

PREDICTING TROUT POPULATION AND FISH ASSEMBLAGE RESPONSES TO
REDUCED RIPARIAN BUFFER WIDTHS IN NORTHERN GEORGIA, USA

by

KRISTA LEE JONES

(Under the Direction of Judy L. Meyer)

ABSTRACT

The Georgia State Legislature reduced the mandatory riparian buffer width along Georgia's trout streams from 30 to 15 m. This research evaluated the responses of trout populations and fish assemblages to this buffer width reduction. With a 15-m buffer, stream temperatures will increase by 1.6 to 2.3°C and fine sediment in riffles will increase by 11%, resulting in an over 80% reduction in the biomass of young-of-the-year trout. With a 15-m buffer, 91% of the trout stream network would be too warm ($> 21.5^{\circ}\text{C}$) to support reproducing trout populations. As stream temperatures increase, range-restricted and benthic spawning fishes will decline. Substantial warming of trout streams has occurred along the trout stream network because of existing riparian deforestation. Further deforestation of riparian areas, which is allowed with the 15-m buffer width, will contribute to the thermal warming of trout streams and, thereby, jeopardize their capacity to support trout and native Southeastern fishes.

INDEX WORDS: mountain streams, stream temperature, sediment, riparian buffer widths, forest cover, trout, stream fishes, Southern Appalachian Mountains

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CHAPTER 1

INTRODUCTION

The ecoregion of the Southern Appalachian Mountains has increasing human population and suburban development pressures as land use in the region shifts from agriculture to suburban development (Wear and Bolstad 1998). In particular, these pressures are growing in Northeast Georgia, which encompasses the southern terminus of the Southern Appalachian Mountains. Georgia is the 6th fastest growing state in the United States, and populations in Georgia's Southern Appalachian counties increased 23.8 – 75.5% between 1990 and 2000 (U.S. Census Bureau 2000). Because of the growing human population and associated development pressures, Northeast Georgia faces the challenge of balancing the development of private lands with the protection of natural resources.

Mountain streams are important natural resources in Northeast Georgia. These streams are part of the larger Southeastern ecoregion, which supports the highest diversity and endemism of temperate freshwater fishes in the world and the southernmost naturally reproducing trout populations in the eastern United States (Warren et al. 2000; Benhke 2002). Recreational and economic values are attached to trout fishing in Georgia's mountain streams. Expenditures associated with Georgia trout fishing (i.e., sporting goods, bait, guide services, and travel) are estimated to exceed \$172 million annually (DeMeo et al. 2004). Also, Georgia's trout fishery generates annually more than \$3.4 million in state sales tax, \$2 million in state income tax, and \$4.8 million in federal income tax (Mahari 1996).

In 2001, the Georgia State Legislature reduced the width of mandatory-forested riparian buffers along the State's mountain trout streams from 30 to 15 m. The buffer width was reduced to increase the amount of land available for development. The new 15-m buffer, however, is narrower than the average mandatory buffer width in the Southeastern ecoregion, United States, and Canada (Lee et al. 2004). Potential consequences of reducing the buffer width include elevated stream temperatures and increased delivery of fine sediment to streams (Gregory et al. 1991; Waters 1995; Scott et al. 2002; Pusey and Arthington 2003). Warmer stream temperatures and increased sedimentation can adversely affect the metabolism, growth, and reproductive success of trout (Hausle and Coble 1976; Crisp 2000; Curry et al. 2004; Suttle et al. 2004) and also populations of Southern Appalachian fishes that are generally sediment-intolerant and prefer cool water (e.g., Scott and Helfman 2001).

Because of the potential for elevated stream temperatures and sedimentation with the 15-m buffer, the Georgia State Legislature commissioned this three-year study to evaluate the expected response of the State's trout populations to reduced riparian buffer widths. In addition to providing this information, we also assessed how the entire fish assemblage in these streams may be impacted by the reduced buffer width. The overall goal of this research was to provide the State of Georgia with rigorous scientific information for use in policy and management decisions relating to the State's mountain trout streams.

Research conceptual framework

Prior to this research, quantitative relationships between riparian forest cover and instream habitat conditions had not been determined for streams across Northeast Georgia.

A conceptual model reflecting the current understanding of the linkages between riparian forest and fishes was the foundation for our assessment of the potential impacts of reduced riparian buffer widths on trout populations and fish assemblages within the mountain streams of Northeast Georgia (Figure 1.1).

Forested riparian ecosystems along the stream network are important transition zones from terrestrial to aquatic ecosystems that influence stream temperature dynamics, water quality, sediment regimes, foodweb resources, and habitat heterogeneity (Gregory et al. 1991; Correll 1997; Wallace et. al 1997; Pusey and Arthington 2003). Alterations of the landscape by different landuses have been related to increases in stream temperature and sediment delivery and to decreases in water quality, basal stream food resources, and habitat diversity (Waters 1995; Jones et al. 1999; Scott et al. 2002; England 2003). Since forested riparian buffers can potentially mitigate the impacts of various landuses on stream ecosystems, riparian buffers are common management tools for protecting aquatic systems and fauna. Conversely, deforestation of riparian areas can lead to elevated stream temperatures and sedimentation despite high levels of forest cover elsewhere in the stream's basin. Recommendations for buffer widths that protect aquatic habitat range from 10 m to 90 m and vary based on topography, stream type and size, and presence of fish (FEMAT 1993; Wenger 1999; Lee et al. 2004).

Trout and native Southern Appalachian fishes are apt to be adversely impacted by increasing stream temperatures and fine sediment loading. Trout are cold-water fishes that require coarse bed sediments for spawning (Behnke 2002). Relatively small increases in stream temperature can have important consequences for the rates of metabolism, growth, and development and also swimming speed of trout (Crisp 2000). Barton et al. (1985) found that Southern Ontario streams below 22°C (in trimean weekly maximum temperature) supported

brook trout populations while streams above 22°C were too warm to support trout. The Georgia Department of Natural Resources (DNR) uses a similar temperature criterion to distinguish streams likely and unlikely to support self-sustaining trout populations; Georgia trout streams with maximum weekly average temperatures below 72°F (22.2°C) are classified as primary trout streams where self-sustaining trout populations likely exist while streams with temperatures above 72°F (22.2°C) are classified as secondary trout streams where viable trout populations likely do not exist (although trout may be stocked for anglers; England 1998; Georgia Environmental Protection Division 2004).

In addition to the thermal sensitivity of trout, the reproductive behavior of trout increases their susceptibility to alterations of instream habitat. Trout bury their eggs in spawning redds (nests) that adult trout prepare in coarse gravel particles. Sedimentation reduces the suitability of coarse gravel particles for spawning because fine sediments fill in the interstitial spaces between larger gravel particles and embed the larger particles. Increasing embeddedness clogs the interstitial spaces within trout spawning redds and, thereby, smothers and prevents the emergence of trout fry and decreases food resources (Hausle and Coble 1976; Kondolf 2000; Curry et al. 2004; Suttle et al. 2004). Suttle et al. (2004) suggest that in gravel bed streams, even small increases in fine sediment can impair trout habitat. Thus, because trout require instream habitats with cool stream temperatures and clean streambeds, trout populations are reduced or eliminated by landuses such as urbanization, agriculture, and forestry (for a review, see Marcus 1990) that increase stream temperature (Theurer 1985; Beschta 1987; Beschta 1988; Li 1994; Beschta 1997; Poole and Berman 2001) and/or the rate of sediment delivery to streams (Megahan 1992; Eaglin and Hurbert 1993; Espinosa 1997; Huntington 1998).

Like trout, native fishes of the Southern Appalachian Mountains are likely to be sensitive to instream habitat alterations such as warmer stream temperatures and increased embeddedness of coarse streambeds. These high- and upland fishes are characterized as being sediment-intolerant and preferring cool water whereas fishes in the region's lowlands are characterized as being tolerant of higher sediment and temperature conditions (Scott and Helfman 2001). These mountain fishes tend to spawn directly over streambeds or within rock crevices. Jones et al. (1999) found that as the length of nonforested patches along the stream network increased, riffle embeddedness increased and habitat diversity decreased corresponding with a shift in fish assemblage from fishes dependent on clean substrate particles for spawning toward fishes that tend to guard eggs in benthic nests. Similarly, Sutherland et al. (2002) reported that the abundance of Southern Appalachian fishes requiring clean gravel substrate for spawning decreased as the embeddedness of streambed substrate particles and streambed instability increased. Because of their limited ranges, specialized habitat requirements, and benthic spawning behavior, these high- and upland fishes are vulnerable to past and present alterations of instream habitat (Etnier 1997; Warren et al. 1997; Harding et al. 1998; Warren et al. 2000). As the region's mountain fishes are adversely impacted by alterations of instream habitat, they are being replaced by native lowland and cosmopolitan fishes leading to the homogenization of the region's fish fauna (Scott and Helfman 2001).

Research overview

The conceptual model linking riparian forest conditions and trout and Southern Appalachian fishes (Figure 1.1) functions as the framework for this research. Overall, this research contributes to understanding how the reduction in the width of State of Georgia's mandatory

riparian buffers will alter local instream habitats and, thereby, impact the self-sustaining trout populations and fish assemblages of Southern Appalachian streams in Northern Georgia. In Chapter 2, I quantify the relationships between riparian forest cover, instream habitat, and trout populations and evaluate the impact of reduced buffer widths on instream habitat and trout populations at both the stream segment and stream network scales. In Chapter 3, I relate Southern Appalachian fish assemblages to instream habitat and landscape conditions in order to determine how these fish assemblages may be affected by the reduction in riparian buffer width. In the concluding chapter, I discuss the implications of the State's changing riparian buffer policies on the future capacity of Georgia's Southern Appalachian streams to sustain trout and the region's native fishes.

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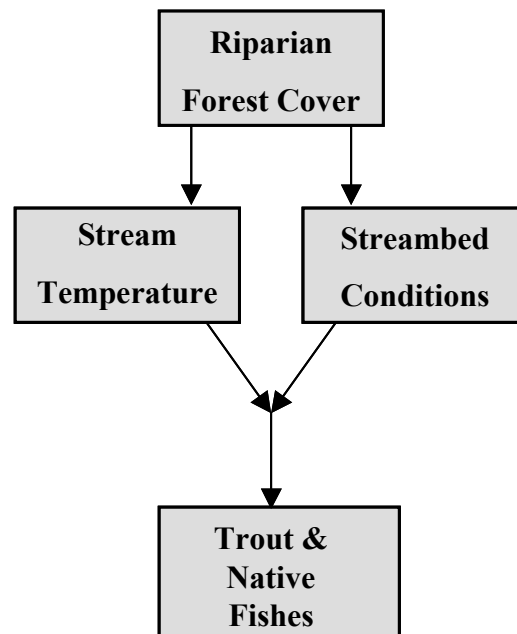


Figure 1.1. Conceptual model used as the research framework for linking riparian forest cover to trout populations and fish assemblages in Southern Appalachian trout streams draining Northeast Georgia. Riparian forest cover influences stream temperature and streambed conditions, which in turn likely influence the region's self-sustaining trout populations and fish assemblages.

CHAPTER 2

PREDICTING TROUT POPULATION RESPONSE TO REDUCED RIPARIAN BUFFER WIDTHS IN NORTHERN GEORGIA, USA¹

¹ Jones, K.L., G.C. Poole, J.L. Meyer. 2004. To be submitted to *Ecology and Society*.

ABSTRACT

In 2000, the Georgia State Legislature reduced the width of mandatory-forested riparian buffers along the State's trout streams from 30 m to 15 m. This research evaluated the likely impact of this reduction in buffer width on trout populations by: 1) quantifying relationships between riparian forest conditions, instream habitat, and young-of-the-year rainbow and brown trout (*Oncorhynchus mykiss* and *Salmo trutta*); 2) applying these quantified relationships at the stream segment and stream network scales to determine the efficacy of 15-m buffers for protecting instream habitat; 3) examining existing forest conditions along the stream network; and 4) assessing the existing thermal alteration of trout streams. Stream temperatures were consistently and negatively related with % riparian forest cover and elevation. In this study landscape, riparian forest cover overwhelms any influence of basin forest cover in determining stream temperature. Fine sediment in riffles was negatively related with % riparian forest cover and maximum reach velocity. Biomass of young-of-the-year trout (< 150 mm in total length) was negatively related with stream temperatures, riffle embeddedness scores, and maximum reach depth. At the stream segment scale, reducing the width of mandatory-forested buffer from 30 m to 15 m will increase stream temperatures by 1.6 to 2.3°C depending on summer weather conditions and riffle embeddedness scores will increase by 4.2 points across a range of maximum stream velocity conditions. These seemingly small increases in stream temperature and riffle embeddedness will reduce the biomass of young-of-the-year trout by 81% to 88%, depending on elevation and summer weather conditions. Within the trout stream network, 63% of streams segment are likely to support reproducing trout populations with the presence of a 30-m buffer; this percentage drops to 9% with a 15-m buffer. These quantitative analyses at both the stream segment and trout stream network scales imply that a 15-m buffer is not effective at

maintaining the instream conditions necessary for self-sustaining trout populations. Due to existing disturbance of riparian forests, substantial alteration of the thermal conditions of trout streams has occurred along the trout stream network in Northern Georgia. Further deforestation of riparian areas will contribute to the warming of trout streams. Results of this study suggest that the continued ability of Georgia's mountain streams to maintain trout populations and support the recreational trout fishery is apt to be undermined by warmer stream temperatures and increased fine sediment delivery anywhere the width of riparian buffers is reduced to 15 m.

Introduction

Northeast Georgia encompasses the southern terminus of the Southern Appalachian Mountains, an ecoregion where human population and development pressures are increasing (Wear and Bolstad 1998). Georgia is the 6th fastest growing state in the United States, and human populations in Georgia's Southern Appalachian counties increased 23.8 – 75.5% between 1990 and 2000 (U.S. Census Bureau 2000). Coupled with higher human populations is suburban development, which is replacing agriculture as the region's dominant landuse and is projected to continue within riparian areas (Wear and Bolstad 1998). Because of these human population and development pressures, Northeast Georgia faces the challenge of balancing the protection of natural resources with the development of private lands.

In Georgia, trout and the streams that support them are important natural resources. Trout are associated with clean and healthy waters and fishing trips in rhododendron-lined mountain streams. Trout are not only objects of recreational pursuit but also sources of revenue for the State's economy. Expenditures associated with trout fishing are estimated to exceed \$172

million annually in Georgia (DeMeo et al. 2004). Annually, Georgia's trout fishery generates \$3.4 million in state sales tax, \$2 million in state income tax, and \$4.8 million in federal income tax (Mahari 1996).

In 2000, the Georgia State Legislature reduced the minimum width of mandatory-forested riparian buffers along its Southern Appalachian trout streams from 30 to 15 m in order to increase the amount of developable land. This new 15-m buffer is narrower than the average mandatory buffer width in the Southeastern ecoregion, United States, and Canada (Lee et al. 2004). Reducing the buffer width may elevate stream temperatures and increase delivery of fine sediment to streams (Gregory et al. 1991; Waters 1995; Scott et al. 2002; Pusey and Arthington 2003). Warmer stream temperatures and increased sedimentation can adversely affect the metabolism, growth, and reproductive success of trout (Hausle and Coble 1976; Crisp 2000; Curry et al. 2004; Suttle et al. 2004). Because of the potential for elevated stream temperatures and sedimentation with the 15-m buffer, the Georgia State Legislature commissioned this study to evaluate the impact of reduced riparian buffer widths on the State's trout populations.

Study landscape

The study landscape is located within the Blue Ridge physiographic province of Northeast Georgia, U.S.A. (Figure 2.1) and crosses the Chattahoochee, Coosa, Savannah, and Tennessee river basins. Under a closed forest canopy, streams in this area are cool (17.5 – 20.0°C maximum 7-day average maximum temperature) and swiftly flow over bedrock, gravel, and cobble streambeds. These mountain streams support the southernmost naturally reproducing populations of native brook trout (*Salvelinus fontinalis*) and introduced brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*) in the eastern United States (Benhke 2002).

Also, these streams belong to the larger Southeastern ecoregion, which supports the highest levels of temperate freshwater fish diversity and endemism in the world (Warren et al. 2000).

Our study focused on the State of Georgia's trout stream network, which is designated by the Georgia Department of Natural Resources (DNR). Georgia DNR classifies trout streams as either "primary" trout streams where self-sustaining trout populations likely exist or as "secondary" trout streams where viable trout populations likely do not exist but where trout are nonetheless stocked to support a put-and-take trout fishery (England 1998; Georgia Environmental Protection Division 2004).

Methods

A conceptual model linking riparian forest cover and trout populations functioned as the framework for this research (Figure 2.2). We investigated the influence of riparian forest conditions on trout at both the stream segment and stream network scales by: a) characterizing each stream segment in the trout stream network with a spatial database; b) establishing two sets of sampling sites (Group 1 sites represent the range of trout streams across the study landscape while Group 2 sites permitted analyses that separate the effects of riparian forest versus basin forest conditions on instream habitat); c) developing stream segment scale relationships between trout populations and riparian and basin forest conditions; and d) extrapolating these stream segment scale relationships to the trout stream network in order to identify stream segments likely to support trout with 30- versus 15-m buffers.

