roanoke logperch, which were classified based on Jenkins and Burkhead (1993). Smogor (1996) explained how species were classified based on type and diversity of food items, as well as feeding behavior. The proportion of generalist feeders (generalists) represented the proportion of the total number of fish caught that consumed more than two different food types. The proportion of non-benthic invertivores (invertivores) included species that consumed invertebrates but not species that were primarily benthic feeders. Species were classified as both generalist feeders and invertivores if they consumed several types of food, including invertebrates. The proportion of benthic specialist invertivores (benthic invertivores) represented species that fed benthicly, yet only consumed two or fewer food types (Smogor 1996).

Many IBIs use a measure of the proportion of piscivores or specialized carnivores, which is expected to decrease with disturbance. This metric was not included because only one native species currently found in the URRW, chain pickerel, is a specialist carnivore and it is only found in a small portion of the URRW. Two other native specialist carnivores, Roanoke bass and American eel, have been extirpated from the URRW. All other specialist carnivores are introduced species and high proportional abundance of them would be counterintuitive to accepted theories of biological integrity.

I selected two functional metrics related to reproduction: proportion of simple lithophilic spawners (simple lithophils) and number of late maturing species (late maturers). Species classified as simple lithophils use mineral substrate for spawning but do not manipulate their spawning habitat. Species classified as tolerant were excluded from the simple lithophils metric (Smogor and Angermeier 1999b). The number of late maturing species is composed of species in which females mature after two years of age.

I selected two tolerance metrics: proportional abundance of tolerant species (tolerants) and number of intolerant species (intolerants). Classification of species as tolerant or intolerant was taken directly from Smogor (1996). Tolerant species are affected the least by anthropogenic disturbance. Range or abundance of intolerant species has declined, presumably due to anthropogenic disturbance (Smogor 1996). Both metrics are commonly used in IBIs (Karr and Chu 1997), but in my study, the intolerants metric was only used to analyze large sites because torrent sucker was the only intolerant species collected at small and medium sites.

External anomalies such as deformities, lesions, and tumors have been shown to increase with anthropogenic disturbance (Sanders et al. 1999). In addition, external parasites such as black spot, leaches, and anchor worm are often considered anomalies in IBI applications (Rankin 1989). Black spot, which varies greatly in intensity, was usually found on a large proportion of individuals relative to other anomalies, and was often found on individuals unaffected by other anomalies. Therefore, only heavy infestations of black spot, which covered the entire body at a density equal to or greater than eye diameter, were considered anomalous (Rankin 1989). The proportion of individual fish with external anomalies (anomalies) was determined by dividing the number of measured fish that had anomalies by the total number of measured fish. A fish could have more than one anomaly, but it was only counted once because the metric looked at the proportion of fish with anomalies, not proportion of anomalies.

Although metrics were selected *a priori*, several adjustments became necessary after collections indicated potential conflicts. Functional metrics ineffectively characterize fish communities if included species differ in some ecological attributes that result in drastically different responses to degradation. After assessing initial relationships of benthic invertivores and simple lithophils with overall metric trends, I reexamined the rationale for including certain species in these metrics. I removed several species to better reflect the purposes of these metrics.

Values for the benthic invertivores metric are expected to decline with degradation because such specialized feeders would be disadvantaged if benthic invertebrate populations decline, as would be expected with excessive siltation or other forms of degradation (Berkman and Rabeni 1987, Rabeni and Smale 1995). However, in addition to being benthic invertivores, fantail darters were classified as tolerant species, were collected in every sample, and exceeded all other species in abundance. Despite their specialized feeding habits (Matthews et al. 1982), fantail darters are not affected as much by degradation as other members of this metric. Therefore, the benthic invertivores metric would not reflect fish community integrity in the designed manner unless fantail darters and other tolerant species were excluded. Fantail darters and johnny darters were removed from the benthic invertivores metric because they are tolerant species and therefore do not respond to degradation the same as other species in the metric.

I removed rosyside dace, mountain redbelly dace, crescent shiner, white shiner, and rosefin shiner from the simple lithophils metric because these species are considered nest associate spawners and benefit from substrate manipulations by bluehead chubs (Smith 1999). Even though they do not manipulate substrate themselves, they benefit from mounds of clean gravel provided by chubs, which can reduce the effects of sedimentation on spawning success. Bluehead chubs were collected in every sample and therefore nest associate spawners had opportunities to spawn on chub mounds and have their eggs protected at every site. Including nest associate spawners would inflate metric values and fail to depict variation in less abundant species that this metric was designed to represent. Fantail darters, rosyside dace, mountain redbelly dace, and crescent shiners were present at 32 or more sites and constituted 60.7 % of the fish collected, thereby masking any trends related to less abundant benthic invertivores or simple lithophils (Figure 3.1).

Because there is no established metric scoring system for Virginia streams that produces an IBI score, I had to find another way to construct a multimetric index that would combine metrics into one meaningful number. First, I determined the expected change in metric values with increased degradation (metric value increases or decreases). I then separated sites by watershed size class and determined a metric score using Equation 1 if the metric value decreased with degradation or Equation 2 if the metric value increased:

$$\frac{MV_u - MV_{min.}}{MV_{max.} - MV_{min.}} \qquad \text{(Equation 1)} \qquad \frac{MV_{max.} - MV_u}{MV_{max.} - MV_{min.}}, \qquad \text{(Equation 2)}$$

where MV represents a metric value and MV_u is a metric with an unknown value. These equations produced scores for each metric ranging from 0 (worst) to 1 (best). For each watershed size class, one site received a score of 1 and one site received a 0 for each metric. If two or more sites had the same metric value they would also have the same metric score. This was common among species richness metrics and could result in two or more sites with scores of 0 or 1 for an individual metric. After deriving a score for all metrics for each site, I averaged metric scores by site to give a mean metric score that represented overall fish community integrity at each site. Sites from different watershed size classes were then combined into one data set for analysis.

My method for calculating a mean metric score for each site is similar to a typical IBI in that they both combine many metrics to give an overall fish community assessment. The main difference is that my approach is a relative one because it does not attempt to identify which sites in the data set are excellent or poor, only which sites are better or worse than others. By looking at sites within a range of watershed, riparian, and in-stream habitat conditions, this approach used sites within the study area to determine what fish community conditions were possible, rather than relying on established values based on streams elsewhere, which may or may not accurately reflect conditions for small streams in the URRW.

To rate the integrity of fish communities at sites, I divided mean metric scores into three groups: best, intermediate, and worst. I determined boundaries for each group by dividing the difference of the highest and lowest mean metric scores by three, which established three groups of scores with equal ranges of possible scores. I classified mean metric scores from 0.231 to 0.400 as worst, scores from 0.400 to 0.569 as intermediate, and scores from 0.569 to 0.738 as best.

Statistical Analysis

To detect significant differences in metric values among sites grouped by watershed area, I used the Kruskal-Wallis and Tukey studentized range tests. The null hypothesis was that individual metric values were equal for each watershed size class while the alternative hypothesis was metric values differed among size classes. This comparison assessed the importance of separating sites by watershed area.

I assessed variation in values of 12 metrics from 43 sites (intolerants metric was not included) among sites using principal component analysis. This analysis indicated which metrics did the best job of separating sites and determined which metrics responded in similar manners. I also compared individual metric values for each site with mean metric scores using Spearman's rank order correlation coefficient (r_s) to further determine if individual metrics agreed with fish community assessments determined by mean metric scores. I removed the metric being evaluated from the mean metric score calculation before comparing it with the mean metric score. I compared individual species density values from each site with mean metric scores to determine how different species corresponded with overall evaluations of community integrity.

I compared habitat and watershed variables with mean metric scores and individual metric values using Spearman's rank order correlation coefficient. Comparisons using mean metric scores gave an overall evaluation of effects on fish communities while comparisons with individual metrics identified factors that affect certain groups of fishes. The latter comparisons helped determine if land use and stream conditions affected various fish community groups in ways found by previous studies and also assessed metric efficacy for future applications.

I compared forest and disturbed land use at 24 spatial scales with mean metric scores using Spearman's rank order correlation coefficient to determine how land use at different spatial scales affects fish communities. For this analysis, I used various combinations of buffer width and proportion of stream channel network discussed in Chapter 1.

RESULTS

Metric summary and agreement

Mean metric scores ranged from 0.231 for Lick Run to 0.738 for Lower Goose Creek (Table 3.3). Mean metric scores for 18 sites were below 0.400 and were classified as worst sites. Seventeen sites had scores between 0.400 and 0.569 and were classified as intermediate sites. Only 8 sites had scores high enough to be rated best. With the exception of Lower Glade Creek, all best sites were tributaries to the North or South Fork of the Roanoke River (Figure 3.2). Most worst sites drained developed areas near the cities of Roanoke or Salem, but Upper Back Creek, Bradshaw Creek, Middle Tinker Creek and Middle Glade Creek were notable exceptions.

Mean values for all species count metrics (natives, nonnatives, darters or sculpins, suckers, minnows, late maturers) increased significantly with stream size (Table 3.4). Values for four other metrics changed significantly with stream size; invertivores and simple lithophils increased while tolerants decreased. Only generalists, benthic invertivores, and anomalies metrics exhibited no significant change at $\alpha = 0.05$. Tolerants declined from an average of 46 % of the species at small sites to only 25 % at large sites. Simple lithophils increased from small to large sites while specialized feeders showed mixed results from small to large sites. Although invertivores increased significantly from small to large sites, generalist feeders and benthic invertivores showed no significant trends. Mottled sculpins accounted for a large proportion of benthic invertivores and as a result their abundance largely dictated values for the benthic invertivores metric. With mottled sculpins excluded, benthic invertivores represented 1.10 % of the fishes at small sites, 0.80 % at medium sites, and 2.61 % at large sites.

For principal component analysis, 0.451 of the proportion variation among metrics was explained by principal component 1 (eigenvalue 5.413, Table 3.5). Correlations for all species count metrics were $\geq$ 0.30 for principal component one. All six species count metrics contributed to variation along principal component one and were consistent with mean metric score trends. For principal component two, correlations for invertivores and anomalies metrics were > 0.30 while correlations for the benthic invertivores metric was < 0.30 (eigenvalue 1.679, proportion variation 0.140). All family richness metrics were highly correlated with natives (> 0.80) and the nonnatives metric was also positively correlated with native species richness (0.63). Generally, sites with many members of one family also contained many members of other families, as well as many nonnative species.

Values for eight metrics were significantly correlated with mean metric scores using Spearman's rank order correlation coefficients. Natives, darters and sculpins, suckers, minnows, late maturers, benthic invertivores, and simple lithophils were significantly, positively, correlated with mean metric scores (Table 3.6). These results corroborated findings using principal component analysis because values for species count metrics responded similarly at different sites. Benthic invertivores and simple lithophils metrics are generally unrelated to species richness, but both metrics were significantly correlated with mean metric scores. Values for the tolerants metric were significantly, inversely related to mean metric scores. Invertivores was the only metric expected to increase with increased integrity that was not significantly, positively correlated. Nonnatives, generalists, and anomalies, metrics expected to increase with degradation, were not significantly, inversely correlated with mean metric scores. Nonnatives were generally more common at sites with numerous native species and therefore did not show an increase with degradation as expected.

Comparisons of metrics with habitat and watershed variables

Two of 13 habitat variables and eight of 16 watershed or land use variables were significantly correlated with mean metric scores (Table 3.7). In summary, proportions of pools and riffles, site elevation, weed/grass/yard and pasture at sites, watershed disturbed, and forest and disturbed buffers (at 20 %, 60 m and 60 %, 60 m scales) were significantly correlated with mean metrics scores. No variables related to substrate size or depth were significantly correlated with mean metric scores. Correlations with herb/ag land showed mixed results. Mean metric scores were significantly, positively correlated with percent pasture at sites, but correlations with herb/ag land at other scales were both positive and negative, but never significant. Site elevation and disturbed land use appeared to have the most effect on mean metric scores. Mean metric scores increased with elevation but declined as disturbed land use at buffer and watershed levels increased. Mean metric scores were higher where riffles were common but declined at sites where pools were abundant.

All individual metrics were significantly correlated with three or more habitat, land use, or watershed variables (Table 3.8). At least one metric was significantly correlated with 10 of 13 habitat variables and with 10 of 12 watershed and land use variables. More metrics were significantly correlated with watershed area and mean width than other variables, 10 and eight correlations, respectively. No metrics were significantly correlated with three substrate variables, D90, riffle D50, and percent of silt and sand, or with two site land use variables, woody vegetation and pasture. Minnows, simple lithophils, and anomalies metrics were each significantly correlated with four habitat variables. All species richness metrics were positively correlated with mean width and watershed area, which reflect stream size. Anomalies were significantly correlated with nine variables and simple lithophils and tolerants were each correlated with eight variables. Most significant correlations for watershed and buffer land use (60 %, 60 m) were with four functional metrics: tolerants, anomalies, benthic invertivores, and simple lithophils. Number of suckers species was the only family richness metric significantly correlated with any land use variables.

Plots comparing benthic invertivores, simple lithophils, tolerants, and anomalies with disturbed land use in the 60 %, 60 m buffer show how metric values varied with land use (Figure 3.3). All sites with high benthic invertivore proportions were less than 10 % disturbed. With three exceptions, this trend was also true for simple lithophil proportions. Tolerants values varied the most and some sites with very little disturbed land use had high proportions of tolerants. However, tolerants comprised over 28 % of the fish community at all sites that were more than 20 % disturbed. Many sites had very few, if any, fishes with anomalies, but anomalies were higher for sites with disturbed land use.

Benthic invertivores were absent at 17 sites and simple lithophils were not collected at 10 sites. Also, benthic invertivores and simple lithophils constituted less than 1 % of the community at other sites (four and seven, respectively). However, all large sites contained members of each metric. Due to the large number of sites with no members of either metric, there were no clear relationships with forested land use. Abundance of mottled sculpins varied widely among sites and had a large influence on benthic invertivore proportions.

Benthic invertivores and simple lithophils metrics only included species that would be easily impacted by degradation because of their feeding and reproductive strategies. I examined the distribution of species that were included in both metrics: northern hogsucker, bigeye jumprock, Roanoke logperch, and Roanoke darter. Northern hogsuckers were collected in 11 different samples; six of these were at large sites. Except for one Roanoke darter at Middle North Fork, all individuals of the other three species were collected at large sites. In all, 12 sites contained one or more of these species, but only one individual northern hogsucker represented this group at small sites. Values for the proportion of the four species at 12 sites ranged from 0.0006 for Lower Elliott Creek to 0.0226 for Lower Bottom Creek. Lower Glade Creek contained the most members of this group (29), followed closely by Lower Goose Creek (27). Only Lower Glade Creek contained all four species because it was the only site where a Roanoke logperch was collected. The other three species were collected at Lower Goose Creek and were collected in Bottom Creek when the middle and lower sites are considered collectively.

Density values for 16 different species were significantly correlated with mean metric scores (Table 3.9). All of these species exhibited positive trends with mean metric scores except bluntnose minnow and white sucker. Densities for all sucker species, except white sucker, were significantly, positively correlated with mean metric scores ($r_s > 0.39$, $P < 0.008$ for all). All species in this analysis that were classified as intolerant (bigeye jumprock, torrent sucker, and orangefin madtom) were significantly, positively correlated with mean metric scores. All species classified as tolerant, except creek chubs and green sunfish, were negatively correlated with mean metric scores. However, bluntnose minnows and white suckers were the only tolerant species significantly correlated with mean metric scores.

Comparisons of mean metric scores with buffer land use at 24 spatial scales revealed significant relationships for forest and disturbed land use but not for herb/ag land use (Figure 3.4). Relationships between mean metric scores and forested land at various spatial scales were generally linear. Mean metrics scores were low for sites with low percentages of forest and increased as forested land increased. The highest r_s between forest land use and mean metric scores occurred for the 5 %, 150 m buffer area ($r_s = 0.612$, $P < 0.0001$). As more of the stream channel network was included, (i.e., 20 %, 60 %, entire network), r_s values declined, especially from the 60 % network area to the entire stream network area. Also, as buffer width increased, r_s values increased consistently across various proportions of stream channel network.

For disturbed land, the highest r_s value with mean metric scores was for the 10 %, 150 m buffer ($r_s = 0.637$, $P < 0.0001$, Figure 3.4). However, r_s values were high for all buffers regardless of stream network area or buffer width. Entire watershed land use was not as strongly correlated with mean metric scores as buffer land use, but the relationship was still significant ($r_s = -0.422$).

Graphs of disturbed land in entire watersheds and mean metric scores indicated that fish communities can suffer at relatively low levels of disturbance (Figure 3.6). Most of the sites with highest mean metric scores (i.e., best sites) had less than 2 % disturbed land within their watersheds. Mean metric scores declined precipitously from 2 % or less disturbed land and level out between 10 and 20 % disturbance. Seven sites were over 21 % disturbed in the entire watershed and only two of them had mean metric scores high enough to be classified as intermediate or best. Biological integrity was generally low beyond 20 % disturbance regardless of the amount of disturbed land. Thirteen sites had low amounts of disturbed land and low mean metric scores. When land use only within the 20 %, 150 m buffer was considered, eight of these 13 sites had disturbance levels above 20 %. For the 20 %, 150 m buffer, 15 sites were over 20 % disturbed and only two were not classified as worst sites. As with entire watershed land use, most sites with high integrity had buffer areas with little or no disturbance and mean metrics scores declined precipitously as disturbance increased to 10 %.

DISCUSSION

Metric summary and agreement

Most sites draining urban areas around Roanoke and Salem had low mean metric scores and were classified as worst sites. However, Lower Glade Creek was classified as a best site and had more species than all other sites. It had more introduced species than any other site (eight) and tied with Lower Goose Creek for most native species (23). The headwaters of Glade Creek drain a mosaic of land uses with land closest to stream channels usually in agriculture or development. High species richness at Lower Glade Creek is surprising considering only eight native species were found at the upper and middle sites. This suburban stream between the city of Roanoke and the town of Vinton was an unexpected place to find a fish community with high integrity that included bigeye jumprocks and a Roanoke logperch. Lower Glade Creek was sampled during high discharge, which may have enhanced species richness by fish moving upstream from Tinker Creek. However, my field crew and I collected many species responsible for the high species richness in Glade Creek by seining on 16 August 1999. We collected one juvenile bigeye jumprock by seining. Discharge was at base flow on this day but appeared higher than other streams of similar drainage area during that time of year. Consistent flows may account for some of the diversity at this site. Lower Glade Creek is proof that even sensitive species can survive in urbanized streams and such streams can have high integrity.