Trout stream GIS database

We developed a Geographic Information System (GIS) database of DNR's trout stream network, which was useful for both stream segment and stream network scales of analysis. The trout stream network was divided into stream segments between channel junctions (Figure 2.1C). Stream order, a measure of position in the network (Strahler 1952) where larger orders are indicative of larger streams within the network, was calculated for 8,000 trout stream segments. For segments 2nd order or larger, we determined basin area, elevation, and % forest cover. Percent forest cover was derived from Landsat 1998 data and measured for: 1) the riparian buffer (a 30-m zone located along each side of the stream segment and entire stream network above a segment); 2) the basin of each stream segment (excluding the riparian buffer); and 3) the “adjacent area” of each segment (the land area draining directly to the stream segment) (Figure 2.1C). Although Landsat data yield estimates of % riparian forest cover, % riparian forest cover proved to be a reliable indicator of buffer width when at least 1.2 km of stream was analyzed (Figure 2.3). The GIS database also included a map of Georgia Conservation Lands, or “public lands” controlled by state and federal agencies and non-governmental organizations (NGOs, Table 2.1).

Site selection

We identified sampling sites from the GIS database that were representative of trout streams across the study landscape where we investigated the stream segment scale relationships between riparian forest cover, instream habitat, and trout. To create a pool of potential sampling sites, we selected all segments in the database that were a) 2nd order or greater; b) > 1 km in length; and c) drained basins between 5 and 50 km² (n = 4019). We chose two groups of sites from this pool.

First, cluster analysis (Figure 2.4) identified 28 sites (Group 1) greater than 3rd order and representative of basin area, elevation, and riparian forest cover conditions across the trout stream network (Table 2.2, Figures 2.5). While basin area, elevation, and riparian forest cover values for the Group 1 dataset are independent of each other, % riparian and basin forest cover values are significantly correlated ($p < 0.001$, Figure 2.6). Because of this correlation, we were unable to use this dataset to differentiate between the effects of riparian versus basin forest cover on instream habitat conditions. Therefore, we identified 17 additional sites (Group 2) that covered similar gradients in basin area, elevation, and riparian cover as Group 1 but lacked strong correlation between riparian and basin forest cover values (Table 2.2, Figures 2.5).

Trout sampling

Trout were sampled at all Group 1 sites in summer 2001 and resampled at 10 sites in summer 2002. For quantitative catch data, all habitats within the 50-m reach were sampled extensively in a single pass with a backpack electric shocker, dip nets, and seine net. Extended sampling was then conducted to detect the presence of additional fish species. Overall, we sampled an area approximately 30 times average stream width at each site (range sampled: 88 – 440 m). While brook trout (*Salvelinus fontinalis*) are native to these streams, the only trout species collected were introduced game species, rainbow trout (*Oncorhynchus mykiss*) and brown trout (*Salmo trutta*). Since larger trout (year class 1 and higher) are stocked by Georgia Wildlife Resources Division and are harvested by anglers, their age class structure is an unreliable indicator of trout habitat conditions. Therefore, analysis focused on naturalized young-of-the-year (< 150 mm in length) trout, which are smaller than stocked trout, below harvestable length, and are indicators of reproducing trout populations.

Habitat Sampling

Onset Corp.® HOBO temperature data loggers installed at each site provided a continuous record of stream temperatures during 2001 – 2003 for Group 1 sites and during 2003 for Group 2 sites. For each site, we calculated the average of daily maximum stream temperatures during the warmest 7-day period of the summer (M7DAM temperature, °C) for a quantitative indicator of the peak thermal stress experienced by trout.

Using methods adapted from Gordon et al. (1992), we examined streambed substrate conditions by assessing riffle embeddedness (the % of substrate in riffle habitats surrounded by fine sediments) at all Group1 sites. In order to eliminate observer bias, the same observer estimated % riffle embeddedness using a clear bottom viewbox at all sites with riffle habitat. Riffle embeddedness was assessed in 10 locations within 3 riffles at each site. Each of the 30 observations was assigned a class value based on four classes of increasing embeddedness (1 = 0 – 25%, 2 = 25 – 50%, 3 = 50 – 75%, and 4 = 75 – 100%). Class values were summed for site riffle embeddedness scores; the range of possible riffle embeddedness scores is 30 (0 – 25% riffle embeddedness for all 30 observations) to 120 (75 – 100% riffle embeddedness for all 30 observations). High riffle embeddedness scores indicates poor trout habitat because fine sediments can negatively affect the incubation and emergence of trout fry.

Local stream slope was measured over 100 m, which included the 50-m sampling reach and 25 m above and below the reach. These values are reported as percent change in slope over 100 m. 50 depth and velocity measurements were taken randomly across each study reach to capture the range of depth and velocity conditions. Velocity was measured using a Marsh-McBirney Flo-Mate 2000™ flow meter.

Stream segment scale data analyses

Field data were analyzed with multiple linear regression to quantify the relationships between riparian forest cover, stream temperature and streambed sediments, and young-of-the-year trout populations (Figure 2.2). JMP Version 5 (SAS Institute Inc., Cary, NC, USA) was used for all analyses. Prior to inclusion in regression analyses, variables were tested for normality with the Shapiro-Wilk goodness of fit test and transformed when needed. Young trout biomass (derived from length—weight relationships, Schneider et al. 2000) and basin area data were log transformed whereas % riparian and basin forest cover values were arcsine squareroot transformed.

Based on prior research investigating the influence of landscape variables on stream temperature in the Southern Appalachians (Scott et al. 2002), we developed a multiple regression model to describe the influence of % riparian cover, elevation and basin area on M7DAM temperature. Contrary to the findings of Scott et al. (2002), basin area was not a significant predictor of stream temperature for this dataset, which is likely due to the smaller range of basin sizes investigated in this study. Thus, the form of the regression equation for predicting M7DAM temperatures became:

Equation 1: $M7DAM\ temperature = a + b * [Arcsine(\% \text{ riparian cover}^{0.5})] + c * (Elevation)$

Parameters for Equation 1 were derived separately for 2001, 2002, and 2003 from the Group 1 field data because weather conditions varied among summers. To determine whether riparian or basin forest conditions were driving stream temperature, we plotted riparian and basin cover data

against elevation-adjusted M7DAM temperatures for Group 2 sites. Note that Group 2 temperature data could not be used for interannual analyses because these data were only available for 2003 and that Group 1 data could not be used to assess the relative influences of riparian versus basin forest cover on stream temperature because riparian cover and basin forest cover values were significantly correlated and both strong predictors of stream temperature.

Landscape (% riparian cover in entire stream network above the site, % basin cover, basin area, and elevation) and instream habitat variables (local stream slope, maximum reach velocity, and maximum reach depth) were included in a reverse step-wise regression analysis to identify significant ($\alpha = 0.05$) predictors of streambed sediment conditions (riffle embeddedness scores). Despite the autocorrelation between riparian and basin forest cover, basin forest cover was not a significant predictor of sediment conditions. Thus, we excluded basin forest cover from further analyses of sediment conditions within trout streams. Analyses yielded the following equation for predicting riffle embeddedness:

Equation 2: Riffle embeddedness = $a + b * [\text{Arcsine } (\% \text{ riparian cover}^{0.5})] +$
 $c * (\text{Maximum Reach Velocity})$

Also, M7DAM temperatures, riffle embeddedness, and instream habitat variables (local stream slope, maximum reach velocity, and maximum reach depth) were included in a reverse step-wise regression analysis to determine significant predictors of young trout biomass for sites with young trout in 2001 and 2002. The form of the regression equation for predicting young trout biomass was:

Equation 3: $\text{Log (young trout biomass)} = a + b * (\text{M7DAM temperature}) +$
 $c * (\text{Riffle embeddedness}) + d * (\text{Maximum reach depth})$

In combination, Equations 1, 2, and 3 describe the linkages between riparian cover, instream habitat, and trout populations and allow for a quantitative assessment of the expected response of trout populations to reduced buffer widths (Figure 2.2).

Stream network scale data analysis

We mapped the likelihood of trout persistence in each trout stream segment of the network with either a 30- or 15-m wide riparian buffer. To accomplish this, the probability of young trout presence was related to M7DAM temperatures with a logistic model for the Group 1 sites sampled in 2001 and resampled in 2002. From this model, we identified three categories of trout habitat: low, marginal, and high, which correspond to a < 50%, 50% – 90%, and > 90% probability of young-of-the-year trout being present at a site. Using Equation 1 with parameters derived for 2001 (an intermediate summer, Table 2.3), 2nd – 5th order trout stream segments likely to support trout with a 30- versus 15-m wide riparian buffer were classified and mapped according to these three habitat quality categories.

Additionally, using the GIS database, we assessed the warming of trout streams (“thermal alteration”) as a result of existing riparian and basin forest cover conditions on both private and public lands. We define “existing thermal alteration” as the difference between the current stream temperature and the stream temperature expected under a full 30-m wide riparian buffer for a given stream segment.

Results

Local effects of network riparian cover on trout populations

First, parameter estimates from Equation 1 demonstrated that 2001, 2002, and 2003 M7DAM temperatures were consistently and negatively related with % riparian forest cover and elevation (Table 2.3). For the Group 1 dataset, both % riparian cover and % basin cover were related with M7DAM temperatures. This result is not surprising in light of the significant correlation between these two forest cover variables. For the Group 2 dataset where % riparian and forest cover are independent, however, elevation-adjusted M7DAM temperatures were negatively related to % riparian cover only (Figure 2.7). Thus, riparian forest cover overwhelms any influence of basin forest cover on stream temperatures in this study landscape.

Second, in Equation 2, riffle embeddedness scores increased as % riparian forest cover and maximum reach velocity decreased (Equation 2, Table 2.3). Deforestation of riparian areas leads to increased delivery of fine sediments, which can embed larger-sized sediment particles within riffle habitats.

Third, the biomass of young-of-the-year trout was negatively related to M7DAM temperatures, riffle embeddedness values, and maximum reach depth (Equation 3, Table 2.3). Since young trout biomass decreased as both stream temperature and fine sediments in riffles increased, this equation shows that young trout populations will be adversely affected by any landuse that warms streams and increases the delivery of fine sediment to streams.

The conceptual framework of this research becomes a quantitative model because we were able to quantify Equations 1 – 3 that describe the relationships between riparian forests, instream habitat conditions, and young trout populations. We can apply this quantitative model in determining the expected stream temperature and riffle embeddedness conditions and, in turn,

the expected loss in young trout populations if the width of riparian buffers is reduced from 30 to 15 m along the entire stream network above a stream segment. Under this scenario, M7DAM temperatures will warm by 1.6°C in cool, wet summers and 2.3°C in warm, dry summers (Table 2.4). Also, fine sediment in riffle habitats increases by roughly 4.2 across a range of maximum stream velocity conditions (Table 2.5). As a consequence of these modest increases in stream temperature and riffle embeddedness, the biomass of young-of-the-year trout will be reduced by 81% to 88% , depending on elevation and summer weather conditions (Table 2.6).

Stream segments potentially supporting trout with 30- versus 15-m wide buffers

Across the trout stream network, 63% of Georgia's 2nd – 5th order trout stream segments could maintain temperatures associated with high or marginal quality thermal habitat with a 30-m wide forested riparian buffer (Figures 2.8, 2.9). These streams are located primarily in the northern half of the study landscape. The remaining 37% of trout streams would have temperatures associated with low quality thermal habitat and are located mostly in the southern half and western edge of the study landscape. With a 15-m wide forested riparian buffer, less than 9% of these streams could maintain temperatures associated with high or marginal quality thermal habitat. These streams are confined to the very highest elevations of the study landscape. Fully 91% of the State's trout streams would have temperatures indicative of low quality thermal if the stream's associated buffer was 15-m wide (Table 2.7).

Existing forest conditions along the trout stream network

Riparian areas have been preferentially deforested relative to basins, particularly on private lands (Figure 2.5). In contrast, streams with wide riparian buffers are located predominately on public

lands (Figure 2.10). On private lands, riparian buffers are generally less than 30 m wide and 80% of the “adjacent basin areas” (Figure 2.1) are less than 90% forested (Table 2.8). Existing levels of deforestation on private lands is of concern because private lands drain to approximately 60% of the State’s trout streams, including most primary (streams potentially supporting self-sustaining trout populations) and large (3rd through 5th order) trout streams (Table 2.9). Furthermore, the northern half of the study landscape contains 67% of the State’s trout streams, ~75% of which are designated as primary trout streams (Table 2.10). Riparian areas of this core primary trout stream habitat are more deforested than the basins (Table 2.10). While riparian areas have been disturbed on public lands as well, riparian buffers are usually 30 m or wider and basins are nearly completely forested (Table 2.8).

As a consequence of riparian deforestation across the region, existing levels of the thermal alteration of trout streams are substantial (Figure 2.11). Given existing levels of riparian forest cover, one third of the 2nd – 5th order trout streams have temperatures >1°C warmer than expected with a 30-m wide riparian buffer. Most of this existing thermal alteration occurs on private lands, where half of the streams are >1°C warmer than expected with a 30-m buffer. In contrast, only 1 in 20 streams on public lands have experienced the same level of warming.

Discussion

Consequences of the reduced riparian buffer width for self-sustaining trout populations

Our analyses predict seemingly modest increases in both stream temperature and fine sediment if the width of forested riparian buffers is reduced from 30 to 15 m. Yet, relationships observed between instream habitat conditions and young trout populations suggest that these modest changes will substantially reduce trout populations. Only 2.0°C separates high quality thermal

habitat from low quality thermal habitat (Figure 2.8). With a 30-m wide buffer, lower elevation streams will warm to thermal levels unlikely to maintain trout populations ($>21.5^{\circ}\text{C}$) during warm summers. In contrast, where widths of associated buffers are 15 m or less, trout streams at most elevations will likely to be too warm to support trout during cool and warm summers (Table 2.4). Riffle embeddedness scores will increase by 4.2 points if the forested buffer width is reduced from 30 to 15 m. In sampling streams representative of the Georgia trout stream networks, we observed site riffle embeddedness scores ranging from 31 to 47. Since only 16 points separates sites with lowest and highest riffle embeddedness values, a 4.2 increase is representative of 25% of the variation observed in riffle embeddedness conditions across Northeast Georgia. Furthermore, the average riffle embeddedness score for sites where young trout were observed was 37.7 whereas the average for sites where young trout were not observed was 42.2. Since only 4.5 separates these average riffle embeddedness scores, an increase of 4 points appears to be indicative of an increase in fine sediment conditions that is no longer conducive to self-sustaining trout populations. With a 15-m buffer, streams with different flow conditions, particularly slow-moving streams, would have riffle embeddedness scores approaching or exceeding 42.2, the average score above which young trout were not observed (Table 2.5).

As both stream temperature and fine sediments increase and alter trout habitat, the biomass of young-of-the-year trout is reduced by 81% to 88%, depending on summer weather conditions and elevation (Table 2.6). However, the capture efficiency of our trout sampling techniques may influence these estimates of the reduction in trout populations associated with reduced riparian buffer widths. Peterson et al. (2004) found that capture efficiency of stream-dwelling salmonids declined as instream habitat complexity (such as undercut banks and higher

% of cobble substrate) increased. We observed that streams with more extensive riparian buffers tended to have more habitat complexity, such as large woody debris in the channel, bedrock/cobble streambeds, and bars/backwaters. If capture efficiency varied in our sampling, then capture efficiency was likely lower at sites with wider riparian buffer and higher at sites with narrower buffers. Thus, our sampling methods may have underestimated young trout populations at sites with wide buffers. In turn, our estimates for the expected reduction in young trout biomass associated with the reduction in buffer width are likely conservative. Nonetheless, the expected reduction in the biomass of young trout is substantial (over 80%) suggesting that the viability of self-sustaining trout populations in Georgia's trout streams will be undermined by the instream habitat alterations associated with reducing the buffer width from 30 to 15 m.

At the scale of the entire trout stream network, the effect of 15-m buffers on stream thermal conditions and, thereby, on trout populations is striking (Table 2.7, Figure 2.9). Although the results of this study do not identify the minimum buffer width necessary to maintain self-sustaining trout populations across Northeast Georgia., our results provide strong evidence that 15-m wide riparian buffers would be inadequate.

Implications of existing riparian buffer conditions for Georgia's trout streams

Disturbance of forest cover within the riparian zone varies across the trout stream network (Figure 2.10). This variation is shaped largely by the region's land ownership patterns and also by topography, enforcement of buffer regulations, and past landuses such as agriculture. Most trout streams start on public lands with wide buffers and then flow through private lands with

variable buffer widths. Existing riparian zones have been preferentially deforested relative to their basins on both private and public lands, despite generally wider buffers on public lands (Figure 2.5).

At high elevations in Northern Georgia, a full forest canopy can shade trout streams and maintain cool stream temperatures despite the potential for substantial solar heating of streams during summer months. Because of this strong relationship between riparian forests and stream temperature, existing levels of riparian deforestation have converted many streams from high to low quality thermal habitat for trout across Georgia's trout stream network (Figure 2.11). In addition, since a majority of Georgia's trout streams flow through private lands (Table 2.9), a substantial portion of the trout stream network is vulnerable to increased riparian deforestation permitted with the reduced riparian buffer width. Further deforestation of riparian zones will lead to increased thermal alterations of the trout stream network, despite high levels of forest cover potentially elsewhere in basins draining to trout streams, and thereby, reduced trout populations. It seems feasible that self-supporting trout populations may ultimately be limited to Georgia's public lands, where riparian buffers are wider (> 30 m) and existing levels of thermal alteration are low (Table 2.8; Figure 2.11), unless actions are implemented to protect and restore riparian buffers on private lands.

In this research, we demonstrated that stream temperature and fine sediment conditions in riffles were best predicted by riparian forest conditions. Prior research has reported that the best predictor of instream habitat conditions is basin landuse (Roth et al. 1996), riparian landuse (Jones et al. 1999; Lammert and Allan 1999; Wang et al. 2003), or landuse interactions at multiple spatial scales (Sponseller et al. 2001; Scott et al. 2002). Determining whether riparian or basin cover is the best predictor of instream conditions requires the statistical separation of the

influences of riparian and basin landuse predictors on response variables. Significant correlations between riparian and basin scale predictors often hinder research attempting to separate these influences. Thus, we specifically identified that Group 2 sampling sites to ensure that riparian along the entire stream network above a sampling site and basin forest cover values were not correlated. This type of sampling design was not done in prior studies. Also, we strategically conducted this research with a conceptual framework that related forest conditions to specific stream habitat parameters (stream temperature and streambed sediment conditions) that were most likely to be strong predictors of trout populations. Analyses that examine the relationships between forest cover conditions and habitat variables describing a single habitat component (e.g., stream temperature) may be more apt in identifying direct linkages than analyses with habitat variables describing multiple components of instream habitat (e.g., habitat diversity indices used by Roth et al. 1996).

While our research did not find basin forest cover to be the driver of stream temperature conditions in Northern Georgia, these results should not be interpreted as evidence that forest conditions in areas beyond the riparian zone do not influence water quality and trout habitat. Our targeted research approach did not examine the wide range of instream habitat conditions that may influence trout populations, such as flow regime, water quality, and basin-scale sediment regimes. These instream habitat conditions are influenced by not only riparian forest conditions but also basin forest conditions. Increasing landuse alterations of basins can overwhelm the capacity for riparian forest cover to support high-quality instream habitat and biotic communities (Roth et al. 1996; Roy 2004). Thus, management of basin forest conditions is also an important component for maintaining instream habitats conducive to self-sustaining trout populations in Northern Georgia.

Conclusions

Riparian buffer zones are critical areas for maintaining the thermal and streambed sediment conditions required to support self-sustaining trout populations in Northern Georgia. Both stream temperatures and fine sediment levels in riffles will increase with a 15-m versus a 30-m wide riparian buffer. Though the changes may appear modest, trout populations are especially sensitive to these expected changes. In essence, the results from this study suggest that the continued ability of North Georgia's trout streams to maintain trout populations and support a trout fishery will be jeopardized anywhere the widths of riparian buffers are reduced to 15 m.

Instream habitat alterations associated with 15-m riparian buffer are expected to reduce young-of-the-year trout populations by over 80% relative to populations expected with 30-m buffers. Furthermore, 91% of trout streams in Georgia would be unlikely to support naturally reproducing trout populations if the width of their associated riparian buffers was reduced to 15-m. Because the majority of the trout stream network is located on private lands where riparian zones have already been substantially deforested, the maintenance and restoration of riparian forests on private lands will be important components in management plans designed to maintain self-sustaining trout populations in Northern Georgia.

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Table 2.1. Data layers included in the Trout Stream GIS database.

Data type	Source	Purpose
30-m Digital Elevation Model (DEM), Georgia	US Geologic Survey	Used to determine elevation and basin area of stream segments
1 : 24,000 hydrography, Georgia	US Geologic Survey	Provided stream locations on the landscape and used to determine stream order.
1998 Landcover of Georgia (30-m resolution Landsat satellite imagery)	Natural Resources Spatial Analysis Laboratory (NARSAL), Institute of Ecology, University of Georgia	Used to determine riparian and basin forest cover conditions.
Georgia Conservation Lands and Georgia Department of Natural Resources	Natural Resources Spatial Analysis Laboratory (NARSAL), Institute of Ecology, University of Georgia	Used to identify trout streams on private versus public lands.
Rules and Regulations for Water Quality Control Chapter 391-3-6, Revised August 2000	Georgia Department of Natural Resources	Used with hydrography layer to identify primary and secondary trout streams

Table 2.2. Range of landscape, biological, and instream habitat variables¹ measured for Group 1 and Group 2 sampling sites.

Variable	Units	Group 1	Group 2
Basin Area	km ²	6 - 43	5 - 45
Elevation	m	396 - 740	378 - 885
% Riparian Forest Cover	%	29 - 100	39 - 88
% Catchment Forest Cover	%	73 - 99	44 - 76
2001 Young trout biomass	grams per 100 m ²	0 - 76.6	
2002 Young trout biomass	grams per 100 m ²	0 - 46.2	
2001 M7DAM temperature	deg C	17.6 - 24.5	
2002 M7DAM temperature	deg C	18.3 - 25.1	
2003 M7DAM temperature	deg C	16.9 - 23.5	18.8 - 23.4
Riffle Embeddedness		31 - 47	
Maximum reach velocity	m / sec	0.44 - 2.2	
Maximum reach depth	cm	31 - 123	
Local reach slope		0.2 - 3.3	

¹ Observations were based on four classes of increasing embeddedness (1 = 0 – 25%, 2 = 25 – 50%, 3 = 50 – 75%, and 4 = 75 – 100%). 10 observations were made in three riffles yielding a total of 30 riffle embeddedness observations, which were then summed for site riffle embeddedness values. The range of possible total riffle embeddedness scores is 30 (0 – 25% riffle embeddedness for all observations) to 120 (75 – 100% riffle embeddedness for all observations).

Table 2.3. Equations 1, 2, and 3 quantify the relationships between riparian forest conditions, stream habitat, and trout populations in Southern Appalachian streams. Model significance is noted as following: $p < 0.01 = **$ and $p < 0.001 = ***$.

Equation	Response Variable	Regression parameters				r^2
		a	b	c	d	
1	M7DAM temp	Intercept	% Riparian cover	Elevation		
		2001	32.9672	-4.4827	-0.0127	0.73***
		2002	33.7122	-4.7164	-0.0118	0.70***
		2003	29.9048	-3.2788	-0.0110	0.57***
2	Riffle embeddedness	Intercept	% Riparian cover	Max velocity		
		54.3744	-8.4651	-5.3289		0.40**
3	Young trout biomass	Intercept	M7DAM temp	Riffle embed	Max depth	
		10.0440	-0.2842	-0.0642	-0.0146	0.72***

Table 2.4. M7DAM temperatures (°C) expected for 30- versus 15-m average riparian buffer widths across the elevation range where young trout were observed. Lower values are expected M7DAM temperatures during a cool, wet summer (Equation 1 for 2003) whereas upper values are expected M7DAM temperatures during a warm, dry summer (Equation 1 for 2002).

Summer Weather	Elevation	30-m buffer	15-m buffer	Temperature Increase
Cool				
	488	20.3	21.9	1.6
	578	19.3	20.9	
	740	17.5	19.2	
Intermediate				
	488	21.1	23.2	2.2
	578	19.9	22.1	
	740	17.9	20.1	
Warm				
	488	21.9	24.3	2.3
	578	20.9	23.2	
	740	19.0	21.3	

Table 2.5. Riffle embeddedness values expected for 30- versus 15-m average riparian buffer widths and for range of observed maximum reach velocities (m/sec) where young trout were observed. The expected riffle embeddedness values for maximum velocity equal to 1.0 m/sec were used to predict young trout biomass in Table 2.6.

Maximum velocity (m/sec)	30-m buffer	15-m buffer	Score Increase
0.4	41.4	45.6	4.2
0.8	39.3	43.5	
1.0	38.2	42.4	
1.2	37.1	41.3	
1.6	35.0	39.2	

Table 2.6. Young trout biomass (grams per 100 m²) expected for 30- versus 15-m average riparian buffer widths, across the elevation range where young trout were observed, and for 3 types of summer weather conditions.

Summer Weather	Elevation	30-m buffer	15-m buffer	% Reduction
Cool				
	488	6.62	1.26	81
	578	12.68	2.41	
	740	40.87	7.75	
Intermediate				
	488	4.12	0.53	87
	578	8.70	1.12	
	740	33.37	4.31	
Warm				
	488	2.30	0.28	88
	578	4.61	0.55	
	740	16.07	1.92	

Table 2.7. Expected percentages of streams supporting high, marginal, and low quality thermal habitat (as defined in Figure 2.8) for 30- versus 15-m average forested riparian buffer widths.

The distribution of these streams across the trout stream network is shown in Figure 2.9.

Thermal Habitat Quality	Riparian Buffer width	
	30-m	15-m
High	18.8%	0.7%
Marginal	44.4%	8.1%
Low	36.8%	91.2%

Table 2.8. Riparian buffer widths and forest cover in the drainage basin adjacent to riparian buffers for public and private lands in North Georgia.

<i>Riparian Buffer Widths</i>	Public Lands	Private Lands
>30 m	68%	19%
24 - 30 m	20%	24%
15 - 24 m	9%	31%
< 15 m	3%	27%
<i>Adjacent Basin Forest Cover</i>		
>90%	80%	19%
75%-90%	16%	35%
50%-75%	3%	37%
<50%	0%	9%

Table 2.9. Distribution of primary and secondary trout streams, trout streams by stream order, and total stream segments for public versus private lands.

<i>Trout Stream Designation</i>	Public Lands	Private Lands
Primary	34.7%	38.4%
Secondary	7.3%	19.6%
<i>Stream Order</i>		
1st	27.0%	30.9%
2nd	8.2%	11.4%
3rd	4.3%	9.2%
4th	1.8%	4.4%
5th	0.5%	2.3%
<i>Total Stream Segments</i>	42.0%	58.0%

Table 2.10. Riparian buffer widths and basin forest cover and the distribution of primary versus secondary trout streams on private lands in the northern and southern halves of the study area.

<i>Riparian Buffer Widths</i>	Northern Counties	Southern Counties
>30 m	13%	31%
24 - 30 m	21%	29%
15 - 24 m	32%	28%
< 15 m	34%	12%
<i>Adjacent Basin Forest Cover</i>		
>90%	19%	20%
75%-90%	34%	36%
50%-75%	38%	35%
<50%	9%	8%
<i>Trout Stream Designation</i>		
Primary	49.6%	16.6%
Secondary	176.4%	16.4%

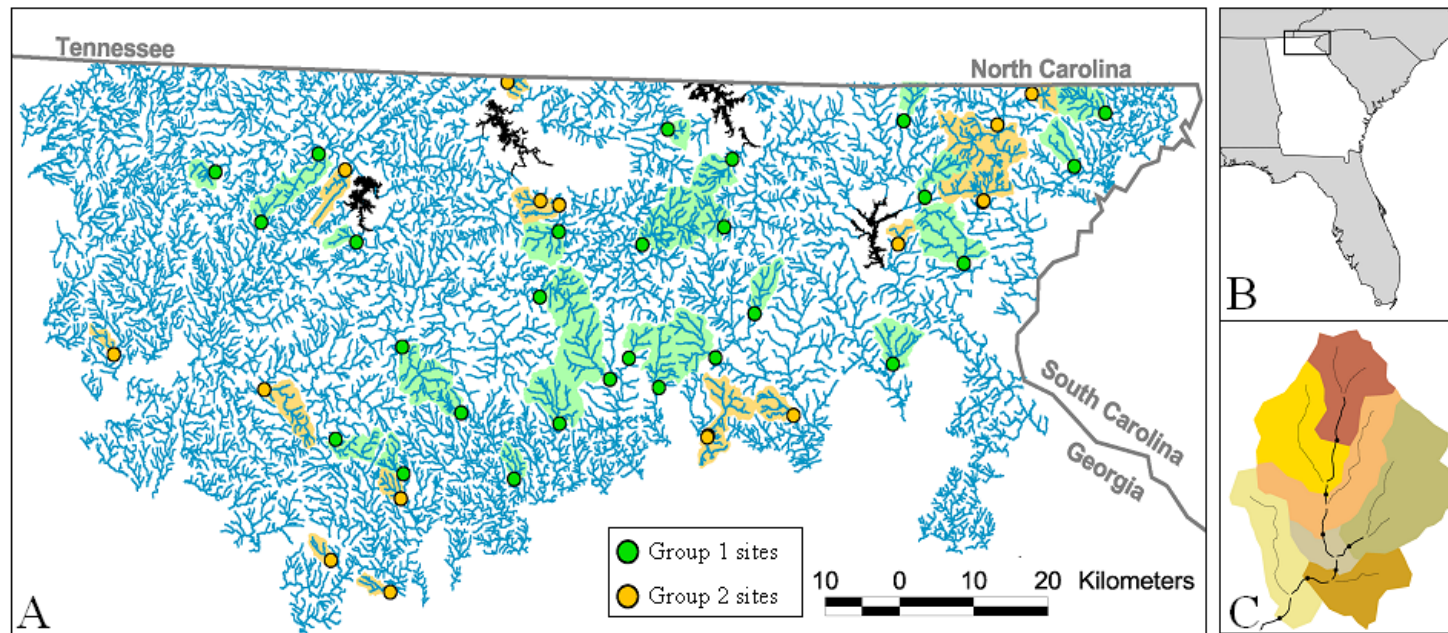


Figure 2.1. Study landscape and sampling sites located in Northeast Georgia. A) The Georgia trout stream network classified by the Georgia Department of Natural Resources with the locations of Group 1 and Group 2 sites. B) Location map outlining the study landscape in the northeastern corner of Georgia. C) Example of a hypothetical stream network. Points denote 2nd order or large stream segments. For each segment 2nd order or larger, we delineated: 1) the entire upstream basin draining to the segment; and 2) the “adjacent area,” or the area along the stream segment directly draining into that segment. For example, the adjacent areas for each of the seven streams segments are denoted with different colored areas. An analysis of forest cover conditions on private versus public lands was conducted with % forest cover values derived for the adjacent areas.

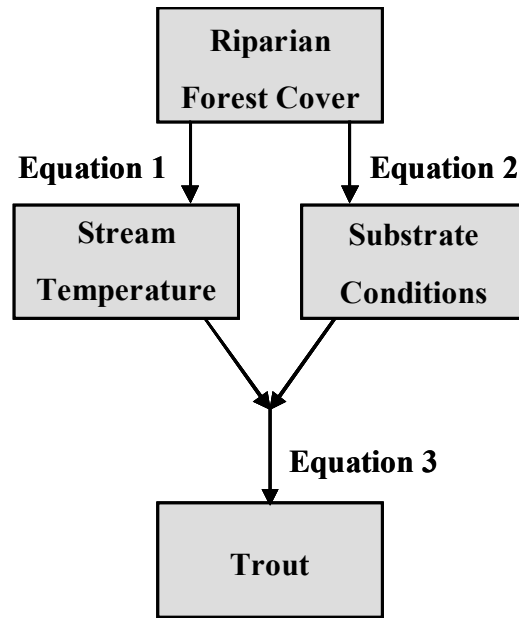


Figure 2.2. Conceptual model used as framework for linking riparian forest cover to trout populations in Northeast Georgia. Riparian forest cover influences stream temperature and substrate conditions, which in turn influence trout populations.

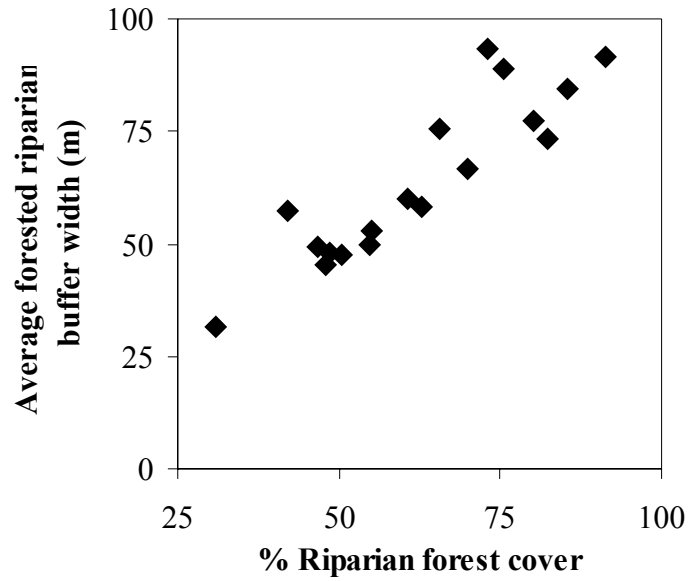


Figure 2.3. Percent riparian forest cover derived from satellite Landsat imagery was significantly related to average forested riparian buffer widths derived from infrared aerial photography ($r^2 = 0.83$, $p < 0.0001$). Analyses were completed for 18 sites located in Rabun and White counties. Both riparian values were calculated within a 30-m zone on each side of the stream for the entire network above a sampling site. Based on this relationship, we could relate 30- and 15-m average forested riparian buffer widths to % riparian cover values.