Upper Back Creek had the third lowest mean metric score despite draining an area that is over 90 % forested. The low mean metric score was primarily due to low species diversity. Only five species were collected, the fewest of any site, and Upper Back Creek was the only site void of suckers. Conversely, three sucker species were collected downstream at Middle Back Creek. The downstream portion of my site was approximately 10 m upstream from the confluence of Martins Creek with Back Creek. Species may remain downstream of this confluence rather than moving upstream where water volume is less. Also, at the time of sampling, a manmade dam of stream substrate had been constructed between the lower end of my site and the confluence with Martins Creek. This structure, built to provide “swimming” opportunities for young children, likely diminished upstream fish movement, except during high flows. Additionally, a swimming pool intake pipe is located just upstream of my site so water exchange with the stream is a possible culprit for the low fish diversity.

Bradshaw Creek and Mason Creek are both low gradient streams draining karst topography. Both streams have a portion of their discharge flowing underground, but unlike some other URRW tributaries (i.e., Dry Branch, Brake Branch, and Dry Run), they do not dry completely near their mouths. Both streams contained chain pickerel, a native species absent in other URRW tributaries. These creeks are also anomalous because they have wide pools with depths approaching one meter separated by short narrow riffles. Total species abundance was very low in Bradshaw Creek, probably due to five predators, three of which are nonnative, that composed nearly 14 % of the fish community. Due to the unusual hydrology and abundance of predators, land use practices have less effect on the fish community at Bradshaw Creek than at most sites. Fish communities in Upper Back Creek and Bradshaw Creek indicate that land use practices are not always the most important factors influencing fish communities.

Middle Tinker Creek and Middle Glade Creek also had low mean metric scores. Herb/ag land covers over 65 % of the Middle Tinker Creek watershed and is mainly responsible for site conditions. Siltation and turbidity are worse at this site than any other; 50 % of the streambed is either silt or sand. Anomalies were more common than at all other sites with 7.4 % of the fish affected. Middle Glade Creek drains a large amount of herb/ag land and the site is located within a cattle pasture. Only eight native species were collected at Middle Glade Creek. Conditions in Middle Tinker Creek and Middle Glade Creek indicate that development is not the sole cause of impaired stream ecosystems in the URRW - agriculture can impact streams as well.

As expected, values for all species richness metrics increased from small to large sites (Table 3.4). Numerous studies have recorded longitudinal species additions with stream size (e.g., Sheldon 1968, Whiteside and McNatt 1972) and IBI studies have taken into account increases in species richness (e.g., Fausch et al. 1984, Steedman 1988, Crumby et al. 1990). My data indicated that this trend is important to consider when dealing with relatively small streams less than 10 m wide. This trend was even evident when comparing small and medium sites because mean native species richness differed by 2.6 species. I recommend separating sites by watershed area before comparing fish community attributes. Otherwise, large depauperate streams will have a fish community, at least in terms of species richness, more similar to smaller streams. For example, if a large stream in the URRW only contains 14 native species, it is missing several species that would be there under normal conditions and the stream lacks integrity. A medium size stream may only contain 13 native species, which is a reasonably high number for comparably sized streams, but is less than the large site. If these sites are compared with the same criteria, the large site would be considered better in terms of native species richness. However, considering its size, more native species should be present and therefore it should be considered inferior to the medium site.

Most functional metrics also showed consistent trends from small to large sites. Generally, fish communities at small sites had more trophic and reproductive generalists while more specialized fishes were limited to larger sites. Most studies fail to account for longitudinal functional changes in fish communities even though these trends are accepted tenets of aquatic ecology (Vannote et al. 1980, Smogor and Angermeier 1999a). As with species richness, failing to account for such fish community changes can cause sites with poor integrity to be considered better than they actually are. For example, if 40 % of the species at a large site were classified as tolerant, that proportion would be considered high and would reflect a lack of integrity because tolerants averaged only 25 % at large sites (Table 3.4). However, compared to small sites, 40 % is not high because the average values for small sites was 46 %. Once again, biological integrity will be poorly evaluated if differences in stream size are not taken into consideration (Karr and Chu 1997).

Family richness metrics had a large influence on mean metric scores because they measured attributes that are usually similar for sites. Principal component analysis revealed that all species count metrics are correlated with one another. Some redundancy exists between metrics measuring family level species richness and number of native species due to the additive nature of their relationship. Nonnative species increased when other richness metrics increased. The presence of several nonnative species may indicate that the community lacks integrity, but it does not necessarily indicate that stream conditions are poor, especially if those introduced species are somewhat intolerant of degradation. Naturally reproducing populations of rainbow and brown trout may indicate a lack of integrity but simultaneously indicate suitable conditions for reproduction of two species easily affected by degradation (Wohl and Carline 1996).

Comparisons of individual metrics with mean metric scores using Spearman’s rank order correlation coefficients corroborated results from principal component analysis. Especially noteworthy is the significant negative trend of tolerants because this metric measures an aspect of the fish community unrelated to species richness and declined while most other metrics increased. Benthic invertivores and simple lithophils metrics were also significantly correlated with mean metric scores ($r_s > 0.52$ for each), yet they were not simply a function of species richness. The relationship of these three metrics with mean metric scores and with habitat and land use variables (Table 3.8) indicates that they are useful for IBIs.

Correlations among natives, suckers, and late maturers were very high, suggesting some redundancy by using all three metrics. Seven of the 12 species classified as late maturers were suckers and three of the other late maturing species were only collected once. Therefore, these two metrics are not very different, at least for URRW fish communities. Although both seem to represent important aspects of fish communities, future inclusion of both in IBI applications is questionable if suckers are the primary late maturing species. All functional metrics, except intolerant species and invertivores, effectively characterized various aspects of fish communities and were correlated with several habitat and watershed conditions unrelated to stream size. Invertivores are not as sensitive as benthic invertivores to degradation (Berkman and Rabeni 1987). Although including metrics that respond to varying degrees of impairment is important (Karr et al. 1986, Karr and Chu 1997), the invertivores metric was not very sensitive to impacts because included species varied in other ecological aspects.

Comparisons of metrics with habitat and watershed variables

Comparisons of mean metric scores with habitat and watershed level variables indicated that several factors, resulting from both natural variation and anthropogenic impacts, have large effects on fish community integrity. I separated sites by watershed size class before calculating mean metric scores to reduce the influence of stream size. This approach successfully reduced stream size effects because watershed area and mean stream width were not significantly correlated with mean metric scores.

Elevation and mean metric scores were highly correlated, indicating that higher elevation streams have greater integrity. This trend could be due to natural changes in fish communities from colder, higher gradient streams to warmer, lower gradient streams. However, elevation was strongly correlated with disturbed land use ($r_s = -0.666$) and lower elevation land is preferred for development and agriculture. Therefore, separating the effects of natural changes due to elevation and the effects of land use changes was difficult.

Mean metric scores increased as percent forest increased in buffers and entire watersheds. Based on the precipitous decline in mean metric scores as disturbed land use increased, small amounts of disturbance can cause detectable impacts on fish communities. Disturbed land use within all buffers of varying widths and stream network areas exerted a strong influence on fish communities. Herb/ag land use did not have a consistent effect on fish communities (Wang et al. 1997). Sites in agricultural areas often had high species richness and biomass and were more often similar to well-forested sites than developed sites (Lenat and Crawford 1994, Harding et al. 1998). Although some sites, such as Middle Tinker Creek and Middle Glade Creek, had large amounts of agricultural land use and low mean metrics scores, others, namely Mill Creek and Middle North Fork, had high scores despite traversing cattle pastures.

My findings suggest that considering land use close to stream channels is important when trying to link watershed land use patterns with fish communities. This is especially true for mountainous areas such as the URRW where headwaters are often forested while lower elevation areas are developed. By considering land use in entire watersheds only, the importance of the spatial distribution of various land uses is lost because land conditions far from streams are weighted equally with those close to streams (Steedman 1988, Wang et al 1997). Studies that only look at land use within entire watersheds may fail to find the strongest connections of land use with stream and fish community conditions. The importance of land use close to stream channels can be appreciated by merely examining the wealth of literature devoted to riparian ecosystems and the importance of their integrity to aquatic communities (e.g., Gregory et al. 1991, Naiman et al. 1993, Rabeni and Smale 1995).

Comparisons of forested land use at various spatial scales with mean metric scores indicated that the influence of forests increased when wider buffers were considered but declined when larger stream network areas were considered. Correlations of disturbed land use with mean metric scores were more consistent regardless of buffer width and stream area network. These results indicate that forested conditions close to sites have large influences on fish communities, but forested conditions far away have less influence. In other words, fish communities can be impacted in streams with well-forested headwaters if conditions downstream are degraded. Disturbed conditions seem to influence fish communities regardless of their position in watersheds. Therefore, fish communities are likely to be impacted by development if it occurs in headwaters or in floodplains closer to stream channels.

Areas far from streams could possibly be developed with minimal effects on fish communities if riparian areas along stream channel networks are preserved (May et al. 1997).

All of my sites in disturbed watersheds had disturbed buffer areas, thereby preventing the opportunity to see if forested riparian areas can maintain fish community integrity despite dense development farther from stream channels (Steedman 1988). My findings support proposed greenways which protect riparian areas along stream corridors. They also support most stream restoration activities, which usually occur along riparian areas despite recommendations for a watershed level approach to stream restoration (e.g., Allan et al. 1997, Williams et al. 1997). While land use in buffers had effects on fish communities, land use within five meters along sites had no apparent effect. Therefore, merely planting trees in narrow areas along stream channels is unlikely to have a measurable effect unless the area treated is measured in kilometers rather than meters. Such small-scale activities may have effects but they may not improve stream conditions enough at sites to detect fish community improvements. Such efforts will be especially ineffective if upstream perturbations continue.

Unlike most studies on land use effects, I did not find strong linkages between agricultural land use and fish community integrity (e.g., Berkman and Rabeni 1987, Rabeni and Smale 1995, Allan et al. 1997, Walser and Bart 1999). Seven sites were surrounded by cattle pastures: Mill Creek, Middle North Fork, Lower North Fork, Lower Wilson Creek, Upper Elliott Creek, Upper Tinker Creek, and Middle Glade Creek. Only Middle Glade Creek was considered a worst site while Mill Creek and Middle North Fork were classified as best sites. Discharge at Mill Creek is enhanced by several springs that provide consistent flows and temperatures year-round. Additionally, most of the watershed, including land just upstream from my site, is forested. Abundant springs and forested conditions upstream are able to offset problems due to onsite cattle grazing.

The high rating for Middle North Fork is primarily due to high species richness. Nineteen native species were collected at Middle North Fork, which is four more than were collected at all other medium sites and also more than were collected at two large sites. As a result, Middle North Fork had more darters or sculpins, suckers, and native minnows than all other medium sites and had the highest mean metric score for medium sites. In conclusion, some sites with nearby and watershed level agricultural activities were impacted while others were not. Natural conditions at these sites, such as springs, high elevation, high gradient, and large substrate size may be able to offset agricultural impacts in some cases (Wang et al. 1997).

Relationships between habitat variables and mean metric scores were generally weak, indicating that the habitat variables I measured were poor indicators of fish community integrity. Only proportion of pools and proportion of riffles were significantly correlated with mean metric scores. Many catostomids and other species classified as benthic invertivores or simple lithophils prefer sites with high proportions of riffles and low proportions of pools. Riffles are important spawning areas for simple lithophils and are important foraging areas for benthic invertivores (Berkman and Rabeni 1987, Smith 1999). Amounts of pool and riffle habitat undoubtedly have effects on fish community composition due to habitat preferences of species. However, proportion of riffles increased with elevation while pools decreased. Correlations of elevation with pools, riffles, and land use conditions hinder linking habitat type with fish community integrity.

Watershed land use affects in-stream conditions and these conditions in turn affect fish communities (e.g., Beschta and Platts 1986, Harding et al. 1998). However, fish community variables were not as strongly correlated with habitat variables as with land use variables. Water quality and changes to hydrologic regimes are two obvious avenues for impairment that I did not examine (May et al. 1997). Physicochemical conditions also have a strong influence on fish community composition, especially at small headwater sites (Matthews and Styron 1982, Schlosser 1990). Land use effects on in-stream conditions that affect fish are complex and although a thorough understanding of the pathways is lacking, land use is clearly a driving factor.

Comparisons of individual metrics with habitat and watershed variables identified numerous influences on fish communities, but variables relating to stream size had the most influence (Table 3.8). This analysis indicated that species richness was dictated by stream size measures, but variables relating to anthropogenic impacts had little effect on species richness. Species richness metrics increased with variables such as stream width, depth, discharge, and watershed area. Substrate size and land use did not appear to be important factors influencing species richness metrics. Therefore, a fish community assessment of my data using only measures of species richness and diversity would fail to identify land use impacts.

Metrics that assessed fish community composition from a functional ecology perspective were more effective at revealing land use effects and generally responded to impacts in a manner consistent with expectations. Generalists and tolerants were more proportionally abundant at sites with smaller substrate. Proportional abundance of simple lithophils was greater at sites with more riffles, fewer pools, and greater variation in substrate size. Generalists and tolerants were more abundant at disturbed sites and less abundant at well-forested sites.

Comparisons of individual species with mean metric scores showed that density of 16 species agreed with entire fish community assessments using mean metric scores. This comparison is potentially useful for identifying indicator species that reflect fish community integrity or species that are particularly sensitive to degradation. Density trends for all sucker species included in analysis, except white sucker, agreed with mean metric scores. This analysis is somewhat circular because species that are included in more metrics are more likely to agree with mean metric scores. For example, all sucker species were included in five or six metrics. However, these results tend to validate inclusion of these species in so many metrics.

Comparisons of mean metric scores with forest and disturbed land use at several partial watershed scales indicated that wide buffers are important for maintaining fish community integrity. Correlations were also highest when an area representing only 5 % of the watershed was considered. Also, correlations drastically declined when entire stream network areas were considered. These results indicate that conditions along stream channels far from sites have less impact than more localized conditions. The caveat with the latter finding is the reference to a particular section of stream. Every site within a stream network has an area of local impacts and protecting all stream reaches would require protection of a buffer around the entire watershed channel network, unless some stream reaches are considered more important for protection than others.

Forests constituted a higher proportion of land use within narrow buffers closest to stream channels than in wider buffers. Disturbed land use exhibited the opposite trend and was more abundant in wider buffers. Regardless of which buffer width has the most influence on fish community integrity, stream restoration and protective measures should concentrate on stream buffers rather than entire watersheds. Wider protective corridors will be more effective at protecting fish communities than narrow buffers (Davies and Nelson 1994) and longer lengths of streams will benefit if greater amounts of stream channel networks are protected (Rabeni and Smale 1995).

The best sites, in terms of fish community integrity, were found in the North and South Forks of the URRW, with the exception of Lower Glade Creek, which had a high mean metric score despite being located in a Roanoke suburb. Goose Creek and Bottom Creek have well-forested watersheds and form the South Fork of the Roanoke River at their confluence. In addition, the highest peaks and highest elevation streams in the URRW are found in the Bottom Creek watershed. The Nature Conservancy owns land in both watersheds protecting the Bottom Creek Gorge and land near twin waterfalls on Lick Fork, the main tributary of Goose Creek. Orangefin madtoms and bigeye jumprocks live in both of these streams. With the combination of forested watersheds, aesthetically pleasing high gradient streams, and high integrity fish communities, these watersheds are benchmarks for the URRW and warrant protection (Angermeier and Winston 1999). Between the headwaters of Bottom and Goose Creeks and the headwaters of the Blackwater River system is a plateau traversed by Highway 221 and the Blue Ridge Parkway. With its close proximity to Roanoke, this area could become more developed in the coming decades and impact the current integrity of these streams.

Mean metric scores declined precipitously as disturbed land use increased to a level between 10 and 20 % (Figure 3.5). After land use passed this threshold, mean metric scores were low regardless of whether disturbed land use constituted 30 or 60 % of land use. This trend is very similar to that found by Wang et al. (1997) and the threshold concept and effects of low levels of disturbance are supported by other studies (Klein 1979, Schueler 1994, May et al. 1997). From a planning viewpoint, this threshold concept suggests that watershed land use remain below 10 % if possible, if preserving high biological integrity and protecting rare species is the goal. However, once disturbed land use passes this threshold, stream integrity is impaired and more development will do little to worsen stream conditions because sensitive species have probably been eliminated already. Therefore, from a large scale perspective, as many watersheds as possible should be left undisturbed so that biological integrity will remain high. Future development should be concentrated in already disturbed watersheds rather than spreading across a large geographical area that will affect a large number of streams (Schueler 1994). Because continued urban development is inevitable, some watersheds must be developed, or sacrificed, so that biological integrity can remain high in other watersheds.

Because mean metric scores declined quickly but reached a plateau at 20 % disturbance, my findings suggest that streams with some integrity are better candidates for restoration than those that have very little integrity. Biological integrity will not improve noticeably if watershed and riparian conditions are improved in heavily developed watersheds. Streams that are only slightly impacted, but have high potential for improved integrity, should be restoration targets (May et al. 1997). Riparian areas along large streams are obvious restoration targets because rare species were limited to large streams and land use within a small area close to sites appeared to have the most influence on fish communities.

The North Fork of the Roanoke River and Elliott Creek are my nominations for restoration consideration. All sites within these watersheds were classified as intermediate or best and cattle grazing, which can be mitigated by riparian restoration, is the primary impact in both watersheds. Riparian restoration is already occurring at grazing sites along the North Fork (Mike Pinder, Virginia Dept. of Game and Inland Fisheries, personal communication). Elliott Creek is another important target because it is the first large tributary of the South Fork below the confluence of Goose and Bottom Creeks. During high flows, Elliott Creek is more turbid than the South Fork at the confluence (personal observation). Four orangefin madtoms were collected at Lower Elliott Creek, and the lower portions of Elliott Creek are primarily forested. However, grazing is common in the headwaters and close proximity to Christiansburg makes the area a potential target for continued residential development. Limiting land use impacts through riparian restoration will improve downstream conditions throughout the North Fork and Elliott Creek watersheds.

Sampling streams of various sizes allowed me to weigh the advantages and disadvantages of using each stream size to assess land use effects on fish communities. Small streams (10 – 15 km^2 watershed areas) are advantageous for sampling because they are numerous and can be effectively sampled without a lot of time or labor. Because their watersheds are small and the diversity of land use types is more limited, linking watershed land use with in-stream conditions and fish communities should be easier for small streams. However, physicochemical factors can have more effect on small streams than large ones (Schlosser 1990) and cloud land use relationships with fish communities. From a fish community perspective, small streams will have fewer species and are less likely to have species of concern or ecological specialists. Specialized species sensitive to degradation are good indicators of high integrity and are usually more abundant in large streams (Smale and Rabeni 1995).