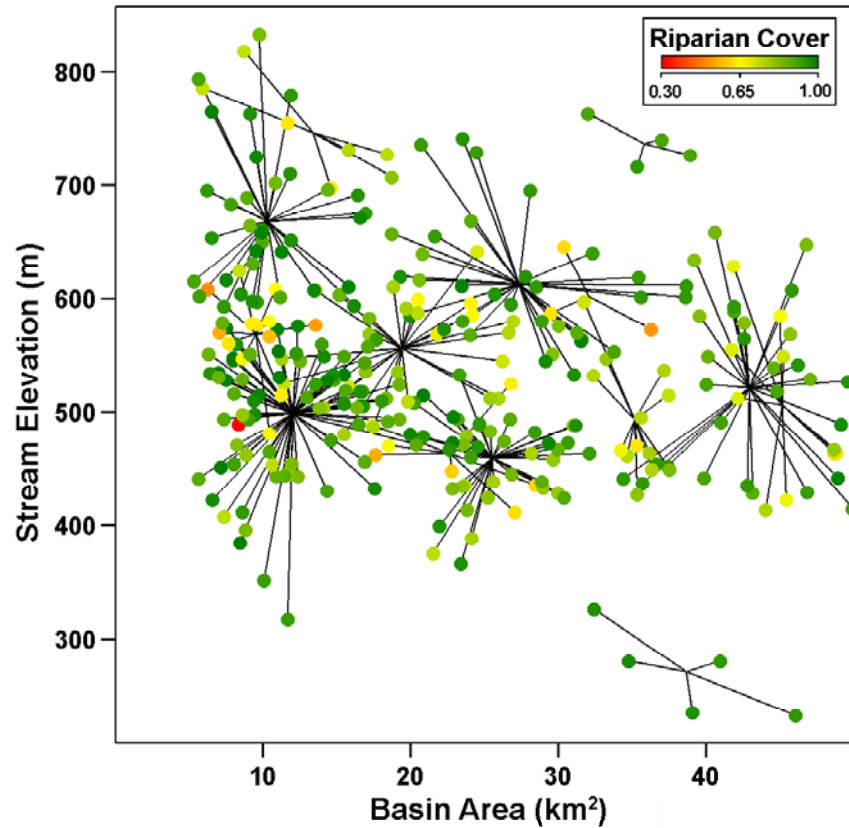


Figure 2.4. Results from a cluster analysis of potential sampling sites for Group 1 using elevation, basin area, and % riparian forest cover. Dots represent characteristics of individual potential sampling sites. Lines group potential sampling sites into groups (clusters) where sites within each group are more similar to each other than to sites in other groups. Group 1 sampling sites were chosen proportionally from each group so that the final Group 1 dataset was representative of the range of trout habitat across the stream network.

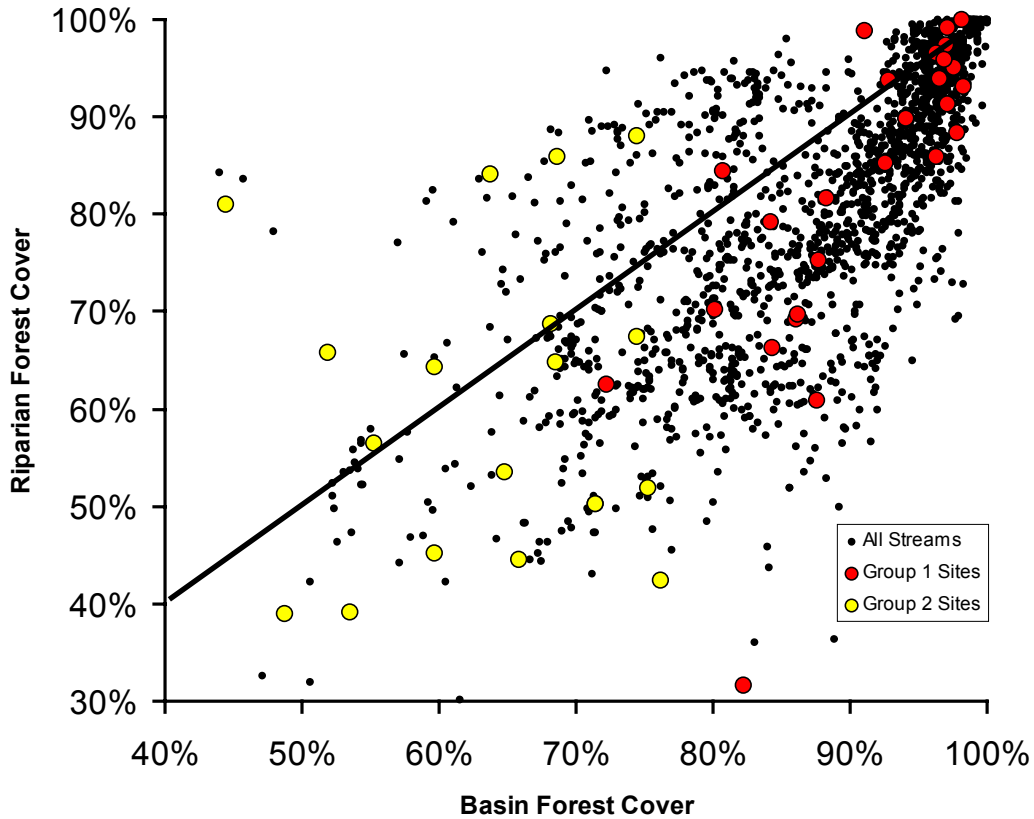


Figure 2.5. % Basin forest cover versus % riparian forest cover for the pool of potential sampling sites that are: a) 2nd order or greater; b) > 1 km in length; and c) drained basins between 5 and 50 km² (n = 4019). The Group 1 dataset (n = 28; “red” dots) are greater than 3rd order and representative of basin area, elevation, and riparian forest cover conditions across the trout stream network. The Group 2 dataset (n = 17, “yellow” dots) were specifically chosen to have % riparian and basin forest cover values that were uncorrelated. Solid black line represents a 1:1 line where % riparian and basin forest cover values are equal.

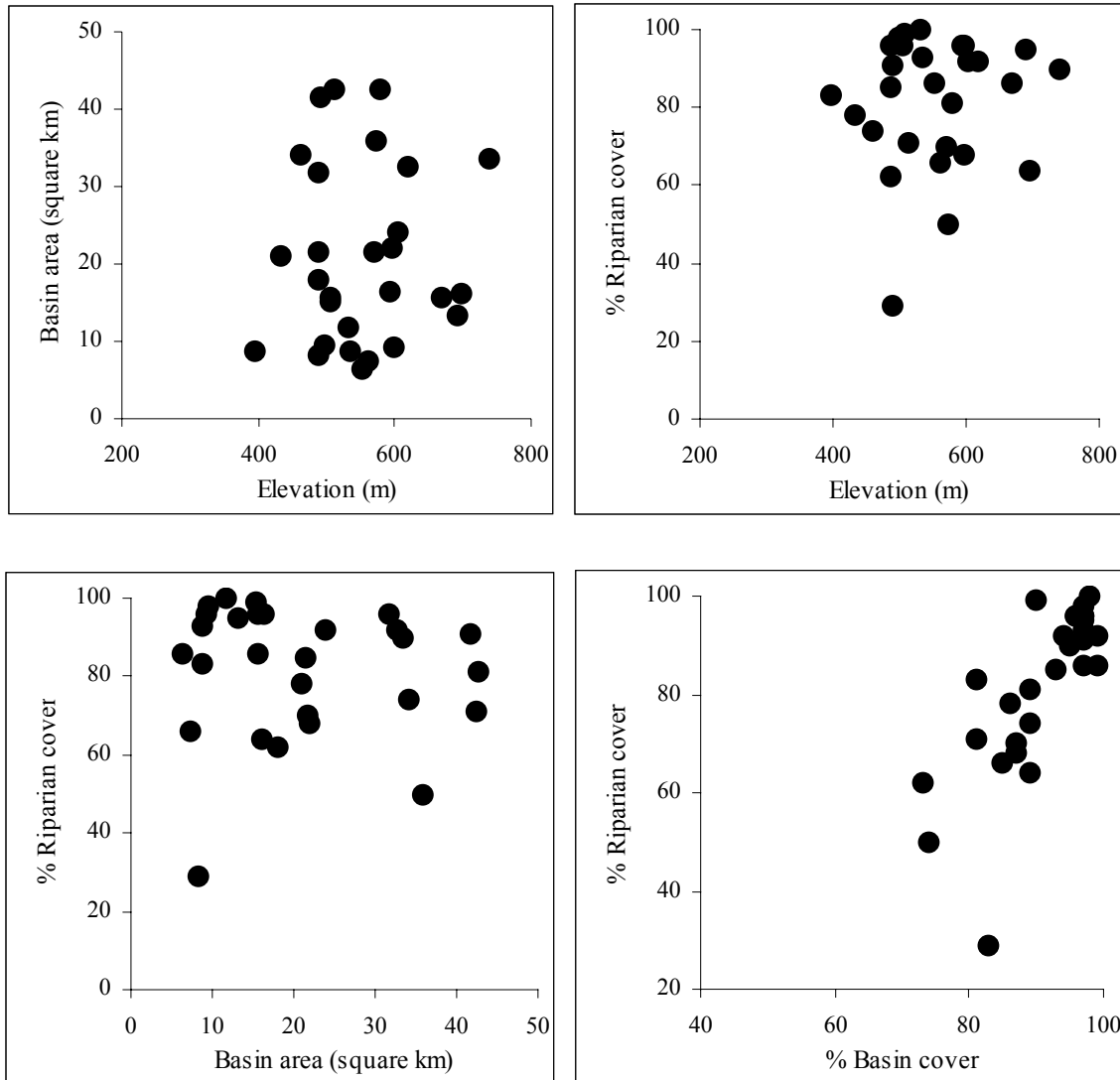


Figure 2.6. Plots of basin area, elevation, and % riparian forest cover against one another illustrate the range of sampling site characteristics and lack of autocorrelation between these independent variables within Group 1 dataset. Values for % riparian forest cover and % basin forest cover, however, are significantly correlated for Group 1 ($r^2 = 0.59$; $p < 0.0001$).

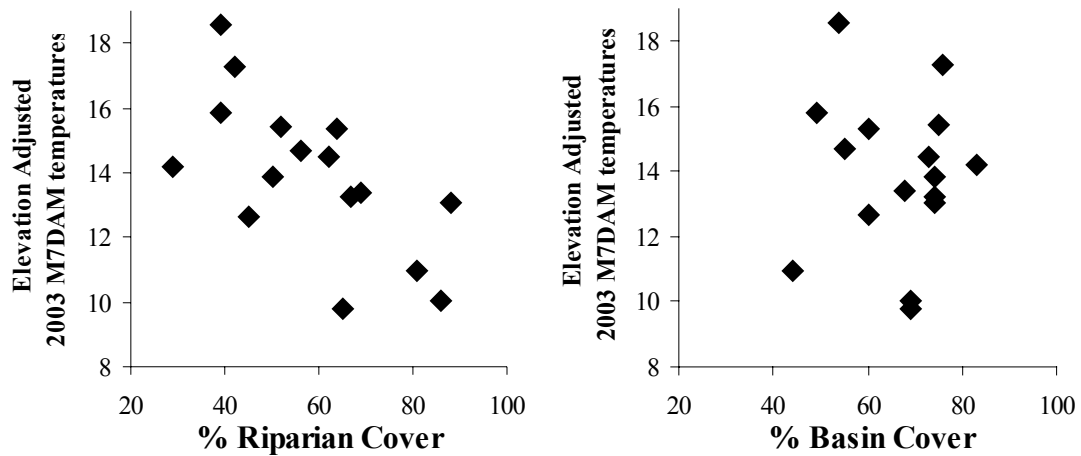


Figure 2.7. Group 2's elevation adjusted M7DAM temperatures related to % riparian and basin forest cover values. Group 2 M7DAM temperatures were algebraically adjusted with the elevation parameter value from Equation 1, derived from Group 1 field data. These elevation-adjusted temperature values were significantly and negatively related with riparian cover ($r^2 = 0.38$; $p = 0.01$) but were not significantly related with basin forest cover ($p = 0.5$).

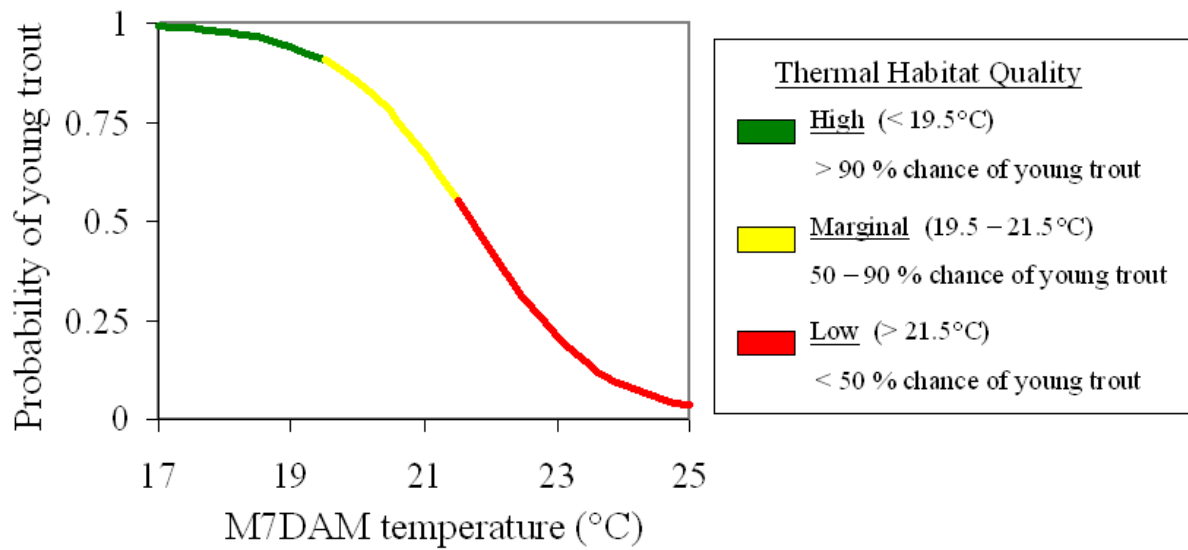


Figure 2.8. Logistic model relating the likelihood of young trout presence to observed M7DAM stream temperatures (°C) for Group 1 sites sampled for trout in 2001 and resampled in 2002. ($p < 0.0001$). From this relationship, we derived three categories of M7DAM temperatures indicative of thermal habitats supporting trout populations.

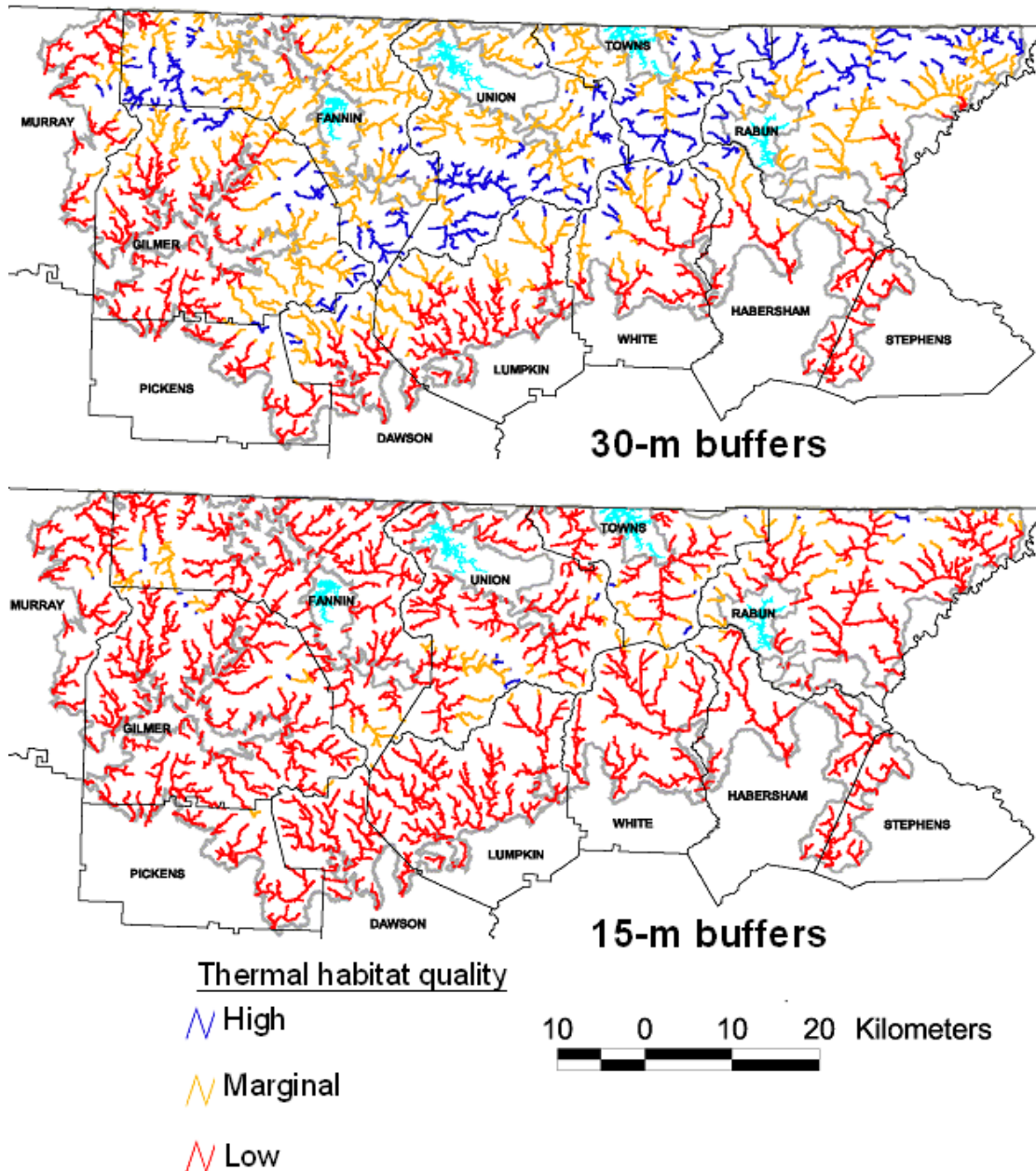
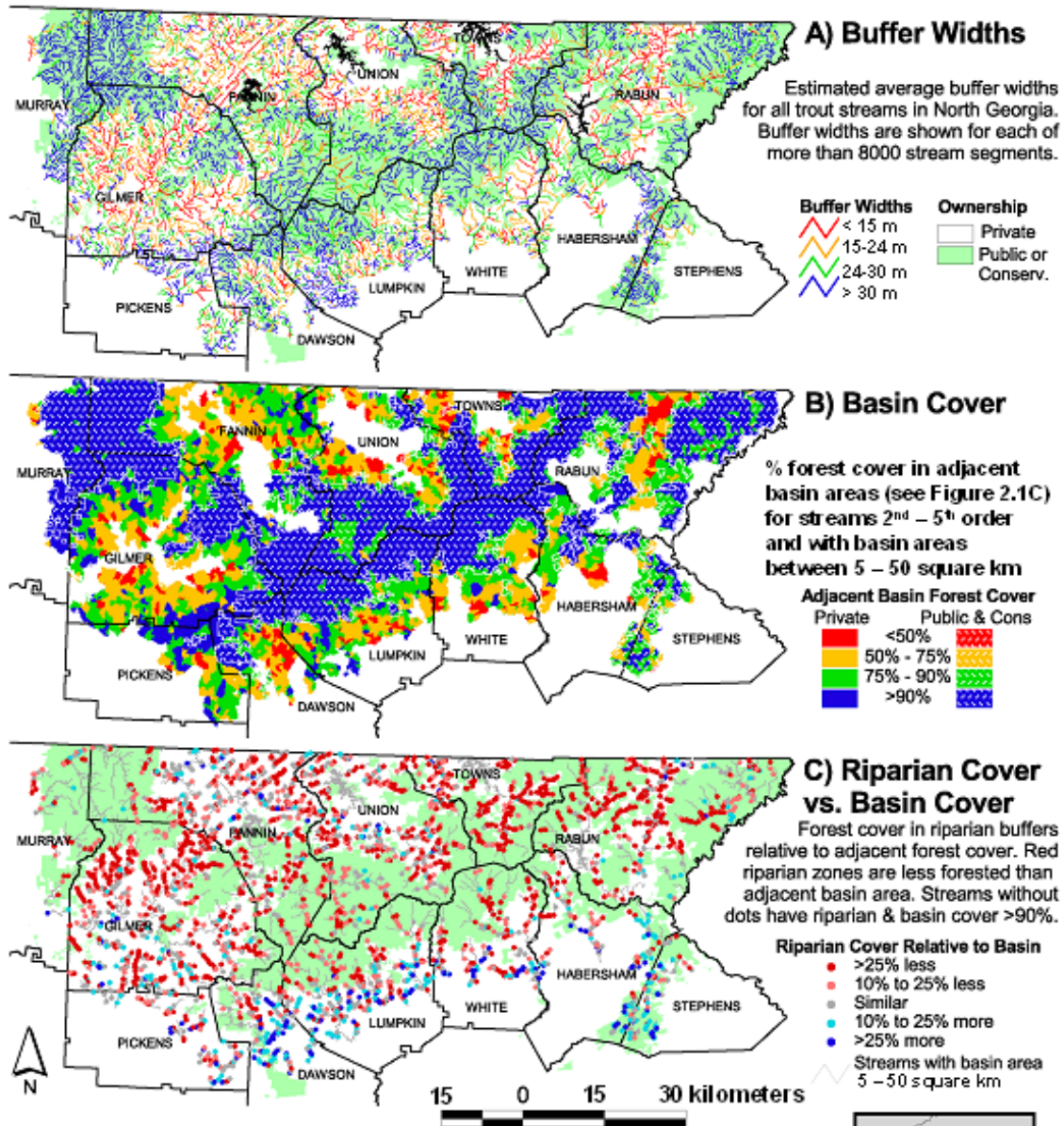


Figure 2.9. Distribution of M7DAM temperatures and associated thermal habitat quality expected with 30- and 15-m buffers across the Georgia trout stream network. M7DAM temperatures were calculated using Equation 1—2001 (an intermediate summer) with 30- versus 15-m buffer widths. Thermal habitat qualities were classified based on M7DAM temperatures indicative of habitats supporting young trout (Figure 2.8).



Riparian Buffer Conditions on Trout Streams of North Georgia

Riparian buffer widths and forest cover in the adjacent basin for North GA trout streams, by county and land ownership. Forest cover data are derived from satellite imagery (Landsat) collected in 1998.



Figure 2.10. Existing riparian buffer widths and forest cover in adjacent basins (Figure 2.1) of North Georgia trout stream segments by land ownership. Forest cover data were derived from satellite imagery (Landsat 1998).

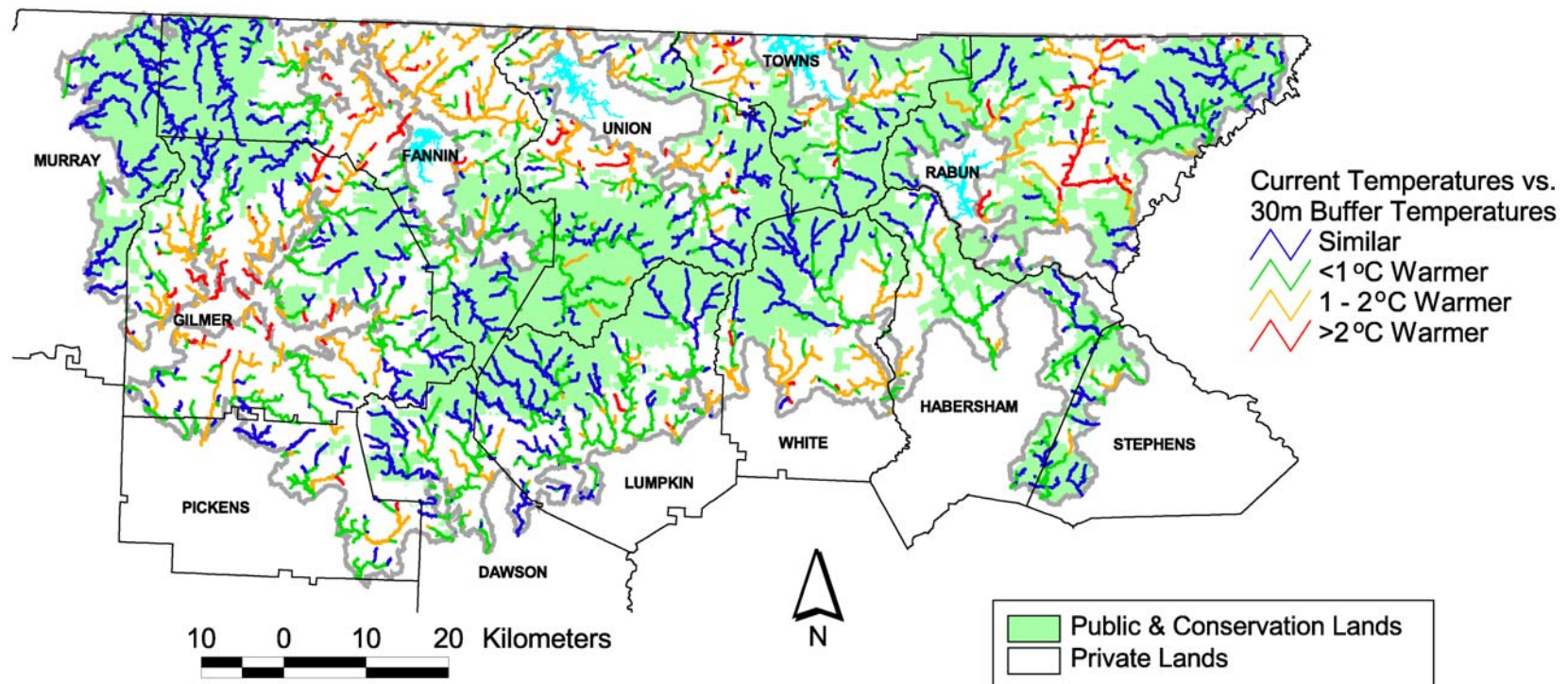


Figure 2.11. Existing levels of trout stream thermal degradation. Summer M7DAM temperatures were estimated for current landscape conditions relative to temperatures expected with 30-m buffers. M7DAM temperatures were estimated using Equation 1—2002.

CHAPTER 3

RIPARIAN BUFFERS, STREAM TEMPERATURE, AND FISH ASSEMBLAGES IN THE
SOUTHERN APPALACHIAN REGION OF NORTHERN GEORGIA, USA¹

¹ Jones, K.L., M.C. Freeman, G.C. Poole, and J.L. Meyer. To be submitted to *Southeastern Naturalist*.

ABSTRACT

The Georgia State Legislature reduced the mandatory-forested riparian buffer width along its Southern Appalachian trout streams from 30 to 15 m in order to accommodate growing human population and development pressures on private lands. To determine the impact of this narrower riparian buffer width on fish assemblages in these streams, we: 1) identified 28 sampling sites representative of trout streams in Georgia's Blue Ridge physiographic province; 2) quantitatively sampled instream habitat conditions and fishes at these sites; 3) investigated the relationships between fish assemblages and instream habitat variables; 4) examined variation in fish assemblages among sites and habitat and landscape gradients; and 5) related habitat-scale predictors of fish assemblages to landscape conditions. Information-theoretic and non-metric multidimensional scaling analyses supported that as stream temperatures increase, fishes that are benthic spawners and restricted to upland areas are likely to be adversely affected. Riparian forest cover drives stream temperature conditions in Northern Georgia. Thus, thermal alteration of trout streams will occur with further deforestation of riparian areas. Alteration of instream habitat via thermal warming will likely increase the similarity in stream temperature conditions between highland and lowland streams. This homogenization of instream habitat will potentially encourage invasion by lowland fishes while decreasing the suitability of these habitats for fishes unique to high- and upland streams of the Southeastern United States.

Introduction

Human population and suburban development pressures are increasing in the Southern Appalachian Mountains, particularly within the riparian areas of the region (Wear and Bolstad 1998). With these pressures come reduced forest cover and increased urbanization. Both

deforestation and urbanization are linked with warmer stream temperatures, increased sediment delivery, altered stream channels, and nutrient enrichment (Waters 1995; Jones et al. 1999; Scott et al. 2002; Snyder et al. 2003; Jones 2004). In turn, these alterations in stream habitat are related to shifts in fish assemblages as fishes with specialized habitat requirements and ecologies are replaced by generalist fishes (Weaver and Gorman 1994; Harding et al. 1998; Scott and Helfman 2001; Snyder et al. 2003; Walters et al. 2003).

As disturbance of the landscape increases in the Southern Appalachians, the potential for homogenization of the fish fauna also increases (Scott and Helfman 2001). This highlands ecoregion is part of the larger Southern United States region, which supports the highest diversity and endemism of temperate freshwater fishes in the world (Warren et al. 2000). The Southern United States has over 600 native freshwater fishes, and 37 of the region's 51 drainages have at least one unique fish taxon (Warren et al. 2000). Fishes of the Southern Appalachian highlands are characterized as being intolerant of fine sediments and preferring cold water, whereas fishes in the region's lowlands are characterized as being tolerant of fine sediment and warmer temperature conditions (Scott and Helfman 2001). Also, highland fishes tend to spawn directly over streambeds or within rock crevices. Because of their limited ranges, specialized habitat requirements, and benthic spawning behavior, these highland fishes are vulnerable to instream habitat alterations associated with land development and conversion (Etnier 1997; Warren et al. 1997; Warren et al. 2000). As instream habitats are degraded, highland fishes are being replaced by native lowland fishes leading to the homogenization of the region's fish fauna (Scott and Helfman 2001).

The Georgia State Legislature reduced the mandatory forested riparian buffer width along Southern Appalachian trout streams from 30 to 15 m to accommodate growing human population and development pressures. If narrower riparian buffer requirements result in continued disturbance of riparian forests, then warmer stream temperatures and increased fine sediment are predicted to degrade habitats necessary to support self-sustaining trout populations (Chapter 2). Like trout, we expect fishes unique to the high- and upland streams of Southern Appalachians to be adversely affected by increased riparian deforestation because of their benthic spawning behaviors and sensitivity to thermal conditions.

To determine the impact of narrower riparian buffers on fish assemblages in Southern Appalachian trout streams, we: 1) identified sampling sites representative of streams in Georgia's Blue Ridge physiographic province; 2) quantitatively sampled instream habitat conditions and fishes at these sites; 3) investigated the relationships between fish assemblages and instream habitat variables that have been demonstrated to be strong predictors of fish assemblages in the Southeastern United States; 4) examined variation in fish assemblages among sites and also habitat and landscape gradients; and 5) related habitat-scale predictors of fish assemblages to landscape conditions.

Methods

Study landscape and site selection

This research focused on the State of Georgia's trout stream network, which is designated by the Georgia Department of Natural Resources (DNR; England 1998; Georgia Environmental Protection Division 2004). The study landscape is located within the Blue Ridge physiographic province of Northeast Georgia, U.S.A. and includes the Chattahoochee, Coosa, Savannah, and Tennessee rivers (Figure 3.1).

We developed a Geographic Information System (GIS) database containing landscape attributes for the trout stream network. The trout stream network was divided into stream segments between channel junctions (Chapter 2). Stream order (Strahler 1952) was calculated for the ~8,000 trout stream segments. Basin area, elevation, and % forest cover were determined for segments 2nd order or larger. Percent forest cover was derived from Landsat 1998 data and measured for the riparian buffer (a 30-m zone on each side of the stream and along the entire stream network above a segment) and the basin (excluding the riparian buffer). See Chapter 2 for further details on the trout stream GIS database.

Using the landscape data available in the GIS database, we identified sampling sites representative of the landscape conditions across the trout stream network. First, we selected a pool of potential sampling sites by extracting all stream segments from the GIS database that were a) 3rd order or greater; b) > 1 km in length; and c) drained basins between 5 and 50 km² (n = 413). Based on a cluster analysis of these potential sites (Figure 3.2), 28 sampling sites were chosen that represent the variation in basin area, elevation, and riparian forest cover observed for streams in Northeast Georgia (Table 3.1). We chose a set of sites (n = 28) that lacked autocorrelation between basin area, elevation, and riparian forest cover values (Figure 3.3) and

were located in separate watersheds to ensure sampling independence. Percent riparian forest cover, however, was autocorrelated with basin forest cover (Figure 3.3).

Habitat sampling

At each of 28 sites, we designated a 50-m sampling reach where we collected instream habitat and fish data during summer 2001. Onset Corp. ® HOBO temperature data loggers installed at each site continuously monitored stream temperature at thirty minute intervals. From these data, we calculated the average of daily maximum stream temperatures during the warmest 7-day period of the summer (M7DAM temperature, °C) as a quantitative indicator of peak temperature conditions experienced by fishes.

Substrate conditions were evaluated following the Wolman method (Wolman 1954); we walked in a zigzag pattern throughout the study reach and measured the intermediate axis of 100 randomly chosen particles. From these Wolman pebble count data, we calculated the proportion of coarse substrate (particles greater than 64 mm in diameter) to capture the amount of coarse sized particles available within a reach for foraging, refugia, and spawning activities (Waters 1995).

Local stream slope was measured over 100 m, which included the 50-m reach and 25 m above and below the reach. These values are reported as percent slope over 100 m. 50 depth and velocity measurements were taken randomly across each study reach to capture the range of depth and velocity conditions. Velocity was measured using a Marsh-McBirney Flo-Mate 2000.™

Fish sampling

For quantitative catch data, all habitats within the 50-m reach were sampled extensively in a single pass with a backpack electric shocker, dip nets, and seine net. Extended sampling outside of the 50-m reach was then conducted to detect the presence of additional fish species. Overall, we sampled an area approximately 30 times average stream width at each site (range sampled: 88 – 440 m). Trout were identified to species (rainbow, *Oncorhynchus mykiss*, or brown trout, *Salmo trutta*), classified as naturalized or stocked based on fin condition and coloring, measured for total length (mm), and released. All other fishes (excluding young-of-the year) were euthanized, preserved for identification, and archived at the Georgia Natural History Museum.

Since the fish species collected at a sampling site varied by river drainage, species were assigned to spawning guilds and biogeographic designations (Table 3.2) to minimize a potential drainage bias in the statistical analyses. Spawning guilds were designated following Scott (2001), Sutherland et al. (2002), and Walters (2002). For species without reported spawning guilds, we determined the appropriate guild from life history information found in Etnier and Starnes (1993) and Jenkins and Burkhead (1994). Range distributions were assigned following Mayden (1987), Scott (2001), and Walters (2002).

Both rainbow and brown trout are introduced game species in Northeast Georgia. Since larger trout (year class 1 and higher) are stocked by Georgia Wildlife Resources Division and are harvested by anglers, their age class structure is an unreliable indicator of habitat conditions in a given stream. In the following analyses, therefore, we only included naturalized young-of-the-year (< 150 mm in length) trout, which are smaller than stocked trout, below harvestable length, and are indicators of reproducing trout populations.

Statistical models

Field data were analyzed with two statistical methods to determine the relationships between fish assemblages, instream habitat, and landscape conditions in Northeast Georgia. Prior to analyses, all variables were tested for normality with the Shapiro-Wilk goodness of fit test (JMP Version 5.0, SAS Institute Inc., Cary, NC, USA) and were transformed when necessary. Log transformations were applied to local stream slope and basin area, whereas the arcsine squareroot transformation was applied to % riparian cover, % catchment cover, proportion of coarse substrate, and relative abundance variables.