Large streams (watershed areas 70 - 80 km^2) are less common and require more time and effort to sample effectively. Effective sampling is important because some species that can explain a lot about stream conditions are rare and will not be collected unless a large area is thoroughly sampled. In a confined geographical area, there may not be enough large streams of equal watershed areas to sample without sampling some of them more than once. In the URRW, no rare species were found at small and medium sites so sampling large sites was important to determine suitable conditions for these species. Conversely, there are only eight large streams so my sampling scheme would need to be modified to have a suitable sampling size if I only sampled large streams. Large field crews of eight or more people, which I felt were necessary for thorough sampling, are not possible for many projects so smaller crews are called upon to sample as effectively as possible. In conclusion, small and large sites each have advantages and disadvantages and medium size stream are able to improve the disadvantages to varying degrees. As long as researchers understand the advantages and disadvantages of each size, understand fish community trends related to stream size, and realize the important of accounting for these trends, they can choose stream sizes that are most appropriate for the questions they wish to answer.

Table 3.1.-Thirteen metrics, their abbreviations, expected response to anthropogenic degradation, and number of species included in each metric. Torrent sucker was the only intolerant species collected at small and medium sites so number of intolerant species was only included for large sites.

Metric	Metric Abbreviation	Expected Response to Degradation	Number of Species Included
Species Richness (Family Level)			
number of native species	natives	decreases	35
number of nonnative species	nonnatives	increases	14
number of darter or sculpin species	darters and sculpins	decreases	6
number of sucker species	suckers	decreases	9
number of native minnow species	minnows	decreases	13
Trophic Ecology			
prop. abundance of generalist feeders	generalists	increases	18
prop. abundance of non-benthic invertivores	invertivores	decreases	19
prop. abundance of benthic, specialist invertivores excluding tolerants	benthic invertivores	decreases	7
Reproductive Ecology			
prop. abundance of simple lithophils excluding tolerants and nest associates	simple lithophils	decreases	12
number of late maturing species (>2 years)	late maturers	decreases	12
Tolerance			
prop. abundance of tolerants	tolerants	increases	11
number of intolerant species**	intolerants	decreases	4
Individual Health			
prop. abundance of individuals with anomalies	anomalies	increases	49

** too few species to use at small and medium sites

Table 3.2.-Classification of species for each metric based on Smogor (1996). The total column indicates the number of metrics in which each species is included.

Species	natives	nonnatives	darters and sculpins	suckers	minnows	generalists	invertivores	benthic invertivores	simple lithophils	late maturers	tolerants	intolerants	anomalies	**Total**
<u>Salmonidae</u>														
rainbow trout		**1**											**1**	**2**
brown trout		**1**											**1**	**2**
brook trout		**1**											**1**	**2**
<u>Esocidae</u>														
chain pickerel	**1**												**1**	**2**
<u>Cyprinidae</u>														
central stoneroller	**1**				**1**								**1**	**3**
rosyside dace	**1**				**1**		**1**						**1**	**4**
common carp		**1**				**1**				**1**	**1**		**1**	**5**
cutlips minnow	**1**				**1**		**1**						**1**	**4**
white shiner	**1**				**1**		**1**						**1**	**4**
crescent shiner	**1**				**1**		**1**						**1**	**4**
rosefin shiner	**1**				**1**	**1**	**1**						**1**	**5**
bluehead chub	**1**				**1**	**1**				**1**			**1**	**5**
spottail shiner	**1**				**1**		**1**						**1**	**4**
swallowtail shiner	**1**				**1**		**1**		**1**				**1**	**5**
mimic shiner	**1**				**1**	**1**	**1**						**1**	**5**
mtn. redbelly dace	**1**				**1**	**1**							**1**	**4**
bluntnose minnow		**1**				**1**					**1**		**1**	**4**
fathead minnow		**1**				**1**					**1**		**1**	**4**
blacknose dace	**1**				**1**	**1**	**1**				**1**		**1**	**6**
longnose dace		**1**					**1**		**1**				**1**	**4**
creek chub	**1**				**1**	**1**					**1**		**1**	**5**
<u>Catostomidae</u>														
white sucker	**1**			**1**		**1**				**1**	**1**		**1**	**6**
northern hogsucker	**1**			**1**				**1**	**1**	**1**			**1**	**6**
Roanoke hogsucker	**1**			**1**		**1**	**1**		**1**				**1**	**6**
silver redhorse	**1**			**1**		**1**	**1**		**1**	**1**			**1**	**7**

Species	natives	nonnatives	darters and sculpins	suckers	minnows	generalists	invertivores	benthic invertivores	simple lithophils	late maturers	tolerants	intolerants	anomalies	**Total**
<u>Catostomidae</u>														
golden redhorse	**1**			**1**		**1**	**1**		**1**	**1**			**1**	**7**
v-lip redhorse	**1**			**1**		**1**	**1**		**1**	**1**			**1**	**7**
bigeye jumprock	**1**			**1**				**1**	**1**	**1**		**1**	**1**	**7**
black jumprock	**1**			**1**		**1**	**1**		**1**				**1**	**6**
torrent sucker	**1**			**1**		**1**			**1**	**1**		**1**	**1**	**7**
<u>Ictaluridae</u>														
yellow bullhead	**1**					**1**							**1**	**3**
brown bullhead	**1**					**1**				**1**			**1**	**4**
orangefin madtom	**1**							**1**				**1**	**1**	**4**
margined madtom	**1**						**1**			**1**			**1**	**4**
<u>Poecilidae</u>														
eastern mosquitofish		**1**					**1**				**1**		**1**	**4**
<u>Cottidae</u>														
mottled sculpin	**1**		**1**					**1**					**1**	**4**
<u>Centrarchidae</u>														
rock bass		**1**											**1**	**2**
redbreast sunfish	**1**												**1**	**2**
green sunfish		**1**									**1**		**1**	**3**
pumpkinseed	**1**						**1**						**1**	**3**
bluegill sunfish		**1**					**1**				**1**		**1**	**4**
smallmouth bass		**1**											**1**	**2**
spotted bass		**1**											**1**	**2**
largemouth bass		**1**											**1**	**2**
<u>Percidae</u>														
fantail darter	**1**		**1**								**1**		**1**	**4**
johnny darter	**1**		**1**								**1**		**1**	**4**
riverweed darter	**1**		**1**					**1**					**1**	**4**
Roanoke logperch	**1**		**1**					**1**	**1**	**1**		**1**	**1**	**7**
Roanoke darter	**1**		**1**					**1**	**1**				**1**	**5**

Table 3.3.-Mean metric scores for all sites ordered from lowest to highest mean metric score. Sites with scores below 0.400 were considered worst, scores from 0.400 to 0.569 were intermediate, and best sites had scores above 0.569.

Site	Mean Metric Score	Rating	Site	Mean Metric Score	Rating
Lick Run	0.231	worst	Lower Wilson Creek	0.456	intermediate
Middle Glade Creek	0.234	worst	Upper Mason Creek	0.461	intermediate
Upper Back Creek	0.252	worst	Upper Bottom Creek	0.477	intermediate
Lower Tinker Creek	0.262	worst	Upper Goose Creek	0.499	intermediate
Bradshaw Creek	0.267	worst	Upper Elliott Creek	0.505	intermediate
Lower Mudlick Creek	0.272	worst	Middle Smith Creek	0.506	intermediate
Buffalo Creek	0.284	worst	Lick Fork	0.519	intermediate
Lower Mason Creek	0.292	worst	Middle Bottom Creek	0.525	intermediate
Middle Tinker Creek	0.298	worst	Lower Back Creek	0.533	intermediate
Lower Peters Creek	0.305	worst	Falling Branch	0.559	intermediate
Middle Mason Creek	0.316	worst	Lower North Fork	0.561	intermediate
Middle Back Creek	0.318	worst	Lower Elliott Creek	0.565	intermediate
Upper Peters Creek	0.331	worst	Brake Branch	0.566	intermediate
UTWORC	0.351	worst	Upper Smith Creek	0.570	best
Wolf Creek	0.371	worst	Lower Glade Creek	0.640	best
Horners Branch	0.371	worst	Purgatory Creek	0.653	best
Laymantown Creek	0.377	worst	Upper Wilson Creek	0.667	best
Deer Branch Trib.	0.390	worst	Lower Bottom Creek	0.681	best
Upper Glade Creek	0.411	intermediate	Middle North Fork	0.685	best
Upper Mudlick Creek	0.420	intermediate	Mill Creek	0.726	best
Dry Branch	0.445	intermediate	Lower Goose Creek	0.738	best
Upper Tinker Creek	0.449	intermediate			

Table 3.4.-Mean values for 13 metrics grouped by watershed size class. Significance using the Kruskal-Wallis test determined at $\alpha = 0.05$. The Tukey studentized range test was used to determine differences among the three groups. Groups with different letters were significantly different. General trends of increased or decreased values from small to large sites are indicated for metric values significantly different with the Kruskal-Wallis test.

Metric	Mean values for each size class			*P*-value
	Small Sites	Medium Sites	Large Sites	
<u>Species Richness (Family Level)</u>				
number of native species	9.5 A	12.1 B	19.4 C	<0.0001
number of nonnative species	1.0 A	1.8 A	3.9 B	0.001
number of darter or sculpin species	1.6 A	1.8 A	3.9 B	<0.001
number of sucker species	1.8 A	2.4 A	5.3 B	<0.001
number of native minnow species	5.6 A	6.2 A	8.1 B	0.004
<u>Trophic Ecology</u>				
prop. abundance of generalist feeders	0.48	0.46	0.43	0.795
prop. abundance of non-benthic invertivores	0.17	0.21	0.28	0.038
prop. abundance of benthic, specialist invertivores excluding tolerants	0.087	0.048	0.053	0.227
<u>Reproductive Ecology</u>				
prop. abundance of simple lithophils excluding tolerants and nest associates	0.030 A	0.037 A	0.107 B	0.023
number of late maturing species (>2 years)	2.8 A	3.8 B	5.6 C	0.0001
<u>Tolerance</u>				
prop. abundance of tolerants	0.46 A	0.31 B	0.25 B	0.003
*number of intolerant species	NA	NA	1.6	
<u>Individual Health</u>				
prop. abundance of individuals with anomalies	0.015	0.019	0.023	0.109

* too few species to use at small and medium sites

Table 3.5.-Raw metric values for all 43 sites were used to assess metric utility using principal component analysis. The top part shows correlations for metric for the first three principal components. The correlation matrix shows correlations among individual metrics.

Metric	PC 1	PC 2	PC 3
number of native species	0.419	0.050	-0.043
number of nonnative species	0.305	0.157	0.317
number of darter or sculpin species	0.362	-0.071	0.151
number of sucker species	0.389	-0.151	-0.140
number of native minnow species	0.335	0.189	-0.176
proportional abundance of generalist feeders	-0.063	0.265	-0.674
proportional abundance of non-benthic invertivores	0.187	0.377	0.063
number of late maturing species (>2 years)	0.389	-0.044	-0.063
proportional abundance of tolerant species	-0.275	0.094	0.110
proportional abundance of individuals with anomalies	0.069	0.537	0.429
proportional abundance of benthic, specialist invertivores excluding tolerant species	0.036	-0.555	0.330
proportional abundance of simple lithophils excluding tolerants and nest associates	0.255	-0.298	-0.236
eigenvalue	5.413	1.679	1.345
proportion variation	0.451	0.140	0.112

Correlation Matrix Metric	natives	nonnatives	darters and sculpins	suckers	minnows	generalists	invertivores	late maturers	tolerants	anomalies	benthic invertivores	simple lithophils
natives	1.000											
nonnatives	0.631	1.000										
darters and sculpins	0.833	0.577	1.000									
suckers	0.895	0.499	0.793	1.000								
minnows	0.846	0.491	0.533	0.638	1.000							
generalists	-0.126	-0.258	-0.213	-0.075	0.053	1.000						
invertivores	0.389	0.332	0.233	0.233	0.413	0.021	1.000					
late maturers	0.885	0.618	0.702	0.880	0.669	-0.101	0.263	1.000				
tolerants	-0.579	-0.302	-0.403	-0.514	-0.452	0.019	-0.424	-0.519	1.000			
anomalies	0.163	0.378	0.221	-0.050	0.106	-0.040	0.303	0.077	0.031	1.000		
benthic invertivores	-0.021	0.007	0.179	0.112	-0.137	0.356	-0.106	0.050	-0.227	-0.227	1.000	
simple lithophils	0.505	0.342	0.480	0.609	0.292	0.001	0.059	0.507	-0.366	-0.186	0.138	1.000

Table. 3.6.-Spearman's rank order correlation coefficients (r_s) for comparisons of mean metric scores with values for individual metrics. Metrics used for correlations were removed from mean metric scores prior to correlations. Significance assessed at $\alpha = 0.05$.

Metric	Spearman's r_s	*P*-value
Species Richness (Family Level)		
number of native species	0.524	0.0003
number of nonnative species	0.186	0.233
number of darter or sculpin species	0.408	0.007
number of sucker species	0.683	<0.0001
number of native minnow species	0.377	0.013
Trophic Ecology		
proportional abundance of generalist feeders	0.054	0.731
proportional abundance of non-benthic invertivores	0.130	0.405
proportional abundance of benthic, specialist invertivores excluding tolerant species	0.521	0.0003
Reproductive Ecology		
proportional abundance of simple lithophils excluding tolerants and nest associates	0.534	0.0002
number of late maturing species (>2 years)	0.560	<0.0001
Tolerance		
proportional abundance of tolerant species	-0.429	0.004
Individual Health		
proportional abundance of individuals with anomalies	-0.166	0.287

Table 3.7.-Spearman's rank order correlation coefficents (r_s) comparing habitat and land use variables with mean metric scores. Significance was assessed at $\alpha = 0.05$; this corresponds to an absolute r_s value of 0.30. Ordered in decreasing order from strongest to weakest correlation.

Variable	r_s	*P*-value	Variable	r_s	*P*-value
site elevation	0.613	<0.0001	canopy closure	-0.237	0.126
60 %, 60 m buffer disturbed	-0.584	<0.0001	discharge	0.227	0.144
20 %, 60 m buffer disturbed	-0.569	<0.0001	D90	0.203	0.192
60 %, 60 m buffer forest	0.501	0.0006	% of silt & sand	-0.173	0.267
20 %, 60 m buffer forest	0.470	0.0015	20 %, 60 m buffer herb/ag	-0.173	0.268
watershed disturbed	-0.422	0.005	watershed area	0.172	0.271
weeds/grass/yard - site	-0.393	0.009	median depth	-0.170	0.277
proportion of pools	-0.356	0.019	riffle D50	0.156	0.324
proportion of riffles	0.322	0.036	pool D50	0.094	0.555
pasture - site	0.321	0.036	impervious surfaces - site	0.062	0.695
st dev. of substrate	0.274	0.075	watershed herb/ag	0.051	0.747
st. dev. of depth	-0.271	0.079	mean width	0.039	0.806
60 %, 60 m buffer herb/ag	-0.267	0.083	woody vegetation - site	-0.024	0.880
watershed forest	0.258	0.095	maximum observed depth	0.002	0.989
D50	0.249	0.108			

Table 3.8.-Spearman rank order correlation coefficients comparing individual metrics with habitat and watershed variables. Only r_s values significant at $\alpha = 0.05$ are shown ($|r_s| > 0.30$). Buffer land use was determined at the 60 %, 60 m scale. Total significant is a count of the number of significant correlations.

Variable	natives	nonnatives	darters or sculpins	suckers	minnows	generalists	invertivores	benthic invertivores	simple lithophils	late maturers	tolerants	anomalies	total significant
mean width	0.615	0.427	0.343	0.436	0.568		0.405			0.423	-0.367		8
canopy closure			-0.305										1
median depth		0.331										0.398	2
st. dev. of depth	0.322	0.366			0.353		0.323		-0.343			0.499	6
proportion of riffles									0.303			-0.377	2
proportion of pools				-0.340				-0.338	-0.503			0.569	4
discharge	0.397			0.447	0.342					0.478			4
D50						-0.414					-0.329		2
D90													0
st dev. of substrate					0.358				0.318				2
riffle D50													0
pool D50						-0.481							1
% of silt & sand													0
% woody veg. - site													0
% imperv. surf. - site		0.329								0.397			2
% pasture - site													0
% weed/grass/yard - site					-0.310			-0.362					2
watershed area	0.712	0.549	0.529	0.553	0.508		0.466	0.316	0.314	0.605	-0.625		10
site elevation				0.387				0.523	0.347	0.355	-0.474	-0.452	6
buffer forest								0.315	0.346		-0.513	-0.509	4
buffer herb/ag											0.378		1
buffer disturbed				-0.336				-0.537	-0.341		0.516	0.407	5
watershed forest												-0.356	1
watershed herb/ag										0.392			1
watershed disturbed						0.331		-0.442			0.482	0.332	4
total significant	4	5	3	6	6	3	3	7	8	6	8	9	68

Table 3.9.-Spearman's rank order correlation coefficients (r_s) comparing fish densities with mean metric scores. Only *P*-values < 0.05 (significance level) are reported.