We investigated the relationships between fish assemblage metrics and habitat variables known to be strong predictors of fish assemblages in the Southeastern United States following the information-theoretic statistical approach (Burnham and Anderson 2002). An *a priori* global model was identified based on published research on fishes in the Southeastern United States. Habitat-scale predictors within the global model were stream temperature (Scott 2001), streambed substrate composition (Scott 2001, Sutherland et al. 2002, Walters et al. 2003), and local stream slope (Walters et al. 2003). For global models with good fit (i.e., adjusted $r^2 > 0.25$), relative support for models using a subset of the predictor variables was determined with the Akaike's Information Criterion corrected for small sample size (AIC_c ; Burnham and Anderson 2002). This AIC_c statistic is useful for selecting the best-supported candidate model from a set of models (i.e., the best-supported model has the lowest AIC_c statistic). Also, Akaike weights (w_i) were calculated to determine the weight of evidence for each candidate model. We examined four fish response variables: the relative species richness and abundance of benthic spawners (benthic crevice and gravel spawners) and upland fishes (high- and upland species).

We assessed variation in the relative abundances of separate spawning guilds (Table 3.2) across sites and across habitat and landscape gradients with non-metric multidimensional scaling (NMDS) analysis (PC-ORD 4.1, MjM Software Design™, Glendale Beach, OR, US). For this analysis, the relative abundance of young trout was included as separate spawning guild. This indirect gradient analysis calculates Bray-Curtis distances between sites and clusters sites with similar spawning guild composition (McCune and Grace 2002). Axes representing a large proportion of the variation within the data were overlaid with species data to determine what individual species were potentially driving relationships and, then, were correlated with habitat and landscape variables.

Results

Variation in stream habitat and fishes

Across the sampling sites, we observed variation in M7DAM temperatures, proportions of coarse substrate, local stream slope, maximum reach depth, and maximum reach velocity (Table 3.1). We collected 37 species (5,329 individual fish) representing 8 fish families. The family Cyprinidae made up 51.8% of the total catch. Other families included Cottidae (34.3%), Salmonidae (6.7%), Catostomidae (3.3%), Percidae (2.0%), Centrarchidae (1.3%), Petromyzontidae (0.5%), and Ictaluridae (0.2%). *Cottus spp.* were the single most abundant taxa and was collected at 20 of the 28 sampling sites. Species richness per site ranged from 2 to 14 species. Relative species richness and abundance metrics for benthic spawners and upland fishes were autocorrelated (both Pearson coefficients > 0.70 and p values < 0.001), which was not surprising, as upland fishes tend to be benthic spawners.

Relationships between fish assemblages and habitat-scale predictors

The global model combining M7DAM temperatures, proportion of coarse substrate, and local stream slope explained a substantial amount of the variation in relative species richness for both benthic spawners and upland species (Table 3.3). The M7DAM temperature model was the best-supported candidate model (Table 3.4). The relative species richness of both benthic spawners and upland fishes negatively correlated with M7DAM temperatures (Figure 3.4). As stream temperatures increased, benthic spawning fishes with limited ranges were likely to be adversely affected.

Variation in fish assemblages among sites and across habitat and landscape gradients

Axes 1 and 2 of the NMDS represented 81% of the variation in the relative abundance of spawning guilds across the sampling sites. Each axis represented relatively equal proportions of the variance. In the ordination, benthic nest builders and benthic excavators were located in the bottom left, young trout in the bottom right, and benthic crevice spawners in the top center (Figure 3.5). An overlay of species data showed that *Nocomis leptocephalus* (benthic nest builder), *Lepomis macrochirus* (benthic excavator) and *Oncorhynchus mykiss* (young trout) drove Axis 1 while *Cottus spp.* (benthic crevice spawner) drove Axis 2. Axis 1 was negatively correlated with M7DAM temperatures (Pearson coefficient $r = -0.680$). All other landscape and habitat variables had Pearson coefficients < 0.35 . Axis 2 was not related to any habitat or landscape variables.

Relationships between stream temperature and landscape conditions

The information-theoretic and NMDS analyses supported M7DAM stream temperature as a significant predictor of the relative species richness and abundance of both benthic spawners and upland fishes. Results from Chapter 2 demonstrate that for this study landscape: 1) M7DAM temperatures were consistently and negatively correlated with riparian forest cover and elevation; and 2) riparian forest conditions overwhelm the influence of basin forest conditions and, thus, drive stream temperature (Figure 3.6). As deforestation of riparian areas continues across this study landscape, the thermal alteration of streams will occur. Alteration of stream habitat via thermal warming increases the similarity in stream temperature conditions between highland and lowland streams. Thus, the alteration of thermal conditions within these streams will likely enhance the suitability of highland streams for lowland fishes that are benthic modifiers and wide ranging while decreasing the suitability of these habitats for fishes that are benthic spawners and range restricted.

Discussion

The strength of stream temperature as a predictor of fishes

Our analyses reveal that as stream temperature increases, the relative species richness of fishes that tend to be both benthic spawners and range-restricted declines and that fish assemblages shift from being dominated by young trout to lowland fishes that are benthic modifiers (e.g., *N. leptocephalus* and *L. macrochirus*). Stream temperature is a function of riparian forest cover. Thus, upland fishes will be adversely impacted by increased riparian deforestation, which is permitted with the new 15-m buffer width regulation.

Stream temperature is likely a strong predictor of fishes because this study landscape is the southernmost terminus of the Southern Appalachian Mountains. Under a full canopy, streams are shaded and can maintain cool stream temperatures despite the potential for substantial solar heating of streams during summer months. With deforestation of the riparian zone, stream temperatures increase, leading to the alteration of instream thermal conditions. With warmer stream temperatures, habitats are less conducive for upland fishes and better suited for lowland and cosmopolitan fishes. Also, under a full canopy, we would expect stream temperature to be cooler at higher elevations and, then, gradually warm as streams flow into lower elevation areas. Riparian deforestation, however, can substantially increase stream temperatures so that high elevation streams are warmer than low elevation streams. The warming of high elevation streams could further reduce and fragment habitats suitable for these upland fishes unique to the Southeastern United States.

Research on fishes in the Southeastern United States

Much research in the Southeastern United States has focused on relating fish assemblages to habitat and landscape conditions in order to understand distributions of fishes across the region and how landuses may impact these fishes. We quantitatively sampled habitat variables identified as significant predictors of fish assemblages by prior research on Southeastern fishes. Like Scott (2001), we found stream temperature to be a strong predictor of fishes. Unlike Scott (2001) and Walters et al. (2003), we did not detect significant relationships between fish assemblages and instream habitat variables such as streambed particle composition and local stream slope. Proportions of coarse substrate ranged from 18 to 78% for our sites, which is similar to the range of 9 to 80% reported by Scott (2001). Also, local stream slope conditions

ranged from 0.2 to 3.3% for our sites versus 0.001 to 0.01% for sites sampled by Walters et al. (2003). Stream temperature may be masking any influence of streambed particle composition on the region's fish fauna because our study landscape is located on the southernmost edge of the Southern Appalachian Mountains, whereas the area studied by Scott (2001) was located further north. Also, because of the abundance of stream habitats with coarse streambeds in our study landscape, fishes requiring coarse-sized sediment particles for spawning and foraging habitat are less likely to be limited to higher gradient patches in this system compared to fishes in the lower slope system studied by Walters et al. (2003).

Unlike other studies of Southeastern fishes, however, we did not restrict our sampling to one river drainage. Instead, our underlying sampling design was intended to capture the range of trout habitat across elevation and riparian forest cover gradients in streams between 5 to 50 km² in basin area. The trout stream network crosses the Chattahoochee, Coosa, Savannah, and Tennessee river drainages. The Coosa and Tennessee river systems are more species rich than the Chattahoochee and Savannah river systems (Warren et al. 1997). Because of the low number of sampling sites located within individual drainages, we cannot test for significant differences in fish response variables among drainages. In the spawning guild ordination, however, the drainages tend to overlap (Figure 3.5), which suggests that this analysis is less sensitive to species differences among the drainages. To more fully address questions of native fish response to landscape disturbances and instream habitat alterations (e.g., Scott 2001; Walters 2002), the research design should include samples from a range of sites within the river drainages of Northeast Georgia. Our study landscape spans the area between the species-rich and rapidly developing Etowah system studied by Walters (2002) and the Tennessee and French Broad

systems studied by Scott (2001). Understanding the linkages between landscape disturbance, instream habitat, and fishes is necessary to inform management decisions regarding stream ecosystems and aquatic fauna as human population and development pressures increase across the Southeastern United States.

Conclusions

As riparian forest cover declines along streams in this study landscape, the potential for stream habitat degradation increases. In particular, the reduction of the mandatory riparian buffer width along Northeast Georgia streams allows for continued riparian deforestation, which will increase stream temperatures. As stream temperatures warm and stream habitats across the stream network are homogenized, our data and those of others suggest that the fish assemblage of Northeast Georgia streams may shift from being dominated by the region's unique upland and benthic-spawning fishes to cosmopolitan and benthic-modifying fishes.

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Table 3.1. Range of landscape, instream habitat, and fish response variables observed in streams representative of the trout stream network in Northeast Georgia.

	Units	Range
Landscape variables		
Basin Area	km ²	6 - 43
Elevation	m	396 - 740
% Riparian Forest Cover	%	29 - 100
% Catchment Forest Cover	%	73 - 99
Instream habitat variables		
2001 M7DAM temperature	deg C	17.6 - 24.5
Proportion of Coarse Substrate	%	18 - 78
Local stream slope	Change in m over 100 m	0.2 - 3.3
Maximum reach velocity	m/sec	31 - 123
Maximum reach depth	cm	0.44 - 2.2
Fish variables		
Fish Species Richness	#	2 - 14
<i>Relative species richness</i>		
Benthic spawners	%	0 - 100
Upland fishes	%	0 - 100
<i>Relative abundance</i>		
Benthic spawners	%	0 - 100
Upland fishes	%	0 - 100

Table 3.2. Data for species collected in the Chattahoochee, Coosa, Savannah, and Tennessee river drainages. Codes for spawning guilds: BC = benthic crevice spawner; G = gravel spawners; BNB = benthic nest builders; BNA = benthic nest associates; and BE = benthic excavators. Codes for range: H = highland fishes; U = upland fishes; L = lowland fishes; C = cosmopolitan fishes. Spawning guilds were designated following Scott (2001), Sutherland et al. (2002), and Walters (2002). For species without reported spawning guilds, appropriate guild was determined from life history information found in Etnier and Starnes (1993) and Jenkins and Burkhead (1994). Range distributions were assigned following Mayden (1987), Scott (2001), and Walters (2002).

Family name	<i>Scientific name</i>	Common name	Spawning guild	Range
Petromyzontidae				
	<i>Ichthyomyzon gagei</i>	southern brook lamprey	G	L
	<i>I. greeleyi</i>	mountain brook lamprey	G	U
Cyprinidae				
	<i>Campostoma sp.</i>	stoneroller	BNB	L
	<i>Clinostomus funduloides</i>	rosyside dace	BNA	L
	<i>Cyprinella galactura</i>	whitetail shiner	BC	U
	<i>Luxilus coccogenis</i>	warpaint shiner	BNA	H
	<i>L. zonistius</i>	bandfin shiner	BNA	L
	<i>Nocomis leptcephalus interocularis</i>	bluehead chub	BNB	L
	<i>N. micropogon</i>	river chub	BNB	L
	<i>Notemigonus crysoleucas</i>	golden shiner	BE	C
	<i>Notropis leuciodus</i>	Tennessee shiner	BNA	H
	<i>N. lutipinnis</i>	yellowfin shiner	BNA	L
	<i>N. spectrunculus</i>	mirror shiner	G	H
	<i>Rhinichthys atratulus</i>	blacknose dace	G	L
	<i>R. cataractae</i>	longnose dace	G	U
	<i>Semotilus atromaculatus</i>	creek chub	BNB	C
Catostomidae				
	<i>Catostomus commersoni</i>	white sucker	G	L
	<i>Hypentelium etowanum</i>	Alabama hogsucker	G	L
	<i>H. nigricans</i>	northern hogsucker	G	L
	<i>Scartomyzon rupiscartes</i>	striped jumprock	G	U

Family name	<i>Scientific name</i>	Common name	Spawning guild	Range
Ictaluridae				
	<i>Ameiurus brunneus</i>	snail bullhead	BE	L
Salmonidae				
	<i>Onchorhynchus mykiss</i>	rainbow trout	G	U
	<i>Salmo trutta</i>	brown trout	G	U
Cottidae				
	<i>Cottus spp.**</i>	sculpin	BC	U
Centrarchidae				
	<i>Ambloplites rupestris</i>	rock bass	BE	L
	<i>Lepomis auritus</i>	redbreast sunfish	BE	C
	<i>L. cyanellus</i>	green sunfish	BE	C
	<i>L. macrochirus</i>	bluegill	BE	C
	<i>Micropterus coosae</i>	red eye bass	BE	U
	<i>M. dolomieu</i>	smallmouth bass	BE	C
	<i>M. salmoides</i>	largemouth bass	BE	C
Percidae				
	<i>Etheostoma blennioides</i>	greenside darter	G	U
	<i>E. rufilineatum</i>	redline darter	G	H
	<i>Perca flavescens</i>	yellow perch	G	C
	<i>Percina evides</i>	gilt darter	G	H
	<i>P. nigrofasciata</i>	blackbanded darter	G	L
	<i>P. palmaris</i>	bronze darter	G	L

***Cottus spp.* included *Cottus bairdi* (mottled sculpin) and *Cottus carolinae* (banded sculpin)

Table 3.3. Results from global model analyses following the information-theoretic statistic method. Candidate models were further analyzed for global model fits that explained a large amount of variance within the data (i.e., adjusted $r^2 > 0.25$).

	Response variable	Adj. R^2
Relative species richness		
	Benthic spawners	0.446
	Upland fishes	0.444
Relative abundance		
	Benthic spawners	0.216
	Upland fishes	0.09

Table 3.4. Candidate models results for the relative species richness global models for benthic spawners and upland fishes. See the Methods section and Burnham and Anderson (2000) for descriptions of statistics used in the information-theoretic analyses.

	Adj. R^2	SSE	K	AIC	AIC _c	w_i
Relative species richness (benthic spawners)						
CFP + slope	-0.01	1.01	4	-32.444	-30.7	0.00
CFP	-0.035	1.06	3	-33.781	-32.78	0.01
Slope	0.004	1.02	3	-34.25	-33.25	0.01
M7DAM temp + slope	0.463	0.53	4	-40.252	-38.51	0.19
M7DAM temp + CFP	0.464	0.53	4	-40.278	-38.54	0.19
M7DAM temp	0.463	0.55	3	-41.769	-40.77	0.59
Relative species richness (upland fishes)						
CFP + slope	-0.014	1.07	4	-31.687	-29.95	0.00
CFP	0.004	1.09	3	-33.441	-32.44	0.01
Slope	0.022	1.07	3	-33.661	-32.66	0.02
M7DAM temp + CFP	0.445	0.59	4	-39.037	-37.3	0.16
M7DAM temp + slope	0.455	0.57	4	-39.268	-37.53	0.18
M7DAM temp	0.466	0.59	3	-41.036	-40.04	0.63

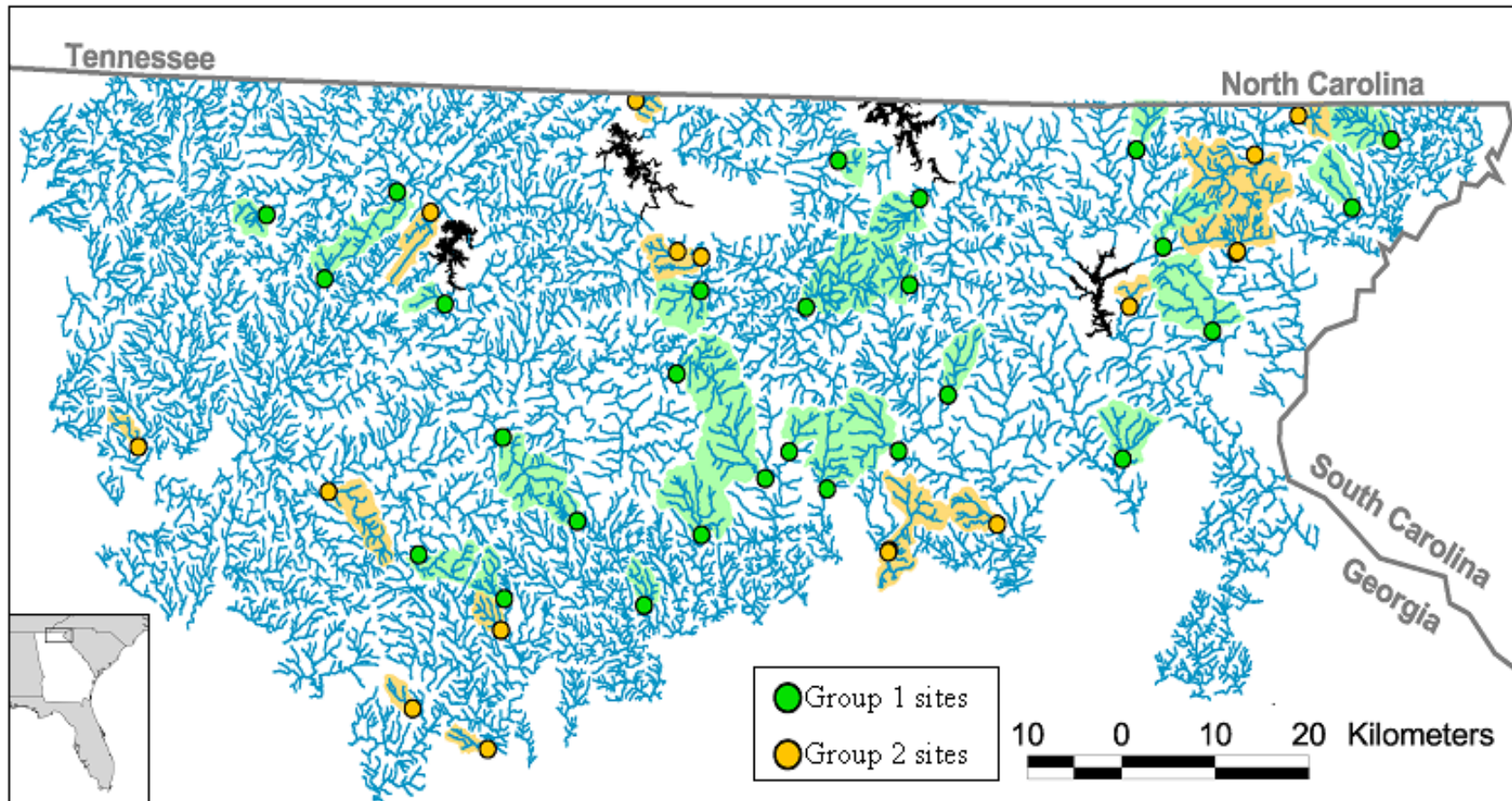


Figure 3.1. Study landscape and sampling sites located in Northeast Georgia. Field data collected at Group 1 sites was used for analyzing the impact of reduced riparian buffers on fish assemblages in Southern Appalachian streams (Chapter 3).

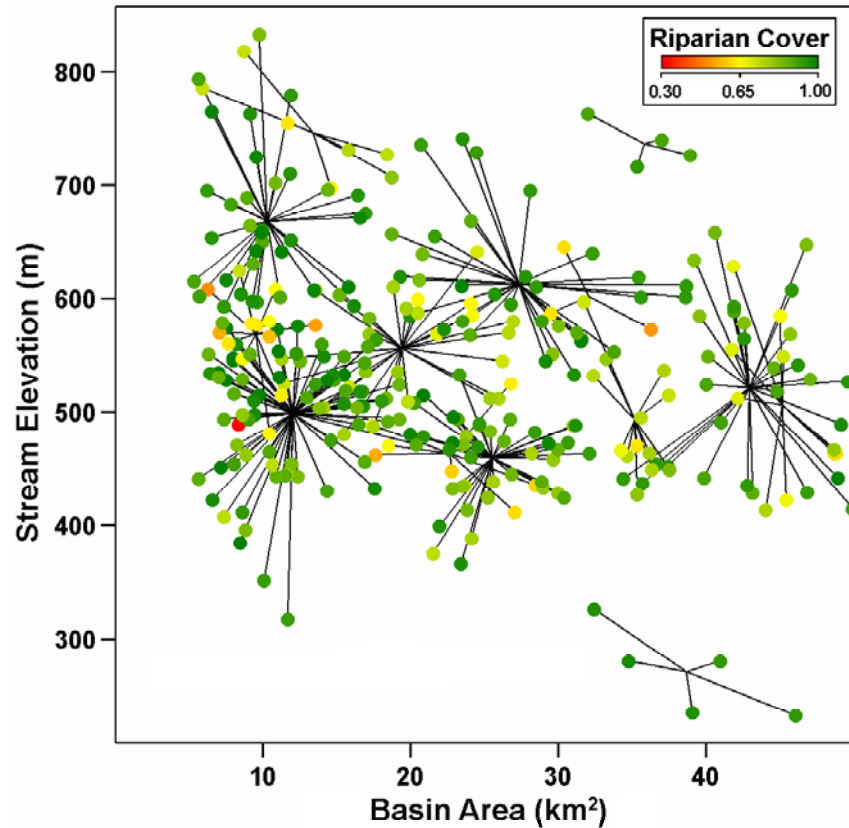


Figure 3.2. Results from a cluster analysis of potential sampling sites using basin area, elevation, and % riparian forest cover. Dots represent characteristics of individual potential sampling sites. Lines group potential sampling sites into groups (clusters) where sites within each group are more similar to each other than to sites in other groups. Sampling sites were chosen proportionally from each group so that the final dataset was representative of the range of streams across the stream network.

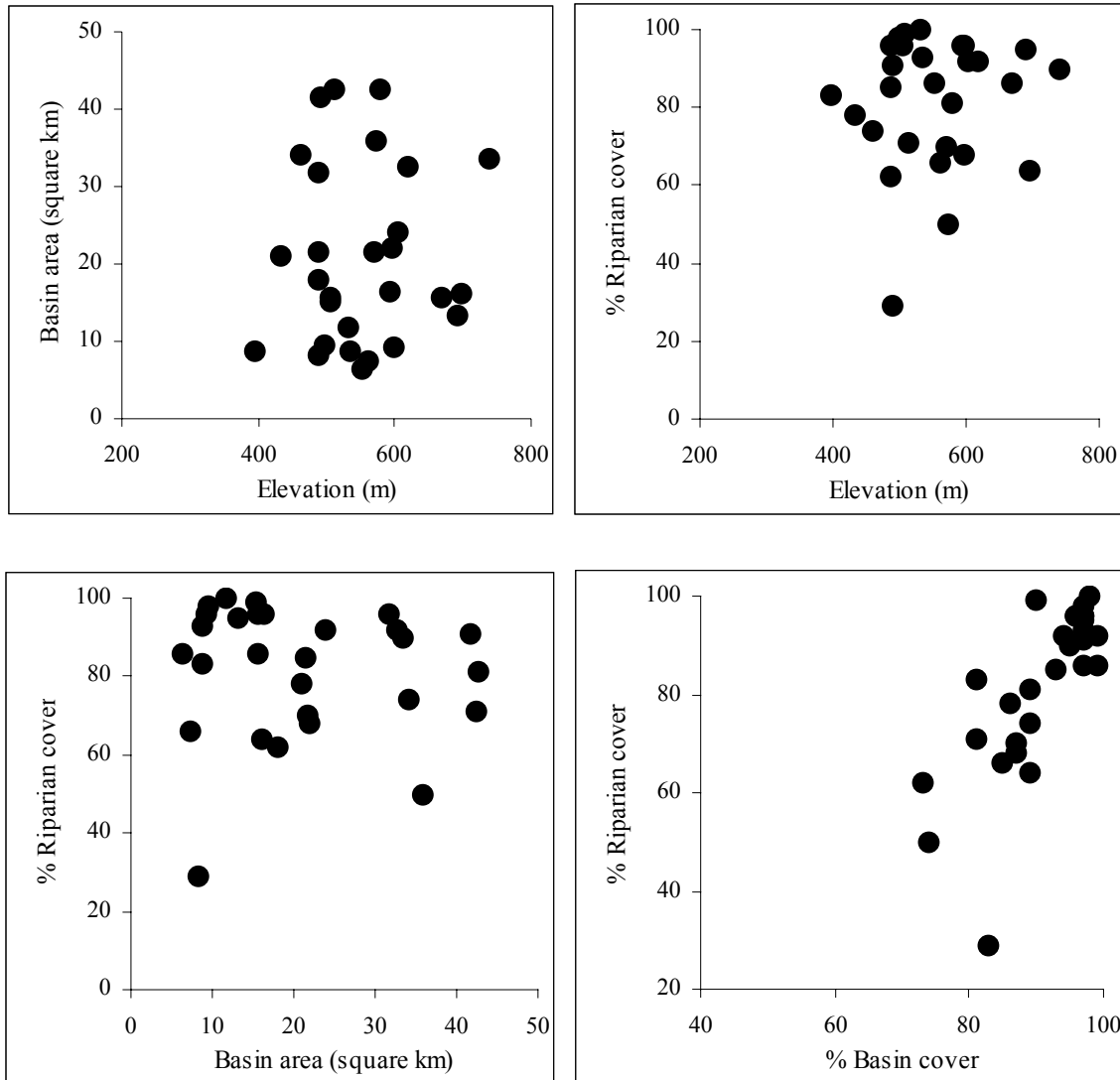


Figure 3.3. Plots of basin area, elevation, and % riparian forest cover against one another illustrate the range of sampling site characteristics and lack of autocorrelation between these independent variables within the dataset. Values for % riparian forest cover and % basin forest cover, however, are significantly correlated ($r^2 = 0.59$; $p < 0.0001$).

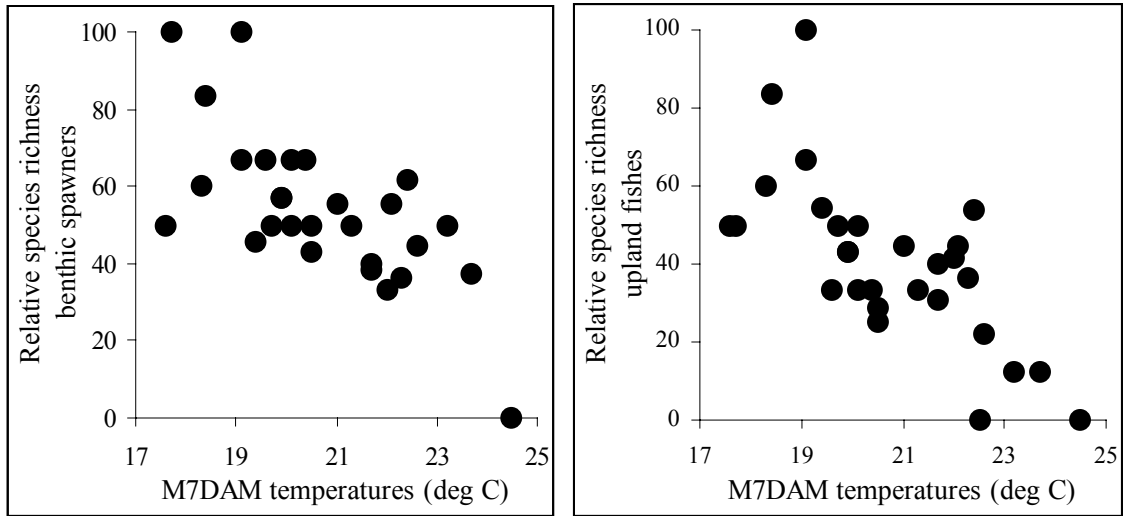


Figure 3.4. Relative species richness for benthic spawners and upland fishes plotted against M7DAM stream temperatures (°C). Both metrics were negatively correlated with M7DAM stream temperatures.

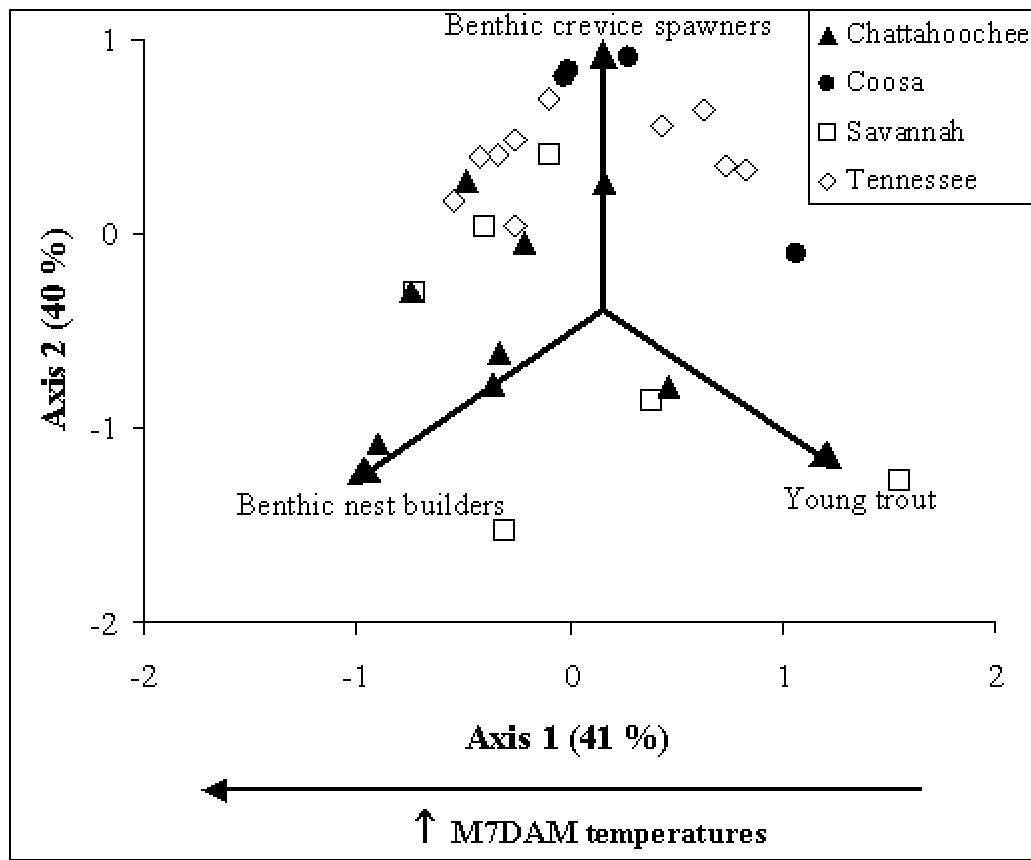


Figure 3.5. Results from NMDS ordination of the relative abundances of benthic crevice spawners, gravel spawners, benthic nest builders, benthic nest associates, benthic excavators, and young trout.

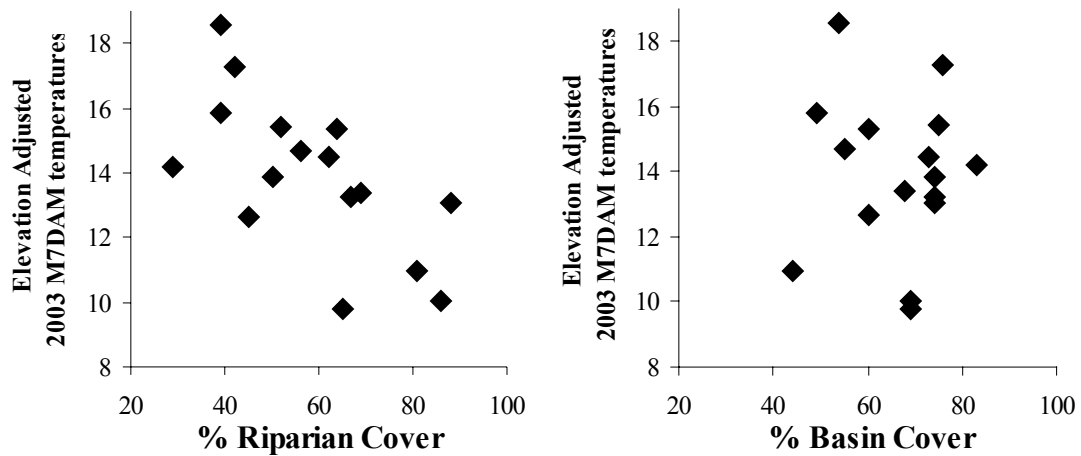


Figure 3.6. Field data from the study landscape of Northeast Georgia that demonstrates riparian forest cover is the driver of stream temperatures. M7DAM temperatures were algebraically to account for the influence of elevation on stream temperature and then plotted against % riparian and basin forest cover. These elevation—adjusted temperature values were significantly and negatively related with riparian cover ($r^2 = 0.38$; $p = 0.01$) but were not significantly related with basin forest cover ($p = 0.5$). See Chapter 2 for further details on this analysis.

CHAPTER 4

CONCLUSIONS

In Chapter 2, I assessed the capacity of Georgia's mountain trout streams to maintain trout populations with 30- versus 15-m wide riparian buffers. Quantitative analyses demonstrated that, across the trout stream network, only 1 trout stream in 10 is likely to support self-sustaining trout populations with 15-m wide riparian buffers. Furthermore, these streams likely to support trout populations are limited to the smallest and highest-elevation streams in the trout stream network; few 2nd order streams and no 3rd – 5th order streams are likely to support self-sustaining trout populations when associated riparian buffer are 15-m wide. Reducing the width of forested riparian buffers increases both stream temperatures and fine sediments in riffle habitats. As a consequence of warmer stream temperatures and increased fine sediments, the biomass of young-of-the-year trout is reduced by over 80%. Forest cover within the riparian buffer is a dominant influence on stream temperature and streambed sediment conditions. Thus, reductions of riparian forests permitted with the 15-m buffer regulation will substantially increase stream temperatures and the abundance of fine sediment. The ability of the Georgia trout stream network to support reproducing trout populations requires the availability of instream habitats with cool temperatures and clean streambeds. Maintaining riparian forest cover is an important management component for protecting these instream habitat conditions necessary for self-sustaining trout populations in Northern Georgia.