Species	Spearman's r_s	*P*-value
rainbow trout	0.345	0.023
brown trout	0.232	
chain pickerel	-0.198	
central stoneroller	0.085	
rosyside dace	0.437	0.003
cutlips minnow	0.366	0.016
white shiner	0.434	0.004
crescent shiner	0.001	
rosefin shiner	0.204	
bluehead chub	0.010	
swallowtail shiner	0.212	
mtn redbelly dace	0.008	
bluntnose minnow	-0.334	0.029
blacknose dace	-0.056	
creek chub	0.068	
white sucker	-0.311	0.042
northern hogsucker	0.520	0.0003
Roanoke hogsucker	0.501	0.0006
golden redhorse	0.399	0.008
bigeye jumprock	0.398	0.008
black jumprock	0.453	0.002
torrent sucker	0.501	0.0006
orangefin madtom	0.376	0.013
margined madtom	0.345	0.023
mottled sculpin	0.657	<.0001
rock bass	0.000	
redbreast sunfish	-0.155	
green sunfish	0.248	
bluegill sunfish	-0.190	
smallmouth bass	0.291	
largemouth bass	-0.175	
fantail darter	-0.041	
johnny darter	-0.139	
riverweed darter	0.201	
Roanoke darter	0.455	0.0022

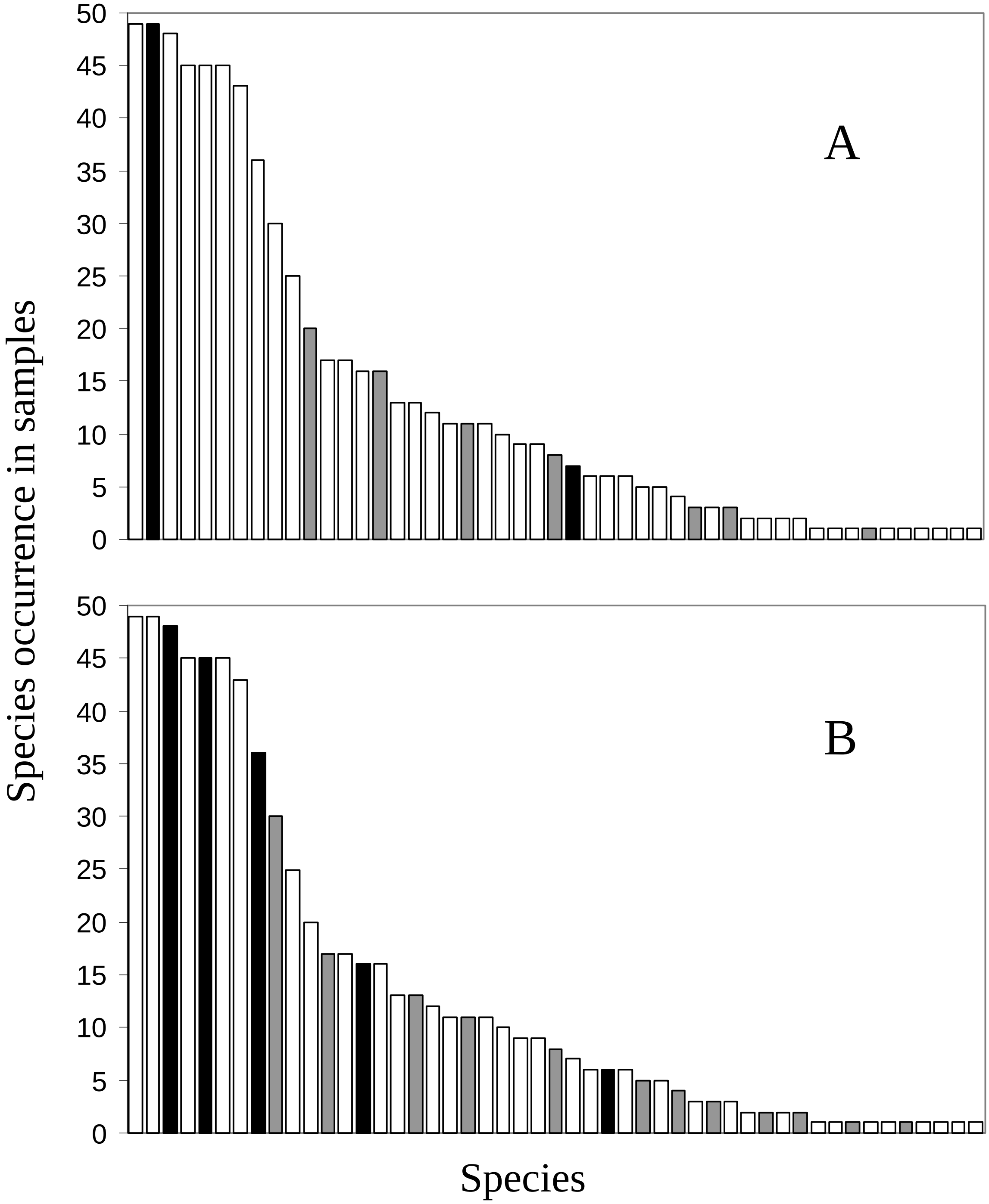


Figure 3.1.- Species occurrence for all species and samples. Species are represented by columns and ordered from most to least common. Chart A represents the benthic invertivores metric with black columns representing tolerant species that are benthic invertivores and gray columns representing other benthic invertivores. Chart B represents simple lithophils with black columns representing nest associates and gray columns representing other simple lithophils. White columns in both charts represent species not included in either metric.

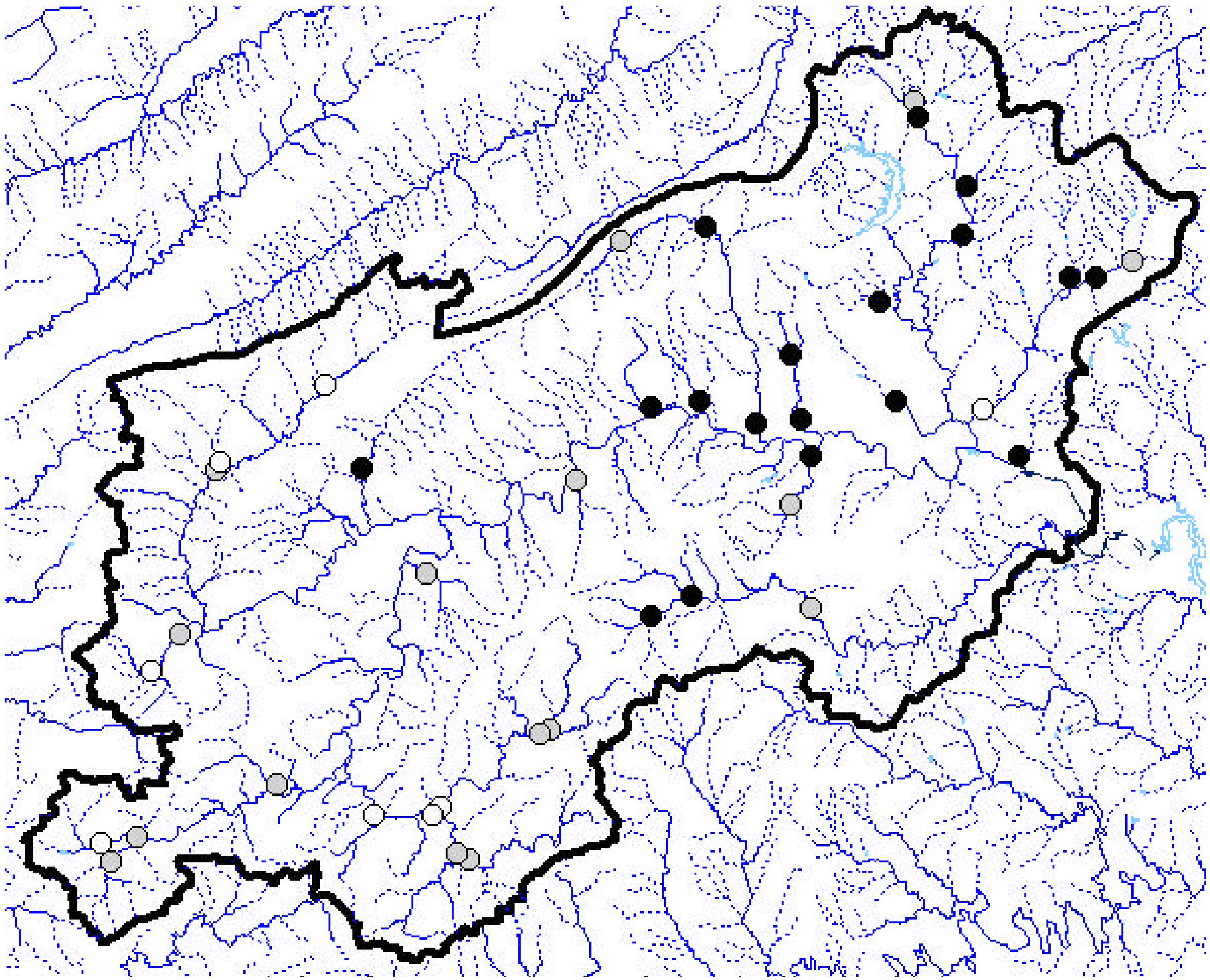


Figure 3.2.-Distribution of best, intermediate, and worst sites in the URRW based on mean metric scores. Best sites are indicated by white circles, gray circles indicate intermediate sites, and black circles indicate worst sites.

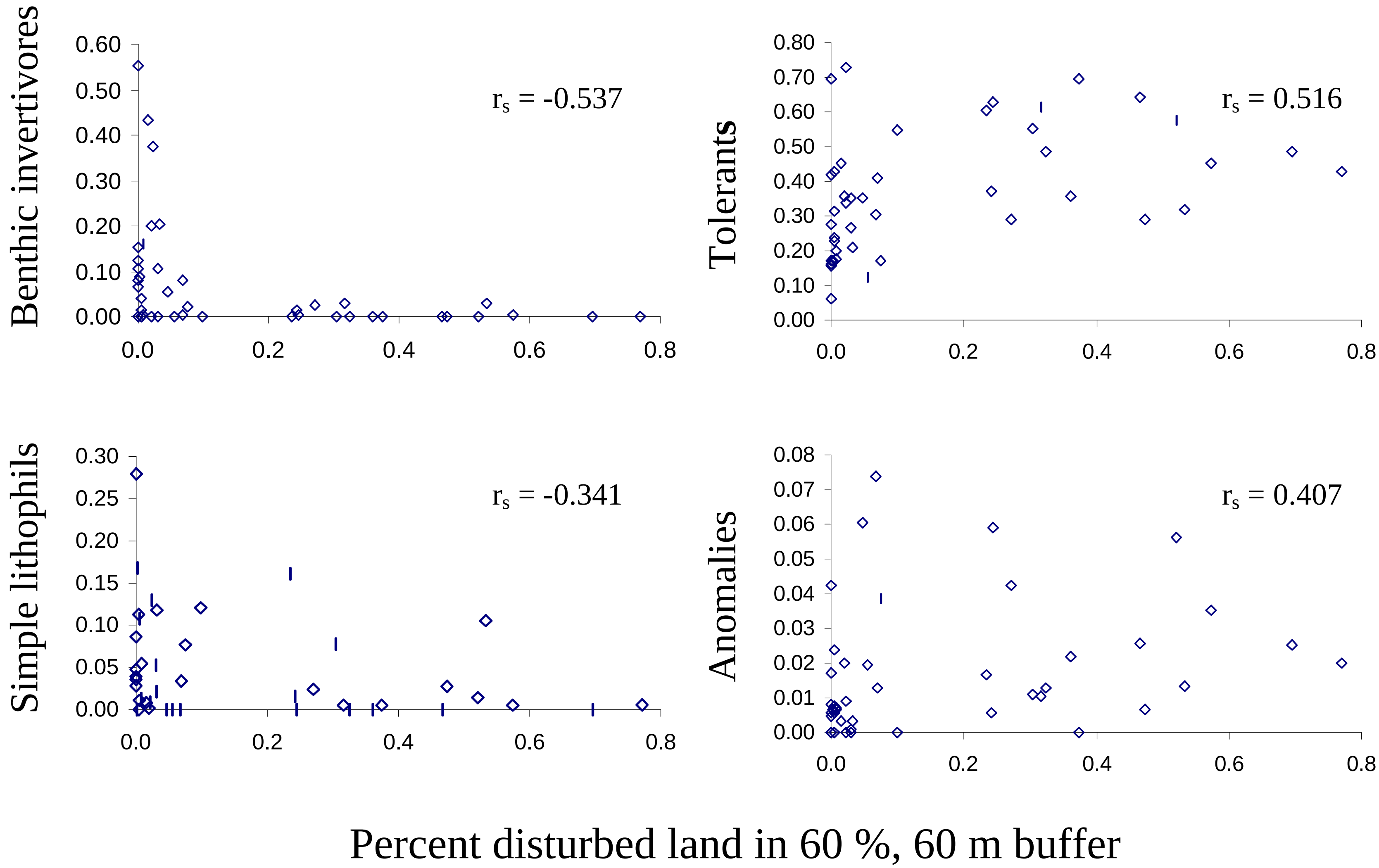


Figure 3.3.-Plots of individual metrics versus disturbed land use in the 60 %, 60 m buffer. Spearman's rank order correlation coefficients, r_s are reported for each comparison.

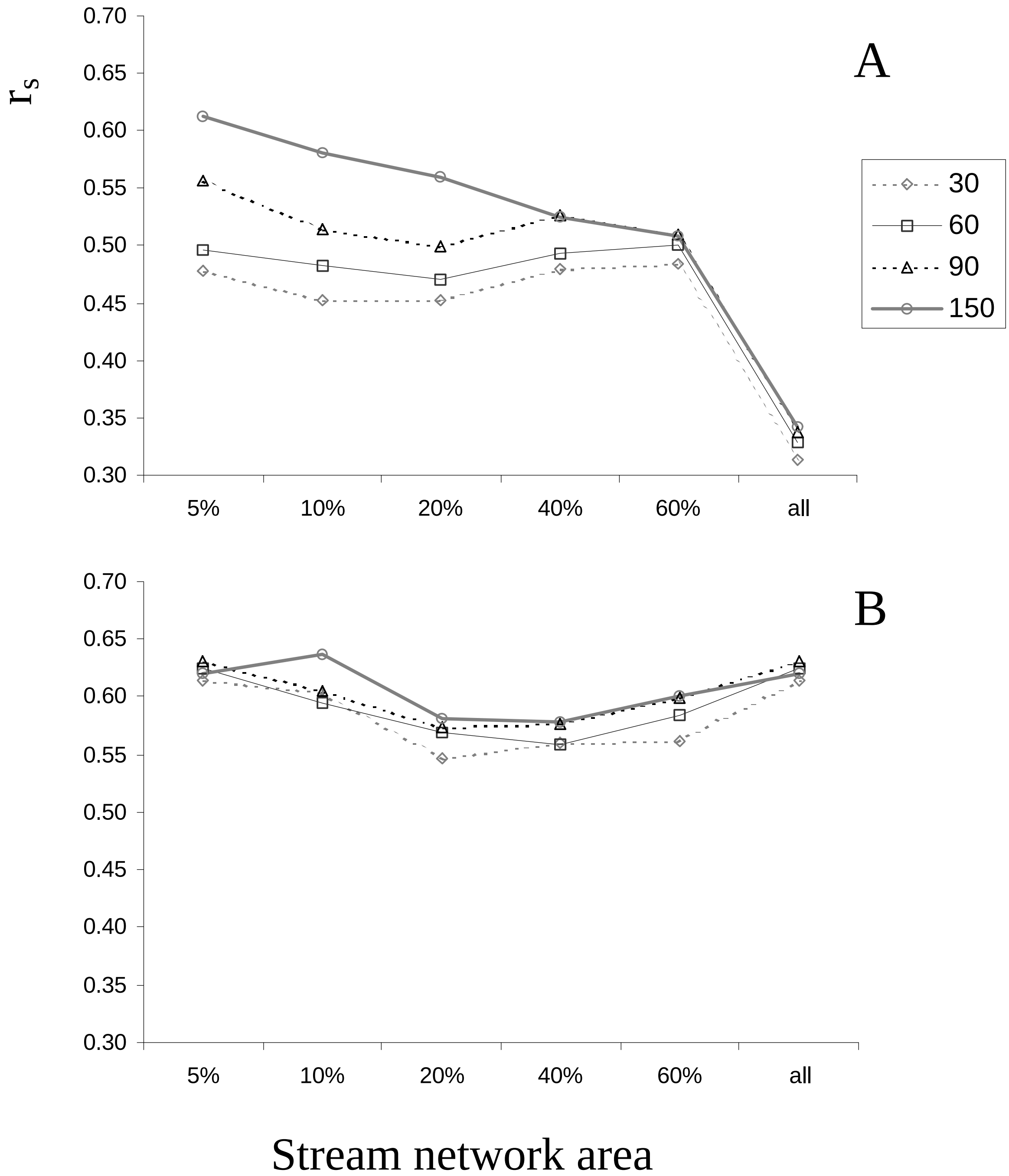
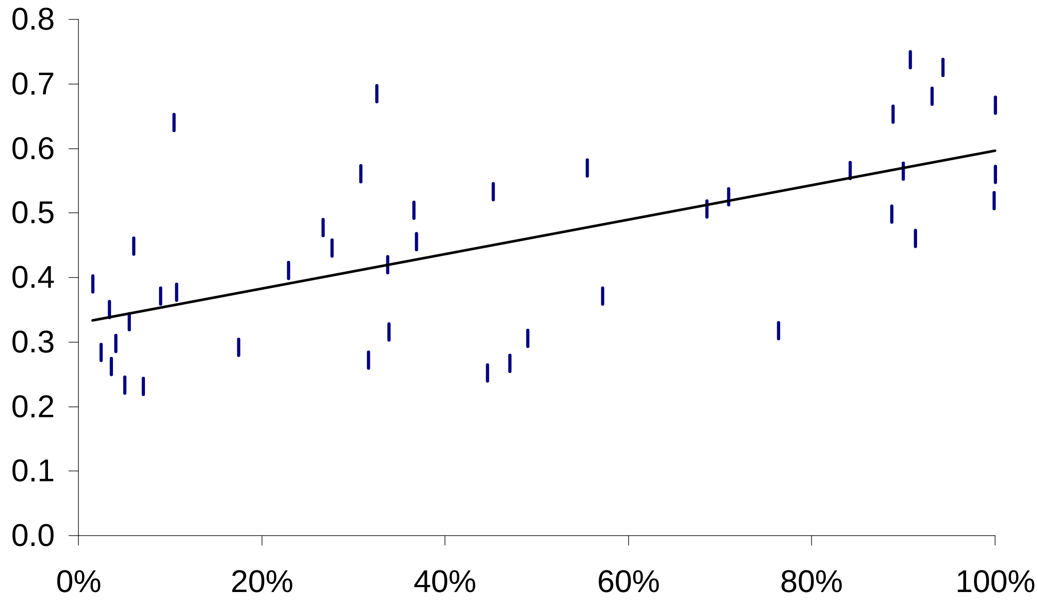
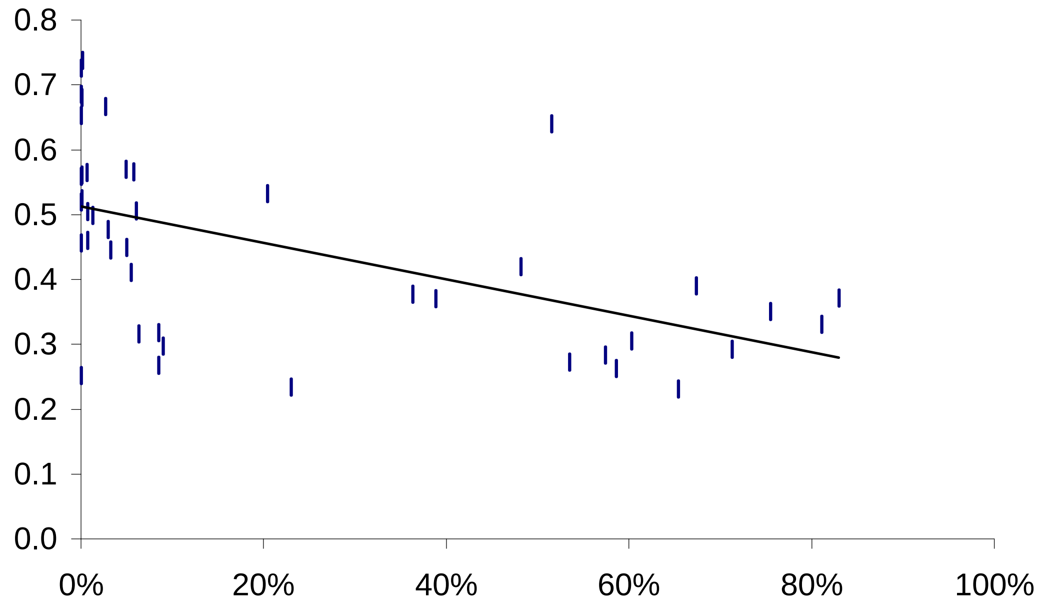


Figure 3.4.-Absolute values of Spearman's rank order correlation coefficients ($|r_s|$) comparing land use at 24 spatial scales with mean metric scores for 43 sites. Lines represent buffers from 30 to 150 m wide for different land uses: A) forest and B) disturbed. Stream network area represents circles corresponding to 5 to 60 % of mean watershed areas and "all" represents entire stream network areas. For entire watershed correlations, r_s values were 0.258 for forest, and -0.422 for disturbed. Spearman's coefficients < |0.30| are not significant at $\alpha = 0.05$.



Percent forest - 5 %, 150 m buffer



Percent disturbed - 10 %, 150 m buffer

Figure 3.5.-Correlation of mean metric scores with buffer land use showing trendline. The chosen buffers were more highly correlated with mean metric scores than all other buffers as determined using Spearman's rank order correlation coefficient (r_s) shown in Figure 3.3.

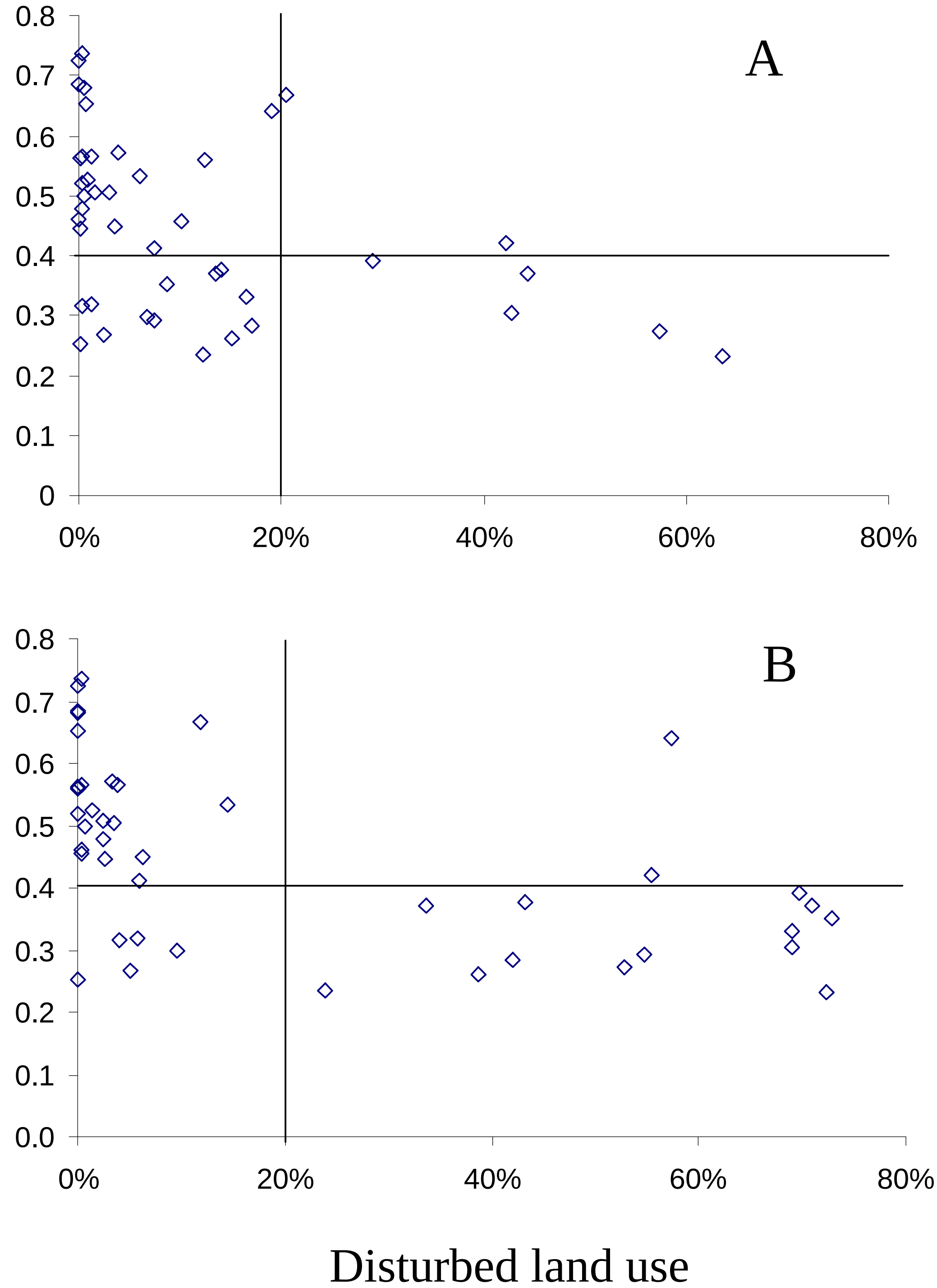


Figure 3.6.-Scatterplots of mean metric scores at sites with varying amounts of disturbed land use in entire watersheds (A) and within 20 %, 150 m buffers (B). Horizontal lines are positioned at mean metric scores of 0.400, which is the boundary between intermediate and worst sites.

CONCLUSIONS

SUMMARY

My results showed that fish communities in upper Roanoke River tributaries are affected by several factors operating at different spatial scales. Proportions of forest and disturbed land within watersheds had a large influence on in-stream habitat conditions and fish communities. Fish communities in streams draining forested watersheds and buffers had high biological integrity. This integrity declined as proportions of forested land declined and disturbed land use increased. Correlations of herb/ag land with habitat conditions and fish community attributes were weaker and most correlations were insignificant. Correlations of forested land with mean metric scores were strongest for wide buffers and small stream network areas. Disturbed land use was strongly correlated with mean metric scores regardless of buffer size. Mean metric scores declined precipitously as disturbed land use increased and then leveled out at disturbance levels above 20 %.

Several factors, such as stream elevation, stream depth, particle size, stream width, and land use had significant influences on fish communities. Mean metric scores were highest at high elevation sites and declined as elevation decreased. Other natural differences among sites, such as proportions of riffles and pools, depth, mean width and watershed area, had significant influences on fish communities. Species richness generally increased as stream size increased. Unlike many previous studies, few correlations comparing substrate variables with fish communities were significant (Berkman and Rabeni 1987, Rabeni and Smale 1995). Disturbed land use, depth, and proportion of pools increased significantly at lower site elevations. Conversely, proportion of riffles, and forested land use increased at high elevation streams. The interconnectedness of these variables made definitively identifying which factors have the most influence on fish communities difficult.

Like previous studies in the URRW, I found a strong trend of increased species richness from small to large streams (Jenkins and Freeman 1972, Hambrick 1973). Small sites averaged 10.8 species, medium sites averaged 13.8 species, and large sites averaged 23.3 species. Species such as fantail darters, bluehead chubs, mountain redbelly dace, white suckers, and crescent shiners were collected at nearly all sites regardless of stream size or quality. Conversely, several species such as orangefin madtoms and bigeye jumprocks were only collected at large sites. In addition to the widely expected increase in species richness, functional attributes of fish communities showed strong trends related to stream size. On average, specialized species increased in abundance from small to large sites while species tolerant of degraded conditions declined with increased stream size. These results underscore the importance of separating sites by watershed area before comparing fish communities. Also, components of the fish community, such as simple lithophils, benthic invertivores, and tolerant species, changed in abundance from small to large streams. In addition to accounting for species richness changes, trends related to functional aspects of fish communities should be considered when evaluating the integrity of streams that differ in size.

Mean metric scores increased linearly as the amount of forest in watersheds and buffers increased. Also, amounts of forest were significantly higher at sites where Roanoke hogsuckers and mottled sculpins were present than at sites where they were not collected. There were no significant trends between herb/ag land use and mean metric scores at any spatial scales. Some sites in agricultural areas had low integrity while others had high integrity. Mean metric scores declined precipitously as disturbed land use at sites increased from 0 to 20 %. After reaching a threshold at disturbance levels of 10 to 20 %, mean metric scores were low regardless of the amount of disturbed land. This trend was true for analysis using disturbed land use in entire watershed and in buffers.

Mean metric scores were more highly correlated with proportions of forest within 5 %, 150 m buffers than with forest at any other spatial scale ($r_s = 0.612$). As larger stream networks were considered, r_s values declined. Mean metric scores were more highly correlated with forest land use in 150 m buffers and r_s values declined for smaller buffer widths, regardless of stream network area. Correlations between mean metric scores and disturbed land use were more uniform regardless of buffer area (r_s ranged from -0.546 to -0.637). Forested land use for entire watersheds was not significantly correlated with mean metric scores ($r_s = 0.258$), but disturbed land use at watershed levels was significantly correlated with mean metric scores ($r_s = -0.422$).

Comparisons of different land use types at 24 spatial scales with substrate size represented by D50 values indicated that conditions far from sites have strong influences on substrate size. Absolute r_s values steadily increased for forest, herb/ag, and disturbed land use as stream network area increased. Additionally, watershed forest ($r_s = 0.698$) and watershed disturbed ($r_s = -0.597$) were better predictors of D50 values than buffer variables for each land use. Therefore, entire watershed land use was a better predictor of substrate size and correlations declined for comparisons using smaller land areas. Conversely, comparisons of land use at different scales with mean metrics scores indicated that land use within small stream network areas and wide buffers was a better predictor of biological integrity and correlations weakened as larger land areas were used for comparisons.

In conclusion, land use practices have a large influence on stream quality and fish communities. Unfortunately, I was unable to adequately determine the pathways through which land use conditions are manifested at local scales that influence fishes. Despite the lack of connection between in-stream conditions and fish communities, land use conditions clearly are driving factors influencing fishes. Improving conditions in upper Roanoke River tributaries will require improving conditions along stream buffers so that natural stream functions will be restored. Buffers along streams with healthy fish communities need to be maintained and enhanced so that current biological integrity will remain high in the future.

IMPLICATIONS AND RECOMMENDATIONS

My study produced results that can benefit both stream ecologists and urban planners. I found strong trends of species addition from small to large sites. Many functional attributes of fish communities related to feeding and reproduction also changed consistently from small to large sites. Researchers need to consider such trends when selecting sites and comparing biological integrity among sites. The mean metric score approach successfully differentiated among sites with varying degrees of integrity without relying on metric standards established elsewhere. This approach is useful for sampling streams confined to small, unique areas or when assessing the relative quality of fish communities at different sites. My results suggest that maintaining intact buffers around streams is important for fish community integrity but entire watershed-scale management is not necessary. Large upper Roanoke River tributaries with above average biological integrity are the best candidates for stream restoration. Relationships between disturbed land use and mean metric scores suggest that planning at large scales should aim to protect currently forested watershed and limit development to watersheds that are already impacted.

Sampling streams within three different watershed size classes enabled me to assess differences in species abundance and distribution related to stream size. I found a strong trend of species addition as stream size increased. Researchers should choose sites with similar watershed areas when selecting sample sites so that natural variation related to stream size is controlled. Different size streams should not be lumped into one data set for analysis without considering increases in species richness that will occur at larger sites. Additionally, I found a general trend of increased specialists (e.g., simple lithophils and benthic invertivores) and a decrease in tolerant species from small to large sites. Therefore, merely adjusting species richness expectations is not enough to account for the natural changes in fish communities. Researchers should also account for changes in functional attributes with stream size. Failure to account for these changes can cause high quality small streams to be rated lower than they deserve or cause large, depauperate streams to be consider better than they actually are.

The mean metric score approach proved to be a useful method for combining many metrics to create one number that represents fish community integrity. I mitigated the effects of fish community changes with stream size by separating sites by watershed size class before calculating mean metric scores. Although this approach can only truthfully express relative fish community quality (i.e., best, worst) instead of absolute quality (excellent, poor), it can effectively assess fish communities in areas where established IBIs do not exist or would be inappropriate.

Fish community attributes were highly correlated with forest and disturbed land use within buffers, but this correlation declined when these land uses were considered for entire watersheds. Forested buffer zones are important for fish and should be protected where they are currently adequate and created or enhanced along streams that are not currently protected. As buffers increased in stream network area, correlations between percent forest and mean metric scores declined. This suggests that only nearby areas need to be protected. The problem is that improving buffer conditions within confined areas will only benefit fish communities close by and a short distance downstream. Forested conditions in headwaters benefit fishes downstream, but the benefits wane with increased distance and local impacts can offset advantages derived from forested headwaters. To protect all stream reaches, extensive areas of protected buffers throughout stream networks are needed. If only certain stream reaches are candidates for improvements, (i.e., reaches containing endangered species), buffers close to these stream reaches should receive higher priority than buffer areas far away. Additionally, common sense says that improving conditions along some stream reaches will do little good if perturbations elsewhere are left unchecked.

My findings suggest that streams that are only slightly impacted and have high potential for improved integrity should be restoration targets (May et al. 1997). Riparian areas along large streams are obvious restoration targets because rare species were limited to large streams and land use within a small area close to sites appeared to have the most influence on fish communities. Large streams with above average integrity, especially those impacted by grazing, are good candidates for restoration because they have the best potential for riparian improvements. These streams are more likely to contain rare species and improving conditions near stream reaches with potential to harbor such species is an effective way to improve their populations as well as fish community integrity.

As the human population continues to increase and spread over the landscape, more forests and fields are developed. Planners need to determine ways to support a growing population while minimizing effects on our natural resources. Planners should take a watershed approach to development. My results suggest that from a large-scale perspective, development concentrated in a few watersheds will have less overall impact on stream integrity and rare species than small amounts of development spread evenly throughout the landscape. Relationships between disturbed land use and mean metric scores exhibited a threshold at disturbance levels of 10 to 20 % because mean metric scores began to decline at very low levels of disturbance but leveled off after reaching disturbance levels of 10 to 20 %. Sites greater than 20 % disturbed were equally poor regardless of amounts of various land uses. Therefore, from a large-scale perspective, streams with degraded watersheds will not decline from their already low integrity if more development occurs in the watershed. However, streams that are currently pristine and represent best possible biological conditions will be impacted by small amounts of disturbance. Sensitive species will be unable to survive in these degraded streams and over time piecemeal extirpation of species from stream reaches will result in extinction from larger areas. If protecting streams with biological integrity is the goal, some streams, preferably ones that are already degraded, must be sacrificed so that others can remain pristine, serve as harbors for rare, sensitive species, and preserve the interactions of aquatic communities with high biological integrity.

LITERATURE CITED

Aadland, L. P. 1993. Stream habitat types: their fish assemblages and relation to flow. North American Journal of Fisheries Management 13:790-806.

Allan, J. D., D. L. Erickson, and J. Fay. 1997. The influence of catchment land use on stream integrity across multiple spatial scales. Freshwater Biology 37:149-161.

Allan, J. D., and A. S. Flecker. 1993. Biodiversity conservation in running waters. BioScience 43:32-43.

Angermeier, P. L. 1995. Ecological attributes of extinction-prone species: Loss of freshwater fishes of Virginia. Conservation Biology 9:143-158.

Angermeier, P. L., and J. R. Karr. 1984. Relationships between woody debris and fish habitat in a small warmwater stream. Transactions of the American Fisheries Society 113:716-726.

Angermeier, P. L., and J. R. Karr. 1986. Applying an index of biotic integrity based on stream-fish communities: considerations in sampling and interpretation. North American Journal of Fisheries Management 6:418-429.

Angermeier, P. L., and R. A. Smogor. 1995. Estimating number of species and relative abundances in stream-fish communities: effects of sampling effort and discontinuous spatial distributions. Canadian Journal of Fisheries and Aquatic Sciences 52:936-949.

Angermeier, P. L., and M. R. Winston. 1999. Characterizing fish community diversity across Virginia landscapes: prerequisite for conservation. Ecological Applications 9:335-349.

Berkman, H. E., and C. F. Rabeni. 1987. Effect of siltation on stream fish communities. Environmental Biology of Fishes 18:285-294.

Beschta, R. L., and W. S. Platts. 1986. Morphological features of small streams: significance and function. Water Resources Bulletin 22:369-379.

Booth, D. B. 1990. Stream-channel incision following drainage-basin urbanization. Water Resources Bulletin 26:407-417.

Bramblett, R. G., and K. D. Fausch. 1991. Variable fish communities and the index of biotic integrity in a western Great Plains river. Transactions of the American Fisheries Society 120:752-769.

Bruton, M. N. 1995. Have fishes had their chips? the dilemma of threatened fishes. Environmental Biology of Fishes 43:1-27.

Castelle, A. J., A. W. Johnson, and C. Conolly. 1994. Wetland and stream buffer size requirements – a review. Journal of Environmental Quality 23:878-882.

Crumby, W. D., M. A. Webb, F. J. Bulow, and H. J. Cathey. 1990. Changes in biotic integrity of a river in north-central Tennessee. Transactions of the American Fisheries Society 119:885-893.

Davies, P. E., and M. Nelson. 1994. Relationships between riparian buffer widths and the effects of logging on stream habitat, invertebrate community composition and fish abundance. Australian Journal of Marine and Freshwater Research 45:1289-1305.

Detenbeck, N. E., P. W. DeVore, G. J. Niemi, A. Lima. 1992. Recovery of temperate-stream fish communities from disturbance: a review of case studies and synthesis of theory. Environmental Management 16:33-53.

Dickson, K. L. 1979. Aquatic ecological classification and evaluation of streams and stream reaches in the upper Roanoke River basin, Virginia. Wilmington District Corps of Engineers.

Dietz, R. W. 2000. The use of GIS for integrated watershed analysis: Integration of environmental models with GIS in the upper Roanoke River watershed. Master's thesis. Virginia Polytechnic Institute and State University, Blacksburg. 81pp.

Ensign, W. E., K. N. Leftwich, P. L. Angermeier, and C. A. Dolloff. 1997. Factors influencing stream fish recovery following a large-scale disturbance. Transactions of the American Fisheries Society 126:895-907.

Evans, J. W. and R. L. Noble. 1979. The longitudinal distribution of fishes in an east Texas stream. The American Midland Naturalist 101:333-343.

Fausch, K. D., J. R. Karr, and P. R. Yant. 1984. Regional application of an index of biotic integrity based on stream fish communities. Transactions of the American Fisheries Society 113:39-55.

Frissell, C. A., Liss, W. J., Warren, C. E., and Hurley, M. D. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental Management 10(2):199-214.

Gorman, O. T., and J. R. Karr. 1978. Habitat structure and stream fish communities. Ecology 59:507-515.

Gregory, S. V., F. J. Swanson, W. A. McKee, K. W. Cummins. 1991. An ecosystem perspective of riparian zones. BioScience 41:540-551.

Gutzwiller, L. A., R. M. McNatt, and R. D. Price. 1997. Watershed restoration and grazing practices in the Great Basin: Marys River of Nevada. Pages 360-380 *in* J. E. Williams, C. A. Wood, and M. P. Dombeck, editors. Watershed restoration: principles and practices. American Fisheries Society, Bethesda, Maryland.

Hambrick, P. S. 1973. Composition, longitudinal distribution and zoogeography of the fish fauna of Back Creek, Blackwater River and Pigg River, tributaries of the Roanoke River in south-central Virginia. Master's Thesis, Virginia Polytechnic Institute and State University, Blacksburg. 95 pp.

Harding, J. S., E. F. Benfield, P. V. Bolstad, G. S. Helfman, and E. B. D. Jones, III. 1998. Stream biodiversity: the ghost of land use past. Proceedings of the National Academy of Science. 95:14843-14847.

Harrelson, C. C., C. L. Rawlins, and J. P. Potyondy. 1994. Stream channel reference sites: An illustrated guide to field technique. General Technical Report RM-245, U. S. Department of Agriculture, Forest Service, Rocky Mountain Forest and Range Experiment Station.

Harvey, B. C. 1987. Susceptibility of young-of-the-year fishes to downstream displacement by flooding. Transactions of the American Fisheries Society 116:851-855.

Hocutt, C. H. 1981. Fish as indicators of biological integrity. Fisheries (Bethesda) 6(6):28-31.

Hunsaker, C. T., and D. A. Levine. 1995. Hierarchical approaches to the study of water quality in rivers. BioScience 45:193-203.

Imhof, J. G., J. Fitzgibbon, and W. K. Annable. 1996. A hierarchical evaluation system for characterizing watershed ecosystems for fish habitat. Canadian Journal of Fisheries and Aquatic Sciences 53(Supplement 1):312-326.

Jenkins, R. E., and C. A. Freeman. 1972. Longitudinal distribution and habitat of the fishes of Mason Creek, an upper Roanoke River drainage tributary, Virginia. The Virginia Journal of Science 23:194-202.

Jenkins. R. E., and N. M. Burkhead. 1993. Freshwater fishes of Virginia. American Fisheries Society, Bethesda, Maryland. 1079 pp.

Jones, R. C., and C. C. Clark. 1987. Impact of watershed urbanization on stream insect communities. Water Resources Bulletin 23:1047-1055.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries (Bethesda) 6(6):21-27.

Karr, J. R., and E. W. Chu. 1997. Biological monitoring and assessment: using multimetric indexes effectively. University of Washington, United States Environmental Protection Agency EPA 235-R97-001, Seattle. 149 pp.

Karr, J. R., K. D. Fausch, P. L. Angermeier, P. R. Yant, and I. J. Schlosser. 1986. Assessing biological integrity in running waters: a method and its rationale. Illinois Natural History Survey, Champaigne, Illinois, Special Publication 5.

Karr, J. R., P. R. Yant, K. D. Fausch, and I. J. Schlosser. 1987. Spatial and temporal variability of the index of biotic integrity in three midwestern streams. Transactions of the American Fisheries Society 116:1-11.

Kauffman, J. B., R. L. Beschta, N. Otting, and D. Lytjen. 1997. An ecological perspective of riparian and stream restoration in the western United States. Fisheries (Bethesda) 22(5):12-24.

Kauffman, J. B., and W. C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management implications... a review. Journal of Range Management 37:430-437.

Kinsolving, A. D., and M. B. Bain. 1993. Fish assemblage recovery along a riverine disturbance gradient. Ecological Applications 3:531-544.

Klein, R. D. 1979. Urbanization and stream quality impairment. Water Resources Bulletin 15:948-963.

Kondolf, G. M., and S. Li. 1992. The pebble count technique for quantifying surface bed material size in instream flow studies. Rivers 3:80-87.

Krebs, C. J. 1989. Ecological methodolgy. Harper and Row, New York.

Lenat, D. R., and J. K. Crawford. 1994. Effects of land use on water quality and aquatic biota of three North Carolina Piedment streams. Hydrobiologia 294:185-199.

Leonard, P. M., and D. J. Orth. 1986. Application and testing of an index of biotic integrity in small, coolwater streams. Transactions of the American Fisheries Society 115:401-414.

Lyons, J. 1992. The length of stream to sample with a towed electrofishing unit when fish species richness is estimated. North American Journal of Fisheries Management 12:198-203.

Lyons, J., S. Navarro-Pérez, P. A. Cochran, E. Santana C., and M. Guzmán-Arroyo. 1995. Index of biotic integrity based on fish assemblages for the conservation of streams and rivers in west-central Mexico. Conservation Biology 9:569-584.

Matthews, W. J., J. R. Bek, and E. Surat. 1982. Comparative ecology of the darters Etheostoma podostemone, E. flabellare and Percina roanoka in the upper Roanoke River drainage, Virginia. Copeia 1982:805-814.

Matthews, W. J., and J. T. Styron, Jr. 1981. Tolerance of headwater vs. mainstream fishes for abrupt physicochemical changes. American Midland Naturalist 105:149-158.

May, C. W., R. R. Horner, J. R. Karr, B. W. Mar, and E. G. Welch. 1997. Effects of urbanization on small streams in the Puget Sound lowland ecoregion. Watershed Protection Techniques 2:483-494.

McDonnell, M. J., and S. T. A. Pickett. 1990. Ecosystem structure and function along urban-rural gradients: an unexploited opportunity for ecology. Ecology 71:1232-1237.

Miller, D. L., P. M. Leonard, R. M. Hughes, J. R. Karr, P. B. Moyle, L. H. Schrader, B. A. Thompson, R. A. Daniels, K. D. Fausch, G. A. Fitzhugh, J. R. Gammon, D. B. Halliwell, P. L. Angermeier, and D. J. Orth. 1998. Regional applications of an index of biotic integrity for use in water resource management. Fisheries (Bethesda) 13:12-20.

Moglen, G. E. 2000. Urbanization, stream buffers, and stewardship in Maryland. Watershed Protection Techniques 3:676-680.

Naiman, R. J., H. Décamps, and M. Pollock. 1993. The role of riparian corridors in maintaining regional biodiversity. Ecological Applications 3:209-212.

Oberdorff, T., and R. M. Hughes. 1992. Modification of an index of biotic integrity based on fish assemblages to characterize rivers of the Seine Basin, France. Hydrobiologia 228:117-130.

Osborne, L. L., and D. A. Kovacic. 1993. Riparian vegetated buffer strips in water-quality restoration and stream management. Freshwater Biology 29:243-258.

Pearsons, T. N., H. W. Li, and G. A. Lamberti. 1992. Influence of habitat complexity on resistance to flooding and resilience of stream fish assemblages. Transactions of the American Fisheries Society 121:427-436.

Poff, N. L., and J. D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology 76:606-627.

Poff, N. L., and J. V. Ward. 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patterns. Canadian Journal of Fisheries and Aquatic Sciences 46:1805-1818.

Rabeni, C. F., and M. A. Smale. 1995. Effects of siltation on stream fishes and the potential mitigating role of the buffering riparian zone. Hydrobiologia 303:211-219.

Rankin, E. T. 1989. The qualitative habitat evaluation index (QHEI): rationale, methods, and application. Ohio Environmental Protection Agency, Ecological Assessment Section, Division of Water Quality and Assessment, Columbus.

Richards, C., and G. Host. 1994. Examining land use influences on stream habitats and macroinvertebrates: a GIS approach. Water Resources Bulletin 30:729-738.

Ryan, P. A. 1991. Environmental effects of sediment on New Zealand streams: a review. New Zealand Journal of Marine and Freshwater Research 25:207-221.

Sanders, R. E., R. J. Miltner, C. O. Yoder, and E. T. Rankin. 1999. The use of external deformities, erosion, lesions, and tumors (DELT anomalies) in fish assemblages for characterizing aquatic resources: A case study of seven Ohio streams. Pages 225-246 in T. P. Simon, editor. Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Boca Raton, Florida.

SAS Institute Inc. 1999. SAS OnlineDoc(TM), Version 7-1 Cary, NC: SAS Institute Inc.

Schlosser, I. J. 1985. Flow regime, juvenile abundance, and the assemblage structure of stream fishes. Ecology 66:1484-1490.

Schlosser, I. J. 1990. Environmental variation, life history attributes, and community structure in stream fishes: implications for environmental management and assessment. Environmental Management 14:621-628.

Schlosser, I. J. 1991. Stream fish ecology: a landscape perspective. BioScience 41(10):704-713.

Schlosser, I. J. 1995. Critical landscape attributes that influence fish population dynamics in headwater streams. Hydrobiologia 303:71-81.

Schueler, T. 1994. The importance of imperviousness. Watershed Protection Techniques 1:100-111.

Sheldon, A. L. 1968. Species diversity and longitudinal succession in stream fishes. Ecology 49:193-198.

Sheldon, A. L. 1988. Conservation of stream fishes: patterns of diversity, rarity, and risk. Conservation Biology 2:149-156.

Simonson, T. D., J. Lyons, and P. D. Kanehl. 1994. Quantifying fish habitat in streams: transect spacing, sample size, and a proposed framework. North American Journal of Fisheries Management 14:607-615.

Smale, M. A., and C. F. Rabeni. 1995. Influences of hypoxia and hypothermia on fish species composition in headwater streams. Transactions of the American Fisheries Society 124:711-725.

Smith, R. K. 1999. Differential stability of spawning microhabitats of warmwater stream fishes. Master's thesis. Virginia Polytechnic Institute and State University, Blacksburg. 100pp.

Smogor, R. A. 1996. Developing an index of biotic integrity (IBI) for warmwater wadeable streams in Virginia. Master's thesis. Virginia Polytechnic Institute and State University, Blacksburg. 126pp.

Smogor, R. A., and P. L. Angermeier. 1999a. Effects of drainage basin and anthropogenic disturbance on relations between stream size and IBI metrics in Virginia. Pages 249-272 in T. P. Simon, editor. Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Boca Raton, Florida.

Smogor, R. A., and P. L. Angermeier. 1999b. Relations between fish metrics and measures of anthropogenic disturbance in three IBI regions in Virginia. Pages 585-610 in T. P. Simon, editor. Assessing the sustainability and biological integrity of water resources using fish communities. CRC Press, Boca Raton, Florida.

Steedman, R. J. 1988. Modification and assessment of an index of biotic integrity to quantify stream quality in southern Ontario. Canadian Journal of Fisheries and Aquatic Sciences 45:492-501.

Surat, E. M., W. J. Matthews, and J. R. Bek. 1982. Comparative ecology of Notropis albeolus, N. ardens and N. cerasinus (Cyprinidae) in the upper Roanoke River drainage, Virginia. American Midland Naturalist 107:13-24.

Vadas, R. L., Jr. 1994. Habitat tools for assessing instream-flow needs for fishes in the upper Roanoke River, Virginia. Doctoral dissertation. Virginia Polytechnic Institute and State University. 346 pages.

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37:130-137.

Wang, L., J. Lyons, P. Kanehl, and R. Gatti. 1997. Influences of watershed land use on habitat quality and biotic integrity in Wisconsin streams. Fisheries (Bethesda). 22(6):6-12.

Walser, C. A., and H. L. Bart, Jr. 1999. Influence of agriculture on in-stream habitat and fish community structure in piedmont watersheds of the Chattahoochee River system. Ecology of Freshwater Fish 8:237-246.

Weaver, L. A., and G. C. Garman. 1994. Urbanization of a watershed and historical changes in a stream fish assemblage. Transactions of the American Fisheries Society 123:162-172.

Whiteside, B. G., and R. M. McNatt. 1972. Fish species diversity in relation to stream order and physicochemical conditions in the Plum Creek drainage basin. American Midland Naturalist 88:90-101.

Williams, J. E., J. E. Johnson, D. A. Hendrickson, S. Contreras-Balderas, J. D. Williams, M. Navarro-Mendoza, D. E. McAllister, and J. E. Deacon. 1989. Fishes of North America endangered, threatened, or of special concern: 1989. Fisheries (Bethesda) 14:2-20.

Williams, J. E., C. A. Wood, and M. P. Dombeck, editors. 1997. Watershed restoration: principles and practices. American Fisheries Society, Bethesda, Maryland.

Wohl, N. E., and R. F. Carline. 1996. Relations among riparian grazing, sediment loads, macroinvertebrates, and fishes in three central Pennsylvania streams. Canadian Journal of Fisheries and Aquatic Sciences 53(Supplement 1):260-266.

Yount, J. D., and G. J. Niemi. 1990. Recovery of lotic communities and ecosystems from disturbance - a narrative review of case studies. Environmental Management 14:547-569.

Appendix A.-Descriptions of all 43 sites ordered by sampling date. References to left or right stream side are determined when looking downstream. The end of the site refers to the upstream end while the start or beginning is the downstream site limit.

VS001 UTWORC (Unnamed Tributary West Of Roanoke College). Sampled 21 May 1998. Salem Quad. Site Length: 141 m. Site begins beside Cypress St. and ends approximately 80 m below Burwell St.

VS002 & VS031 Dry Branch. Sampled 26 May 1998 and 2 June 1999. Glenvar Quad. Site Length: 123 m. Site begins approximately 50 meters upstream of railroad tracks and ends approximately 30 m below the Rt. 612 bridge.

VS003 Horners Branch. Sampled 28 May 1998. Salem Quad. Site Length: 120 m. Site is located below the confluence of Horners Branch and Paint Bank Branch and above railroad tracks. It can be accessed via businesses along Hwy. 460/11 East (West Main).

VS004 & VS032 Purgatory Creek. Sampled 1 June 1998 and 3 June 1999. Pilot Quad. Site Length: 167 m. Site begins approximately 200 m above the confluence with the South Fork at the Alta Mons Camp on Rt. 637.

VS005 Falling Branch. Sampled 2 June 1998. Pilot Quad. Site Length: 162 m. Site begins approximately 50 m above the confluence with Elliott Creek and approximately 10 m above the culvert bridge crossing Split Rail Road.

VS006 and VS033 Mill Creek (North Fork). Sampled 4 June 1998 and 7 June 1999. McDonald's Mill Quad. Site length: 154 m. Site begins approximately 100 m above the confluence with the North Fork and approximately 80 m above Rt. 785.

VS007 Upper Back Creek. Sampled 9 June 1998. Bent Mountain Quad. Site Length: 146 m. Site begins approximately 10 m above the confluence with Martins Creek and ends just below a swimming pool intake just off of Hwy. 221.

VS008 Upper Mason Creek. Sampled 10 June 1998. Salem Quad. Site Length: 188 m. Site ends just below a bridge leading to a small group of houses off of Rt. 622.

VS009 Upper Mudlick Creek. Sampled 11 June 1998. Bent Mountain Quad. Site Length: 143 m. Site is located in Garst Mill Park and begins at the left field edge of the softball field at the end of the parking lot.

VS010 Upper Tinker Creek. Sampled 16 June 1998. Daleville Quad. Site Length: 157 m. Site is located in a cattle pasture and ends below a large pool below Rt. 672.

VS011 Buffalo Creek. Sampled 17 June 1998. Daleville Quad. Site Length: 170 m. Site ends approximately 50 m downstream of an I-81 entrance ramp and can be accessed through a large pasture along Tinker Mountain Rd.

VS012 & VS050 Upper Wilson Creek. Sampled 18 June 1998 and 16 September 1999. Blacksburg Quad. Site Length: 143 m. Site begins approximately 10 m above the Rt. 723 bridge.

VS013 Deer Branch Tributary. Sampled 24 June 1998. Roanoke Quad. Site Length: 147 m. Site is on a unnamed tributary to Deer Branch that parallels Rt. 623. Site can be accessed at an apartment complex accessed via Commander Rd.

VS014 & VS038 Wolf Creek. Sampled 25 June 1998 and 24 June 1999. Roanoke Quad. Site Length: 150 m. Site begins at a low bridge crossing on private property accessed via Niagara Rd.

VS015 Laymantown Creek. Sampled 29 June 1998. Stewartsville Quad. Site Length: 161 m. Site begins approximately 100 m above the Rt. 658 bridge near the intersection of Rt. 658 with Hwy. 460/221.

VS016 Upper Peters Creek. Sampled 1 July 1998. Salem Quad. Site Length: 157 m. Site begins just upstream of the Northwood Dr. bridge crossing and parallels Meadowbrook Rd.

VS017 Upper Glade Creek. Sampled 2 July 1998. Stewartsville Quad. Site Length: 117 m. Site is located upstream of Rt. 603 and ends approximately 30 m downstream of the confluence of 2 tributaries. It can be accessed at the Village Herb Farm on Rt. 607.

VS018 Middle Mason Creek. Sampled 8 July 1998. Salem Quad. Site Length: 178 m. Site is located approximately 1.5 km upstream from a small dam impounding the creek and can be accessed at a private tree farm along Hwy. 311.

VS019 & VS041 Middle Back Creek. Sampled 9 July 1998 and 9 July 1999. Bent Mountain Quad. Site Length: 178 m. Site begins approximately 20 m below a small, local traffic bridge behind a country store on Hwy. 221 and ends downstream of the confluence with Little Back Creek.

VS020 Lower Peters Creek. Sampled 13 July 1998. Salem Quad. Site Length: 197 m. Site begins upstream of the Shenandoah Ave. Bridge and begins at the southern end of a recreation field along Westside Blvd. nearest Shenandoah Ave.

VS021 Lower Mudlick Creek. Sampled 15 July 1998. Roanoke Quad. Site Length: 209 m. Site ends approximately 20 m downstream of a gas station located over the stream channel at the intersection of Brandon Ave. (Hwy 11) and Edgewood St. Site parallels Edgewood St. and can be accessed at Mudlick Kennels.

VS022 Middle Tinker Creek. Sampled 22 July 1998. Daleville Quad. Site Length: 167 m. Site is located upstream of the Glebe Mills gaging station and can be accessed via Ivy Ln. The site is downstream of a railroad bridge and upstream of a small, local traffic bridge.

VS023 Lower Wilson Creek. Sampled 23 July 1998. Ironto Quad. Site Length: 179 m. Site is located in a cattle pasture approximately 1.5 km above the confluence with Cedar Run and can be accessed via a private road from Rt. 723.

VS024 Middle Glade Creek. Sampled 27 July 1998. Stewartsville Quad. Site Length: 191 m. Site ends approximately 50 meters above a bridge for a private road that can be accessed via Rt. 738.

VS025 & VS045 Lick Fork. Sampled 29 July 1998 and 26 July 1999. Check Quad. Site Length: 243 m. Site is approximately 1 km upstream of the confluence with Goose Creek and begins just above a small tributary entering on stream left. It can be accessed from Rt. 653 via a private road/trail that parallels the stream.

VS026 Upper Goose Creek. Sampled 5 August 1998. Check Quad. Site Length: 160 m. Site begins approximately 40 m above confluence with Lick Fork and just above a private bridge. It can be accessed at the same residence as Lick Fork along Rt. 653.

VS027 Lower Back Creek. Sampled 7 August 1998. Bent Mountain Quad. Site Length: 343 m. Site begins approximately 30 m below the Blue Ridge Parkway bridge and ends at a concrete structure behind a soccer field. It can be best accessed via Merriman Rd.

VS028 Lower Glade Creek. Sampled 10 August 1998. Roanoke Quad. Site Length: 338 m. Site ends approximately 50 m above the Gus Nicks Blvd. bridge and parallels Vale Avenue.

VS029 Lower Goose Creek. Sampled 11 August 1998. Check Quad. Site Length: 320 m. Site begins underneath the Rt. 637 bridge and ends downstream of the large pool below the dam just upstream of the Rt. 653 bridge.

VS030 Lower Tinker Creek. Sampled 13 August 1998. Roanoke Quad. Site Length: 327 m. Site begins at a trailer park that can be accessed via McFarland Dr. off of Hwy. 11. A small tributary within the site comes in on the left side of the stream.

VS031 see VS002

VS032 see VS004

VS033 see VS006

VS034 Brake Branch. Sampled 9 June 1999. Elliston Quad. Site Length: 133 m. Site ends approximately 100 m below the Rt. 631 bridge.

VS035 Upper Bottom Creek. Sampled 11 June 1999. Elliston Quad. Site Length: 133 m. Site begins approximately 150 m above the confluence with Mill Creek and approximately 100 m upstream of the Rt. 637 crossing.

VS036 Upper Smith Creek. Sampled 21 June 1999. Riner Quad. Site Length: 104 m. Site begins approximately 1 km below the confluence of 2 Smith Creek tributaries and is located within a pasture along Rt. 675.

VS037 Upper Elliott Creek. Sampled 22 June 1999. Riner Quad. Site Length: 127 m. Site is located in a cattle pasture and ends approximately 200 meters below the Rt. 679 bridge.

VS038 see VS014

VS039 Lick Run. Sampled 30 June 1999. Roanoke Quad. Site Length: 138 m. Site is located within a small, unnamed park area that is property of the City of Roanoke but is no longer maintained. The site begins approximately 50 m below the back corner of the grassy area and is located east of 10^{th} Street, north of Grayson Ave.

VS040 Middle Smith Creek. Sampled 7 July 1999. Riner Quad. Site Length: 132 m. Site ends just below a small tributary that enters the creek approximately 10 m below the Rt. 615 bridge and can be accessed by a church driveway.

VS041 see VS019

VS042 Bradshaw Creek. Sampled 15 July 1999. Glenvar Quad. Site Length: 156 m. Site begins approximately 50 m above a private bridge on the east side of Rt. 622.

VS043 Middle North Fork. Sampled 19 July 1999. McDonalds Mill Quad. Site Length: 117 m. Site begins approximately 30 m above a private driveway crossing and is located in a cattle pasture along Rt. 785.

VS044 Middle Bottom Creek. Sampled 21 July 1999. Elliston Quad. Site Length: 182 m. Site is approximately 1 km below the confluence of Bottom Creek and Mill Creek. A low driveway with several culverts is within the lower half of the site.

VS045 see VS025

VS046 Lower Mason Creek. Sampled 3 August 1999. Salem Quad. Site Length: 280 m. Site is located west of East Salem Elementary School and begins approximately 100 m upstream of the Roanoke Blvd. bridge.

VS047 Lower North Fork. Sampled 5 August 1999. McDonalds Mill Quad. Site Length: 232 m. Site is located along Rt. 785, ends approximately 50 m below the confluence with Mill Creek and approximately 30 m below a small culvert bridge in a cattle pasture.

VS048 Lower Bottom Creek. Sampled 9 August 1999. Check Quad. Site Length: 244 m. Site is located along Rt. 637, begins approximately 50 m above a low culvert bridge for a private driveway, and ends just below a wide, 2-3 m deep pool.

VS049 Lower Elliott Creek. Sampled 11 August 1999. Pilot Quad. Site Length: 353 m. Site begins approximately 50 m above the confluence with Falling Branch and can be accessed along Split Rail Road which crosses Falling Branch.

VS050 see VS012

Appendix B.-Habitat variables for all 43 sites. Substrate variables are described using size codes and corresponding labels.

Site	Site Length (m)	Mean Width (m)	Canopy Closure (%)	D50	D50	D90	D90	Riffle D50	Riffle D50	Pool D50	Pool D50	St. Dev. Substrate	Prop. of Silt and Sand
UTWORC	141	4.1	0.61	9	very coarse gravel	13	large cobble	10	very coarse gravel	10	very coarse gravel	3.4	0.050
Dry Branch	123	3.7	0.76	10	very coarse gravel	13	large cobble	10	very coarse gravel	9	very coarse gravel	3.9	0.080
Horners Branch	120	3.5	0.93	4	fine gravel	7	coarse gravel	5	medium gravel	3	very fine gravel	1.9	0.270
Purgatory Creek	167	5.3	0.87	7	coarse gravel	13	large cobble	10	very coarse gravel	3	very fine gravel	4.2	0.210
Falling Branch	162	6.5	0.31	7	coarse gravel	11	small cobble	9	very coarse gravel	3	very fine gravel	3.6	0.130
Mill Creek	154	3.9	0.02	9	very coarse gravel	13	large cobble	9	very coarse gravel	11	small cobble	3.8	0.040
Upper Back Creek	146	4.5	0.68	8	coarse gravel	13	large cobble	10	very coarse gravel	3	very fine gravel	4.6	0.210
Upper Mason Creek	188	5.9	0.68	10	very coarse gravel	12	medium cobble	10	very coarse gravel	10	very coarse gravel	2.9	0.005
Upper Mudlick Creek	143	4.4	0.64	3	very fine gravel	6	medium gravel	4	fine gravel	2	sand	1.6	0.310
Upper Tinker Creek	157	4.1	0.40	5	medium gravel	12	medium cobble	8	coarse gravel	3	very fine gravel	4.1	0.360
Buffalo Creek	170	4.8	0.60	4	fine gravel	10	very coarse gravel	6	medium gravel	3	very fine gravel	3.5	0.255
Upper Wilson Creek	143	4.2	0.85	6	medium gravel	11	small cobble	7	coarse gravel	4	fine gravel	3.3	0.270
Deer Branch tributary	147	4.1	0.82	5	medium gravel	12	medium cobble	10	very coarse gravel	4	fine gravel	4.1	0.286
Wolf Creek	150	4.3	0.82	3	very fine gravel	14	very large cobble	10	very coarse gravel	2	sand	5.1	0.301
Laymantown Creek	161	4.6	0.78	6	medium gravel	13	large cobble	7	coarse gravel	5	medium gravel	4.6	0.316
Upper Peters Creek	157	3.1	0.33	7	coarse gravel	12	medium cobble	7	coarse gravel	7	coarse gravel	3.2	0.331
Upper Glade Creek	117	3.1	0.28	6	medium gravel	13	large cobble	10	very coarse gravel	2	sand	4.4	0.346
Brake Branch	133	3.4	0.78	10	very coarse gravel	13	large cobble	11	small cobble	5	medium gravel	4.3	0.07
Upper Bottom Creek	133	3.6	0.37	3	very fine gravel	10	very coarse gravel	7	coarse gravel	2	sand	4.1	0.48
Upper Smith Creek	104	3.2	0.32	3	very fine gravel	7	coarse gravel	5	medium gravel	3	very fine gravel	2.8	0.34
Upper Elliott Creek	127	2.8	0.37	3	very fine gravel	11	small cobble	6	medium gravel	3	very fine gravel	3.7	0.40
Lick Run	138	3.8	0.68	3	very fine gravel	11	small cobble	4	fine gravel	2	sand	3.5	0.38

Site	Site Length (m)	Mean Width (m)	Canopy Closure (%)	D50	D50	D90	D90	Riffle D50	Riffle D50	Pool D50	Pool D50	St. Dev. Substrate	Prop. of Silt and Sand
Middle Mason Creek	178	4.3	0.48	9	very coarse gravel	12	medium cobble	10	very coarse gravel	11	small cobble	3.2	0.361
Middle Back Creek	178	5.3	0.62	7	coarse gravel	12	medium cobble	8	coarse gravel	2	sand	3.9	0.377
Lowers Peters Creek	197	5.3	0.48	6	medium gravel	12	medium cobble	12	medium cobble	6	medium gravel	3.9	0.16
Lower Mudlick Creek	209	5.5	0.59	3	very fine gravel	13	large cobble	10	very coarse gravel	3	very fine gravel	4.4	0.24
Middle Tinker Creek	167	4.5	0.69	2	sand	7	coarse gravel	NA	NA - no riffles	2	sand	2.8	0.50
Lower Wilson Creek	179	5.1	0.33	5	medium gravel	10	very coarse gravel	7	coarse gravel	4	fine gravel	3.8	0.21
Middle Glade Creek	191	5.0	0.75	4	fine gravel	10	very coarse gravel	7	coarse gravel	2	sand	3.5	0.31
Lick Fork	243	6.4	0.81	9	very coarse gravel	15	small boulder	10	very coarse gravel	3	very fine gravel	5.0	0.21
Middle Goose Creek	160	3.8	0.7	9	very coarse gravel	14	very large cobble	10	very coarse gravel	8	coarse gravel	4.3	0.11
Middle Smith Creek	132	3.6	0.46	5	medium gravel	13	large cobble	9	very coarse gravel	3	very fine gravel	4.7	0.27
Bradshaw Creek	156	4.9	0.74	9	very coarse gravel	11	small cobble	9	very coarse gravel	9	very coarse gravel	2.5	0.05
Middle North Fork	117	3.2	0.29	9	very coarse gravel	13	large cobble	11	small cobble	8	coarse gravel	4.2	0.11
Middle Bottom Creek	182	6.0	0.67	11	small cobble	15	small boulder	12	medium cobble	11	small cobble	5.4	0.19
Lower Back Creek	342	8.9	0.51	3	very fine gravel	13	large cobble	7	coarse gravel	2	sand	4.8	0.44
Lower Glade Creek	338	8.6	0.57	6	medium gravel	12	medium cobble	9	very coarse gravel	3	very fine gravel	3.6	0.12
Lower Goose Creek	320	8.6	0.4	9	very coarse gravel	12	medium cobble	9	very coarse gravel	1	silt	3.7	0.11
Lower Tinker Creek	327	11.0	0.82	7	coarse gravel	12	medium cobble	7	coarse gravel	7	coarse gravel	3.5	0.09
Lower Mason Creek	280	7.0	0.48	10	very coarse gravel	14	very large cobble	10	very coarse gravel	10	very coarse gravel	4.1	0.09
Lower North Fork	232	6.4	0.52	8	coarse gravel	12	medium cobble	10	very coarse gravel	3	very fine gravel	4.4	0.24
Lower Bottom Creek	244	7.0	0.65	11	small cobble	14	very large cobble	9	very coarse gravel	12	medium cobble	4.6	0.03
Lower Elliott Creek	353	9.2	0.67	4	fine gravel	12	medium cobble	11	small cobble	NA	NA - no pools	4.1	0.22

Appendix B. Habitat variables for all 43 sites. Substrate variables are described using size codes and corresponding labels.

Site	Median Depth (cm)	St. Dev. Depth	Max. Obs. Depth (cm)	Proportion Riffles	Proportion Pools	Proportion Bedrock	Discharge m^3/sec	Prop. Woody Vegetation	Prop. Impervious Surfaces	Prop. Pasture	Prop Weed/ grass/yard
UTWORC	5	9.5	45	0.23	0.22	0.01	0.014	0.59	0	0	0.41
Dry Branch	6	8.5	38	0.29	0.14	0.05	0.008	0.54	0	0	0.45
Horners Branch	12	12.3	51	0.17	0.21	0.04	0.048	0.80	0	0	0.2
Purgatory Creek	10	18.9	92	0.35	0.26	0.01	0.116	0.87	0	0	0.14
Falling Branch	14	10.3	57	0.18	0.12	0.37	0.164	0.79	0	0	0.21
Mill Creek	19	11.6	65	0.20	0.13	0.10	0.263	0.01	0.02	0.97	0
Upper Back Creek	13	7.3	31	0.27	0.03	0.03	0.144	0.29	0.04	0	0.51
Upper Mason Creek	6	10.6	61	0.32	0.27	0.02	0.040	0.79	0	0	0.31
Upper Mudlick Creek	12	8.5	35	0.09	0.06	0.00	0.136	0.12	0.13	0	0.75
Upper Tinker Creek	19	10.3	40	0.31	0.45	0.07	0.105	0.02	0	0.98	0
Buffalo Creek	12	13.6	69	0.17	0.49	0.01	0.093	0.76	0	0	0.24
Upper Wilson Creek	13	10.4	59	0.24	0.20	0.23	0.127	0.59	0	0.28	0.14
Deer Branch tributary	16	17.1	79	0.09	0.47	0.17	0.020	0.49	0.12	0	0.39
Wolf Creek	11	13.1	65	0.34	0.42	0.36	0.017	0.84	0.05	0	0.12
Laymantown Creek	13	9.9	45	0.19	0.21	0.04	0.139	0.87	0	0	0.13
Upper Peters Creek	10	10.4	38	0.27	0.48	0.17	0.011	0.03	0.12	0	0.86
Upper Glade Creek	9	9.4	45	0.28	0.19	0.00	0.017	0.04	0.01	0	0.74
Brake Branch	4	6.7	47	0.42	0.16	0.12	0.014	0.99	0	0	0.01
Upper Bottom Creek	11	10.1	51	0.13	0.28	0	0.037	0.4	0	0	0.6
Upper Smith Creek	10	10.2	56	0.22	0.22	0.01	0.074	0.14	0	0	0.87
Upper Elliott Creek	15	8.8	35	0.21	0.13	0.05	0.042	0.04	0	0.96	0
Lick Run	13	10.4	44	0.26	0.23	0.02	0.062	0.34	0	0	0.55

Site	Median Depth (cm)	St. Dev. Depth	Max. Obs. Depth (cm)	Proportion Riffles	Proportion Pools	Proportion Bedrock	Discharge m^3/sec	Prop. Woody Vegetation	Prop. Impervious Surfaces	Prop. Pasture	Prop Weed/ grass/yard
Middle Mason Creek	8	12.9	55	0.11	0.39	0.20	0.006	0.39	0	0	0.62
Middle Back Creek	13	12.3	52	0.26	0.19	0.00	0.125	0.39	0.08	0	0.5
Lowers Peters Creek	13	13.6	53	0.10	0.56	0.05	0.054	0.53	0	0	0.47
Lower Mudlick Creek	16	11.1	50	0.26	0.15	0.11	0.133	0.08	0.26	0	0.65
Middle Tinker Creek	22	11.5	58	0.00	0.46	0.03	0.096	0.48	0.14	0	0.38
Lower Wilson Creek	13	12.6	81	0.35	0.30	0.03	0.130	0.09	0	0.91	0
Middle Glade Creek	10	10.6	48	0.32	0.10	0.05	0.076	0.08	0.04	0.37	0.42
Lick Fork	9	10.0	47	0.23	0.16	0.05	0.119	0.8	0.05	0	0.07
Middle Goose Creek	11	7.2	37	0.33	0.15	0.10	0.045	0.33	0.28	0	0.4
Middle Smith Creek	10	6.1	26	0.4	0.06	0.14	0.076	0.09	0.14	0	0.78
Bradshaw Creek	28	22	97	0.13	0.85	0.09	0.014	0.78	0	0	0.23
Middle North Fork	10	9.3	39	0.3	0.31	0.06	0.023	0.07	0.02	0.91	0.00
Middle Bottom Creek	6	11.1	54	0.45	0.16	0.02	0.051	0.2	0.28	0	0.53
Lower Back Creek	14	11.8	56	0.21	0.38	0.08	0.113	0.6	0.01	0	0.37
Lower Glade Creek	22	13.4	66	0.14	0.10	0.01	0.521	0.24	0.38	0	0.37
Lower Goose Creek	12	8.7	45	0.35	0.08	0.17	0.181	0.39	0.22	0	0.38
Lower Tinker Creek	22	21.1	80	0.18	0.37	0.25	0.161	0.58	0.05	0	0.32
Lower Mason Creek	11	11.8	59	0.15	0.3	0.04	0.014	0.49	0	0	0.51
Lower North Fork	17	17	69	0.17	0.33	0.04	0.122	0.25	0	0.76	0
Lower Bottom Creek	9	10.5	50	0.28	0.12	0.44	0.045	0.4	0.21	0.12	0.27
Lower Elliott Creek	11	8	40	0.4	0	0.25	0.195	0.77	0	0	0.23

Appendix C.-Watershed level variables for all sites separated by watershed size class.

Site No.	Site	Watershed Area (km^2)	Elevation (m)	Watershed Land Use Forest	Watershed Land Use Herb/ag	Watershed Land Use Disturbed	60 %, 60 m Buffer Land Use Forest	60 %, 60 m Buffer Land Use Herb/ag	60 %, 60 m Buffer Land Use Disturbed
	<u>Small Sites</u>								
VS001	UTWORC	10.5	318	85.0%	6.3%	8.8%	38.7%	23.8%	37.5%
VS002	Dry Branch	12.4	338	96.1%	3.7%	0.2%	87.5%	10.3%	2.1%
VS003	Horners Branch	13.7	320	81.6%	4.7%	13.6%	42.5%	10.8%	46.7%
VS004	Purgatory Creek	14.1	459	87.7%	11.7%	0.7%	93.8%	6.2%	0.0%
VS005	Falling Branch	12.7	474	70.2%	17.3%	12.4%	97.1%	2.9%	0.0%
VS006	Mill Creek	15.9	476	82.6%	17.3%	0.1%	91.6%	8.4%	0.0%
VS007	Upper Back Creek	11.0	451	91.0%	8.8%	0.1%	83.0%	16.9%	0.1%
VS008	Upper Mason Creek	19.3	416	96.0%	3.9%	0.1%	98.8%	0.7%	0.5%
VS009	Upper Mudlick Creek	13.9	317	45.6%	12.2%	42.2%	44.9%	7.7%	47.4%
VS010	Upper Tinker Creek	16.0	382	46.1%	50.5%	3.5%	19.3%	76.1%	4.6%
VS011	Buffalo Creek	14.2	358	46.6%	36.4%	17.0%	7.8%	67.8%	24.5%
VS012	Upper Wilson Creek	11.0	482	46.5%	33.0%	20.5%	62.3%	13.5%	24.2%
VS013	Deer Branch tributary	10.8	316	57.6%	13.4%	29.0%	19.6%	28.4%	52.1%
VS014	Wolf Creek	10.9	281	29.9%	25.8%	44.3%	30.6%	37.0%	32.5%
VS015	Laymantown Creek	11.9	325	61.1%	24.8%	14.0%	38.2%	38.3%	23.5%
VS016	Upper Peters Creek	9.9	320	69.0%	14.5%	16.5%	31.3%	32.6%	36.0%
VS017	Upper Glade Creek	10.7	343	77.5%	14.9%	7.6%	62.0%	28.1%	9.9%
VS034	Brake Branch	14.3	428	98.4%	1.3%	0.4%	93.5%	4.2%	2.3%
VS035	Upper Bottom Creek	15.4	766	88.9%	10.7%	0.4%	85.9%	13.5%	0.6%
VS036	Upper Smith Creek	15.7	591	34.7%	61.4%	3.9%	39.6%	57.4%	3.0%
VS037	Upper Elliott Creek	14.4	585	55.8%	42.7%	1.5%	23.1%	75.4%	1.6%
VS039	Lick Run	11.8	303	4.4%	32.1%	63.5%	15.0%	15.4%	69.6%

Site No.	Site	Watershed Area (km^2)	Elevation (m)	Watershed Land Use Forest	Watershed Land Use Herb/ag	Watershed Land Use Disturbed	60 %, 60 m Buffer Land Use Forest	60 %, 60 m Buffer Land Use Herb/ag	60 %, 60 m Buffer Land Use Disturbed
	Medium Sites								
VS018	Middle Mason Creek	36.0	376	93.2%	6.4%	0.4%	83.2%	14.9%	2.0%
VS019	Middle Back Creek	29.4	397	89.4%	9.4%	1.2%	73.4%	23.6%	3.1%
VS020	Lower Peters Creek	22.0	302	41.1%	16.1%	42.8%	16.4%	6.5%	77.1%
VS021	Lower Mudlick Creek	24.7	296	32.9%	9.7%	57.4%	35.7%	6.9%	57.4%
VS022	Middle Tinker Creek	30.3	377	27.6%	65.5%	6.8%	12.4%	80.9%	6.7%
VS023	Lower Wilson Creek	24.9	456	69.1%	20.7%	10.1%	82.2%	10.9%	6.9%
VS024	Middle Glade Creek	30.1	330	64.2%	23.5%	12.3%	18.5%	51.1%	30.4%
VS025	Lick Fork	31.4	548	87.2%	12.5%	0.3%	96.3%	3.7%	0.0%
VS026	Middle Goose Creek	18.5	523	90.1%	9.4%	0.5%	92.4%	6.7%	0.8%
VS040	Middle Smith Creek	22.3	570	47.8%	49.1%	3.1%	56.1%	40.7%	3.2%
VS042	Bradshaw Creek	42.2	401	91.5%	6.0%	2.5%	86.2%	8.4%	5.5%
VS043	Middle North Fork	26.9	532	62.4%	37.5%	0.1%	36.1%	63.4%	0.5%
VS044	Middle Bottom Creek	33.4	763	73.8%	25.3%	0.9%	85.8%	13.5%	0.7%
	Large Sites								
VS027	Lower Back Creek	69.2	347	83.5%	10.4%	6.1%	80.1%	12.4%	7.5%
VS028	Lower Glade Creek	82.2	276	50.4%	30.5%	19.1%	15.6%	31.2%	53.3%
VS029	Lower Goose Creek	59.7	482	89.1%	10.6%	0.3%	96.1%	3.8%	0.2%
VS030	Lower Tinker Creek	72.5	341	36.9%	48.0%	15.1%	22.4%	46.0%	31.6%
VS046	Lower Mason Creek	75.2	302	86.3%	6.2%	7.5%	68.5%	4.4%	27.1%
VS047	Lower North Fork	83.9	470	73.6%	26.3%	0.1%	67.5%	32.5%	0.0%
VS048	Lower Bottom Creek	71.1	479	85.2%	14.3%	0.5%	93.4%	6.5%	0.0%
VS049	Lower Elliott Creek	71.1	471	72.0%	26.7%	1.3%	93.5%	6.0%	0.4%

Appendix D.-Number of fish of each species, ordered by family, collected at each site during 1998. U = Upper, M = Middle, L = Lower. Sites are ordered by sample date.

Species	VS001 UTWORC 21-May	VS002 Dry Branch 26-May	VS003 Horners Branch. 28-May	VS004 Purgatory Creek 1-Jun	VS005 Falling Branch. 2-Jun	VS006 Mill Creek. 4-Jun	VS007 Back Creek (U) 9-Jun	VS008 Mason Creek (U) 10-Jun	VS009 Mudlick Creek (U) 11-Jun	VS010 Tinker Creek (U) 16-Jun
rainbow trout						4				
brown trout						7				
brook trout						1				
chain pickerel								1		
central stoneroller	6		2	1	12	8		105	2	119
rosyside dace	5		8	20	39	66		101		65
common carp										
cutlips minnow				1						
white shiner				16	1	1				
crescent shiner			1	10	1	1	8	1	19	233
rosefin shiner										2
bluehead chub	8	6	14	41	45	34	94	27	105	145
spottail shiner										
swallowtail shiner										
mimic shiner										
mtn. redbelly dace	37	1	3	39	287	11	130	473	272	290
bluntnose minnow										
fathead minnow										
blacknose dace	31	11	13	19	57	11	38	67	110	4
longnose dace										
creek chub			4					12		
white sucker		10	14	12	4	4		2	6	34
nothern hogsucker						1				
Roanoke hogsucker				2	1					
silver redhorse										
golden redhorse										
v-lip redhorse										
bigeye jumprock										
black jumprock		1				2				
torrent sucker	1			10	22	8			16	
yellow bullhead										
brown bullhead										
orangefin madtom										
margined madtom				7						11
eastern mosquitofish										
mottled sculpin		2		60	67	495				
rock bass										8
redbreast sunfish										12
green sunfish										
pumpkinseed										1
bluegill sunfish					1					38
smallmouth bass										
spotted bass										
largemouth bass										
fantail darter	100	59	19	55	282	118	491	130	54	446
johnny darter										
riverweed darter		20								81
Roanoke logperch										
Roanoke darter										
TOTAL	188	110	78	293	819	772	761	919	584	1489

Appendix D.-Number of fish of each species, ordered by family, collected at each site during 1998. U = Upper, M = Middle, L = Lower. Sites are ordered by sample date.

Species	VS011 Buffalo Creek 17-Jun	VS012 Wilson Creek (U) 18-Jun	VS013 Deer Br. Trib 24-Jun	VS014 Wolf Creek 25-Jun	VS015 Laymantown. Creek 29-Jun	VS016 Peters Creek (U) 1-Jul	VS017 Glade Creek (U) 2-Jul	VS018 Mason Creek (M) 8-Jul	VS019 Back Creek (M) 9-Jul	VS020 Peters Creek (L) 13-Jul
rainbow trout										
brown trout					2				4	
brook trout										
chain pickerel								2		
central stoneroller	42	8	3	14	21	29	115	167	12	39
rosyside dace	14	99	6					90		3
common carp										
cutlips minnow										
white shiner				2						3
crescent shiner	20	2	2	3	2	19	36	43	127	254
rosefin shiner										
bluehead chub	47	51	30	42	64	113	140	24	193	268
spottail shiner										
swallowtail shiner								1		
mimic shiner										
mtn. redbelly dace	367	70	2	145	112	256	343	70	420	541
bluntnose minnow	5			2		42		10		131
fathead minnow										
blacknose dace	669	5	4	68	36	6	244		53	59
longnose dace					3					
creek chub				1						
white sucker	38	6	2	41	7	12	18	15	51	55
nothern hogsucker		1								
Roanoke hogsucker									18	
silver redhorse										
golden redhorse		1								
v-lip redhorse										
bigeye jumprock										
black jumprock									8	
torrent sucker		4	2		136		232			12
yellow bullhead										
brown bullhead										
orangefin madtom										
margined madtom			1					36	23	
eastern mosquitofish										
mottled sculpin		4								
rock bass			2							12
redbreast sunfish		1	11		2	65		18		39
green sunfish										
pumpkinseed										
bluegill sunfish	6		5			1	1	3		
smallmouth bass					1					
spotted bass										
largemouth bass	1		1				4	4		
fantail darter	122	132	67	77	479	204	791	206	272	629
johnny darter			4			2		20		3
riverweed darter	3									
Roanoke logperch										
Roanoke darter										
TOTAL	1334	384	142	395	865	749	1924	709	1181	2048

Appendix D.-Number of fish of each species, ordered by family, collected at each site during 1998. U = Upper, M = Middle, L = Lower. Sites are ordered by sample date.

Species	VS021 Mudlick Creek (L) 15-Jul	VS022 Tinker Creek (M) 22-Jul	VS023 Wilson Creek (L) 23-Jul	VS024 Glade Creek (M) 27-Jul	VS025 Lick Fork 29-Jul	VS026 Goose Creek (M) 5-Aug	VS027 Back Creek (L) 7-Aug	VS028 Glade Creek (L) 10-Aug	VS029 Goose Creek (L) 11-Aug	VS030 Tinker Creek (L) 13-Aug
rainbow trout	1				9		1		2	
brown trout				1	8		1	1	4	
brook trout								1		
chain pickerel										
central stoneroller	116	10	50	133	2	19	248	205	258	13
rosyside dace	1	16	121		156	155	1	1	521	9
common carp		2								
cutlips minnow									11	
white shiner	1						1	129	22	
crescent shiner	19	25	51	35	7	11	739	188	128	53
rosefin shiner							47	12		4
bluehead chub	235	33	188	127	120	136	612	184	250	111
spottail shiner								185		
swallowtail shiner							71	2	3	
mimic shiner									1	
mtn. redbelly dace	687	36	702	888	83	245	328	441	779	39
bluntnose minnow	1							121		23
fathead minnow						4				
blacknose dace	174	1	5	195	14	26	18	28	69	7
longnose dace								19		
creek chub									1	
white sucker	51	35	6	90	14	1	79	102	49	71
nothern hogsucker							7	14	3	2
Roanoke hogsucker			3		1		45		61	
silver redhorse								7		
golden redhorse							12			
v-lip redhorse							5			
bigeye jumprock								2	4	
black jumprock							82	25	38	
torrent sucker	10		66	251	48	53		186	469	2
yellow bullhead							2			
brown bullhead										
orangefin madtom									17	
margined madtom	3	11	8		67	6	89	26	88	6
eastern mosquitofish										
mottled sculpin			8		106	156			227	
rock bass	1							23	3	11
redbreast sunfish	1	11					67	18		13
green sunfish								4		
pumpkinseed										
bluegill sunfish		5						2		1
smallmouth bass							9	4	7	
spotted bass										
largemouth bass										
fantail darter	675	27	826	1498	88	166	372	557	490	346
johnny darter							26	3		
riverweed darter	10	18	2				50	45	41	20
Roanoke logperch								1		
Roanoke darter							2	12	20	
TOTAL	1986	230	2036	3218	723	978	2914	2548	3566	731

Appendix E.-Number of fish of each species, collected at each site during 1999. U = Upper, M = Middle, L = Lower. Sites also sampled in 1998 are indicated by (II).

Species	VS031 Dry Branch (II) 2-Jun	VS032 Purgatory Creek (II) 3-Jun	VS033 Mill Creek (II) 7-Jun	VS034 Brake Branch 9-Jun	VS035 Bottom Creek (U) 11-Jun	VS036 Smith Creek (U) 21-Jun	VS037 Elliott Creek (U) 22-Jun	VS038 Wolf Creek (II) 24-Jun	VS039 Lick Run 30-Jun	VS040 Smith Creek (M) 7-Jul
rainbow trout			8	1						1
brown trout			6							
brook trout										
chain pickerel										
central stoneroller		4	9	6	17	3	1	17		7
rosyside dace		32	151	7	205			4		
common carp										
cutlips minnow										
white shiner		5	6							
crescent shiner		20	5		66	5		4	29	14
rosefin shiner										
bluehead chub		27	31	8	114	36	20	31	77	56
spottail shiner										
swallowtail shiner										
mimic shiner										
mtn. redbelly dace		270	25	104	454	328	17	248	128	709
bluntnose minnow										
fathead minnow										
blacknose dace		67	6	103	13	59	63	112		76
longnose dace										
creek chub					2			9		
white sucker		4		2	18	2	11	24	12	1
nothern hogsucker			1							
Roanoke hogsucker					17	2				10
silver redhorse										
golden redhorse										
v-lip redhorse										
bigeye jumprock										
black jumprock			2							
torrent sucker		19	41	102		38	3			189
yellow bullhead										
brown bullhead										
orangefin madtom										
margined madtom		2				1				1
eastern mosquitofish									2	
mottled sculpin		2	355	294		80	153			346
rock bass										
redbreast sunfish										
green sunfish							3			
pumpkinseed										
bluegill sunfish										
smallmouth bass										
spotted bass				1						
largemouth bass										
fantail darter		62	128	160	616	207	83	148	208	277
johnny darter										
riverweed darter			1							
Roanoke logperch										
Roanoke darter										
TOTAL	0	514	775	788	1522	761	354	597	456	1687

Appendix E.-Number of fish of each species, collected at each site during 1999. U = Upper, M = Middle, L = Lower. Sites also sampled in 1998 are indicated by (II).

Species	VS041 Back Creek (M, II) 9-Jul	VS042 Bradshaw Creek 15-Jul	VS043 North Fork (M) 19-Jul	VS044 Bottom Creek (M) 21-Jul	VS045 Lick Fork (II) 26-Jul	VS046 Mason Creek (L) 3-Aug	VS047 North Fork (L) 5-Aug	VS048 Bottom Creek (L) 9-Aug	VS049 Elliott Creek (L) 11-Aug	VS050 Wilson Creek (U, II) 16-Sep
rainbow trout		1		1	1		2	1		
brown trout					6		16			
brook trout										
chain pickerel		5								
central stoneroller	39	2	63	6		155	46	73	133	10
rosyside dace	3	50	49	465	152	1	284	84	177	155
common carp										
cutlips minnow			2			26		15	40	
white shiner			27			246	115	76	10	
crescent shiner	169	4	95	246	8	68	78	61	97	1
rosefin shiner			6	5						
bluehead chub	175	14	266	128	81	144	252	80	537	70
spottail shiner										
swallowtail shiner										
mimic shiner										
mtn. redbelly dace	758	40	219	312	143		353	31	870	163
bluntnose minnow						149	7			
fathead minnow										
blacknose dace	83			18	21		1		146	1
longnose dace										
creek chub	1			4	2					
white sucker	90	2	1	18	12	12	33	9	25	
nothern hogsucker			1	6			1		1	
Roanoke hogsucker	25		7	13		1	13	16	29	
silver redhorse							3			
golden redhorse			7				12		1	
v-lip redhorse										
bigeye jumprock								5		
black jumprock	7		14			23	32	31	57	
torrent sucker			118		19		89	146	273	3
yellow bullhead										
brown bullhead				6						
orangefin madtom								1	4	
margined madtom	22	3	26	24	56		19	28	93	
eastern mosquitofish										
mottled sculpin					108		176	13	124	5
rock bass		8	7			37	5	3		
redbreast sunfish		7	9			64	4	2		
green sunfish				2			1			4
pumpkinseed				10						
bluegill sunfish				2						
smallmouth bass		1	5			10	7	9		
spotted bass										
largemouth bass	1								1	
fantail darter	162	17	434	216	80	153	254	39	599	70
johnny darter						13				
riverweed darter			20			27	46	17	1	
Roanoke logperch										
Roanoke darter			1			3	10	12	1	
TOTAL	1535	154	1377	1482	689	1132	1859	752	3219	482

VITA

Vann Franklin Stancil was born on 6 February 1973 to Donell and Lou Stancil. He grew up on a family farm in the Glendale community near the town of Kenly in eastern North Carolina. His family grew tobacco, corn, soybeans, and wheat and he spent many summer hours sitting on a tractor. His aunt Virginia Fulghum instilled in him a love of fishing at an early age. The family pond beside his house and his fascination with the fish in it were probably the primary reasons he pursued a career in fisheries.

After two years of career uncertainty at North Carolina State University, he met with Rich Noble of the fisheries and wildlife program and started taking courses toward a BS in Fisheries and Wildlife Sciences. He was also involved with the Jefferson Scholars Program at N.C. State; this led him to also pursue a BA in environmental studies. Five years and 186 credit hours later, he graduated from N. C. State with two degrees. He was fortunate to work on several research projects in a variety of settings such as streams in Trinidad, B. E. Jordan Reservoir in central North Carolina, brook trout streams in Shenandoah National Park, reservoirs in Puerto Rico, and coastal plain rivers of the Chowan River watershed in northeastern North Carolina.

In August 1997 he married the former Amy Jackson, also of Kenly, and they moved to Blacksburg, Virginia shortly thereafter. Vann began graduate school in the Fisheries and Wildlife Sciences Department at Virginia Tech under the tutelage of Don Orth. After nearly three years, over 54,000 stream fishes, and many enjoyable canoe trips on the New River, Vann and Amy returned to North Carolina. This master's thesis detailing land use effects on stream fishes of the upper Roanoke River watershed is the culmination of his work at Virginia Tech.