Also, in Chapter 2, I found that riparian forest cover and maximum reach velocity were significant predictors of fine sediment conditions in riffle habitats (as measured with riffle embeddedness). Although riparian and basin forest cover for the Group 1 dataset are correlated, basin forest cover was not a significant predictor of fine sediment conditions in riffles alone or in combination with any habitat or landscape variable. While streams with high riparian forest cover tend to have high basin forest cover in this study landscape, riparian forest cover is generally lower than basin forest cover. Deforestation, particularly in riparian areas, can increase the supply and delivery of sediment particles to streams (Knighton 1998). The preferential development of low gradient riparian areas is projected to continue in the region (Wear and Bolstad 1998). As riparian forest cover declines and urban land cover increases in these Southern Appalachian catchments, the potential for overland sediment flow to break through or bypass riparian areas will likely increase. Thus, the strength of the relationships that we observed between fine sediment conditions and riparian forest cover may change over time as increasing deforestation is coupled with urbanization.

In Chapter 3, I investigated the potential impact of the narrower riparian buffer width on the fish assemblages supported by Southern Appalachian streams in Northern Georgia. Fish assemblage metrics were related to: 1) instream habitat variables, which prior research had demonstrated to be strong predictors of fish assemblages in the Southeastern United States; and 2) habitat and landscape gradients. Both information-theoretic and non-metric multidimensional scaling analyses supported that as stream temperatures increase, fishes that are benthic spawners and range-restricted are likely to be adversely affected. Since riparian forest cover drives stream temperatures in this study landscape, stream temperatures will warm as a result of increased

riparian deforestation permitted with the reduced buffer width. Thermal warming is one potential alteration of instream habitat conditions that will increase the similarity between highland and lowland streams. This homogenization of instream habitats will likely decrease the amount of habitat suitable for fishes unique to high- and upland Southeastern streams while increasing the amount of habitat suitable for native lowland fishes and also introduced cosmopolitan fishes.

While this investigation of the potential impacts of reduced riparian buffer widths on Southern Appalachian fish assemblages is limited by its underlying sampling design, which involved sampling across four major river drainages, it provides a first glimpse into the study landscape's fish fauna and potential factors contributing to the homogenization of this fauna. These results suggest that instream habitat alterations via thermal warming influence fish assemblages in these potentially cold-water streams. Homogenization of the Southeastern fish fauna is of concern as the region's landscape and aquatic systems are subject to mounting human population and development pressures (e.g., Walters et al. 2003, Scott and Helfman 2001, Warren et al. 2000). Deforestation of riparian areas is one of the many alterations of the landscape that occur with urbanization. Further research is required to investigate species-level fish assemblages shifts for the individual river drainages of the trout stream network and also to determine if additional habitat and landscape factors are influencing these assemblages as development of the landscape increases.

In conclusion, human population and suburban development pressures are increasing on this mountainous landscape through which these Southern Appalachian trout streams flow. Reductions in riparian buffer widths reflect increased opportunities for private land development at the cost of maintaining self-sustaining trout populations and the region's unique fish fauna.

Also, the intensifying development of private lands heralds the increasing urbanization of this forested mountain landscape. Overall, these growing pressures are likely to undermine the future capacity of these Southern Appalachian streams to support trout and the region's endemic fishes unless adequate and forward-looking policies encourage effective mitigation and management of landuse activities and ensure the protection of instream habitat. Proactive, watershed-level management of landuse activities in conjunction with research on the combined effects of increased deforestation, urban landuse, and stormwater infrastructure (see Wang et al. 2003) will be necessary in order to sustain these natural resources in Northern Georgia.

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APPENDICES

Appendix A, Table 1: County, drainage, and landscape data for Group 1 sampling sites. Group 1 sites denoted by ** were also included in the Group 2 sampling site dataset.

SiteID	Site Name	County	Drainage	Basin Area (km ²)	Elevation (m)	% Riparian Cover	% Basin Cover
1	Anderson	Gilmer	Coosa	16.4	594	96	96
2**	Boardtown	Gilmer	Coosa	8.2	489	29	83
3	Boggs	Lumpkin	Chattahoochee	11.8	531	100	98
4	Clay	Lumpkin	Chattahoochee	8.7	396	83	81
5	Cochrans	Dawson	Coosa	9.5	498	98	97
6	Coleman	Rabun	Savannah	13.2	691	95	97
7	Cooper	Union	Tennessee	33.5	740	90	95
8	Corn	Towns	Tennessee	7.4	561	66	85
9	Dick's	Lumpkin	Chattahoochee	41.6	491	91	97
10	Dukes	White	Chattahoochee	31.8	488	96	97
11	E Fork Smith	White	Chattahoochee	15.6	505	96	96
12	Flat	Fannin	Tennessee	6.4	551	86	99
13	Fodder	Towns	Tennessee	22.0	596	68	87
14	Holcomb	Rabun	Savannah	24.0	604	92	94
16	Little Fightingtown	Fannin	Tennessee	9.3	598	96	97
17	Noontootla	Fannin	Tennessee	32.6	618	92	99
19	Shoal	Habersham	Chattahoochee	20.9	434	78	86
20	Soapstone	Towns	Tennessee	15.5	668	86	97
21**	Stekoa	Rabun	Savannah	35.8	573	50	74
22**	Sugar	Fannin	Tennessee	18.0	488	62	73
23	Tiger	Rabun	Savannah	42.6	512	71	81
24	Timpson	Rabun	Savannah	16.1	697	64	89
25	Town (Union)	Union	Tennessee	42.6	579	81	89
26	Town (White)	White	Chattahoochee	21.5	488	85	93
27	Walnut Fork	Rabun	Savannah	15.3	507	99	90
28	West Fork Coosa	Union	Tennessee	21.6	569	70	87
29	West Fork Montgomery	Lumpkin	Coosa	8.8	534	93	97
30	Yahoola	Lumpkin	Chattahoochee	34.1	461	74	89

Appendix A, Table 2: County, drainage, and landscape data for Group 2 sampling sites.

SiteID	Site Name	County	Drainage	Basin Area (km ²)	Elevation (m)	% Riparian Cover	% Basin Cover
47	Amicalola	Dawson	Coosa	6.2	381	86	69
48	Brasstown	White	Chattahoochee	14.4	403	67	74
49	Bridge	Rabun	Savannah	6.5	585	42	76
50	Cochrans trib	Dawson	Coosa	8.4	451	81	44
51	East Fork Coosa	Union	Tennessee	8.8	544	39	49
52	Cox	White	Chattahoochee	9.0	425	66	52
53	Disharoon	Pickens	Coosa	6.5	463	88	74
54	Fir	Gilmer	Coosa	5.5	396	65	69
56	Kiutuestia	Union	Coosa	6.4	561	64	60
57	Licklog	Gilmer	Coosa	23.8	459	54	65
58	Little Tennessee	Rabun	Tennessee	45.6	643	52	75
59	Moccasin	Union	Tennessee	6.1	508	56	55
61	Mudd	Rabun	Tennessee	8.5	886	39	54
62	Tesnatee	White	Chattahoochee	25.4	425	69	68
63	Weaver	Fannin	Tennessee	14.6	473	45	60

Appendix A, Table 3: GPS coordinates for sampling sites. Note: SiteID 52 and 62 have the same GPS coordinates. Looking upstream, 52 is on the right and 62 is on the left.

SiteID	Latitude	Longitude
1	34.591958	84.298267
2	34.8223165	84.411525
3	34.6894085	83.892383
4	34.5556165	84.047925
5	34.5573335	84.2023835
6	34.9533415	83.51645
7	34.752275	84.01905
8	34.9368665	83.846642
9	34.6662	83.91815
10	34.6927165	83.7714585
11	34.741625	83.718742
12	34.8046665	84.2777
13	34.9069085	83.7551915
14	34.967325	83.2341165
16	34.874625	84.4778915
17	34.6937665	84.209058
19	34.6913665	83.5237335
20	34.8337335	83.764358
21	34.869617	83.4021415
22	34.8980165	84.3345
23	34.801667	83.4278165
24	34.871733	83.484467
25	34.8121165	83.8781335
26	34.659025	83.849658
27	34.909075	83.276242
28	34.8234085	83.995175
29	34.624842	84.1242665
30	34.616733	83.986767
47	34.4168325	84.20022972
48	34.61685389	83.65012917
49	34.81670611	83.51672472
50	34.51687694	84.20008833
51	34.85000111	83.98354333
52	34.60011944	83.76689889
53	34.45022139	84.30007972
54	34.66680333	84.60019056
56	34.85009306	84.01671472
57	NA	NA
58	34.93360528	83.38336417
59	34.96689667	84.06676778
61	NA	NA
62	34.60011944	83.76689889
63	NA	NA

Appendix B: Directions to sites on public lands.

SiteID: 3

From Cleveland, take 129 N. Go a few miles past where 19 joins 129. Turn right onto FS Road 443 (Sign for Boggs Creek). Go a mile and continue straight where the gravel road splits to the right and fords the stream. Park at the area beside the large tree trunk (bottom of sampling reach).

SiteID: 5

Take 52 SE from Ellijay. Pass Amicalola Falls State Park. After several miles, turn left on Wesley Chapel Rd. Take the first left on Dan Fowler Rd. Pass chicken houses. Turn left onto Blackhawk Dr. (gravel road). Ford stream, take left fork, and park on right just before second crossing of creek (immediately upstream of road = bottom of sampling reach).

SiteID: 6

From Clayton, take 76 W. Turn right on Persimmon Rd at the fire station. Go by the Tallulah River campground signs. Turn left on Coleman River Rd. Cross over Coleman River. Park on the right on the other side of the bridge. Bottom of sampling reach ~15 m upstream of the bridge.

SiteID: 7

Take 19 N past DeSoto Falls. Turn left on 180 W at Vogel State Park. A mile or so past Lake Winfield Scott, turn right on NFS road 33 (Cooper Creek Rd.) Go to the t-intersection. From the intersection, back track ~1 mile on 33. Fish and habitat sampling reach is located ~1 mile up the road. Go down the hillside and river is running through an open floodplain area with a small tributary flowing into the left side of the river at the bend.

SiteID: 9

Take 19/60 N from Dahlonega. At the split, continue on 19 N. Turn left on FS 34 at the signs for Pisgah Church and Game Check-In. Park at last camping site in the Waters Creek campground. Take the trail away from the camping site. Bottom of sampling reach is located ~50m upstream.

SiteID: 10

From Helen, take 17/75 N and soon turn left on 75 Alt S. Check in with DNR Smithgall for permission. Enter the Conservation Area. Park on the side of the road immediately before the road bends to the left. Follow footpath down to creek. Top of sampling reach ~30 m below the bridge.

SiteID: 11

From Helen, take 17/75 N, and soon turn right on 356 N. After about 2 miles, turn left on Anna Ruby Falls Rd. Go a couple miles and park in gravel area on right across from Smith Creek cottages, before NFS sign (picnic shelters- camping).

SiteID: 14

From Clayton, take Warwoman Rd.(28E) off of 441 N (in front of Taco Bell). Turn left on Hale Ridge Rd. Stay straight on 86C Fork. Pass Mt. Zion Church. Turn right on Overflow Creek Rd. Turn left on 86B. Park after crossing the bridge. Top of sampling reach is at the bottom of the shoal downstream of the bridge.

SiteID: 16

From Blue Ridge, take 5. Turn left onto Hwy 2. Turn left onto Devil's Den Road. Go 1.2 miles and pavement ends. Go another 1.5 miles to NF lands. Park at the area before the road takes a sharp bend to the right. Bottom of sampling reach is located at the engineered pool area immediately down from the parking area.

SiteID: 17

Take 60 N and turn left on Doublehead Gap Rd. (road goes from pavement to gravel). Go several miles. Turn left on NFS road 58 (if you pass the little white "Primitive Baptist Church", you missed the turn). On NFS 58, pass the Blue Ridge NF signs. Park at the first big campground. Top of sampling reach is located ~25 m from the sand bar adjacent to the campgrounds.

SiteID: 20

From Helen, take 17/75 N. Go several miles and turn left on 180. Park at gravel pullover on left between mile markers 2 and 3. Top of sampling reach is located at the parking area.

SiteID: 24

From Clayton, take 76 W. Turn right on County Rd 193 just after crossing Timpson Creek. Take the first onto NFS Rd. 162 (road sign is located a bit up the road). Park at the second small gravel parking area on the right. The bottom of the sampling reach is located immediately down the hill.

SiteID: 27

From Clayton, take a right on Warwoman Rd. Turn left on Henry Rd (CR 235) just past Pleasant Hill Church and Warwoman campground. Continue past houses and take Rd. 340 (looks like a driveway) until it ends at the NF boundary. Sampling reach is located ~50 m up the road. **Note: at end of project, land owner had gated the road blocking public access to the NF.

SiteID: 29

From Dahlonga, take 52 NW. Just past mile 3 marker, turn right on Nimblewill Church Rd. Turn right at stop sign and then left on NFS road 28 at sign for Bull Mt. parking lot. Stay right at fork with road 77. Turn left on SSG B Charles Rd by Ranger camp gym (from this point, you are about 2.5 miles from sampling site). Stay left, ford the creek, and stay left at fork. Ford the creek again and park in the campsite on the other side of the road (look for the metal footpath across the stream upstream of the road). Top of sampling reach is located below the ford.

SiteID: 30

Take 60 N from Dahlonega. After small white house on left, turn left on Ridley Rd. Go about 2 miles to t-intersection and turn left. Go about 0.2 miles and turn left on Walnut Valley Rd. Park at area by chicken houses. Enter the woods across from where you can see the mountains in the distance on the right. End straight down to the stream and the bottom of the sampling reach.

SiteID: 47

Take 53 west from Gainesville (or Dawsonville). Cross over the Amicalola bridge. Park on gravel pad on the right immediately after the bridge. Bottom of sampling reach immediately down the hill.

SiteID: 61

Take 441N from Clayton. Turn right onto GA 246. Right on Old Mud Creek Rd at the Sinclair Gas Station. Turn right into Sky Valley gated community. Turn left the t-intersection by police station. Right on Tahoe rd. Continue until Tahoe dead ends at waterfall. Follow path by running near the stream for ~10 minutes. Go down the hill and enter the stream. Bottom of sampling reach is ~25 m upstream of where stream narrows and tumbles to the left.

Appendix C, Table 1: Trout collected at Group 1 sites in 2001.
Trout class: 1 = Naturalized and 2 = Stocked.

SiteID	Trout Species	Total length (mm)	Trout Class
1	<i>Oncorhyncus mykiss</i>	162	1
		188	1
		195	1
		200	1
	<i>Salmo trutta</i>	267	1
		300	1
3	<i>Oncorhyncus mykiss</i>	174	2
		210	2
	<i>Salmo trutta</i>	75	1
		82	1
		88	1
		95	1
		148	1
		218	2
		222	2
		225	2
		226	2
		243	2
		250	2
		253	2
5	<i>Oncorhyncus mykiss</i>	155	2
		180	2
		190	2
		200	2
		225	2
		232	2
		258	2

SiteID	Trout Species	Total length (mm)	Trout Class
6	<i>Oncorhyncus mykiss</i>	47	1
		49	1
		65	1
		125	1
		128	1
		130	1
7	<i>Oncorhyncus mykiss</i>	220	2
		235	2
	<i>Salmo trutta</i>	43	1
		66	1
		74	1
		75	1
9	<i>Oncorhyncus mykiss</i>	57	1
		60	1
		65	1
		72	1
		76	1
		79	1
		90	1
		93	1
		94	1
10	<i>Oncorhyncus mykiss</i>	44	1
		45	1
		46	1
		47	1
		48	1
		49	1
		50	1
		51	1
		52	1
		53	1
		54	1
		55	1
		56	1
		57	1
		58	1

SiteID	Trout Species	Total length (mm)	Trout Class
10	<i>Oncorhyncus mykiss</i>	59	1
		60	1
		61	1
		62	1
		65	1
		66	1
		67	1
		68	1
		70	1
		71	1
		72	1
		73	1
		75	1
		89	1
		99	1
		100	1
		110	1
		117	1
		123	1
		160	1
		162	1
	<i>Salmo trutta</i>	50	1
		55	1
		56	1
		56	1
		58	1
		60	1
		65	1
11	<i>Oncorhyncus mykiss</i>	40	1
		44	1
		45	1
		53	1
		55	1
		200	1
		210	1
		230	1
		235	2
		235	1

SiteID	Trout Species	Total length (mm)	Trout Class
11	<i>Oncorhyncus mykiss</i>	240	1
	<i>Salmo trutta</i>	70	1
		245	1
12	<i>Oncorhyncus mykiss</i>	65	1
		75	1
		80	1
		155	1
		160	1
		190	1
		210	1
13	<i>Oncorhyncus mykiss</i>	77	1
16	<i>Oncorhyncus mykiss</i>	45	1
		52	1
		54	1
		55	1
		56	1
		60	1
		63	1
		65	1
		66	1
		67	1
		70	1
		71	1
		74	1
		80	1
		84	1
		94	1
		103	1
		106	1
		107	1
		118	1
		124	1
		149	1
		195	1

SiteID	Trout Species	Total length (mm)	Trout Class
17	<i>Oncorhyncus mykiss</i>	50	1
		51	1
		58	1
		59	1
		67	1
		70	1
		72	1
		74	1
		75	1
		80	1
		148	1
		171	1
		235	1
	<i>Salmo trutta</i>	57	1
		58	1
		64	1
		315	1
		400	1
19	<i>Oncorhyncus mykiss</i>	223	2
20	<i>Oncorhyncus mykiss</i>	48	1
		50	1
		52	1
		53	1
		54	1
		55	1
		57	1
		59	1
		60	1
		62	1
		63	1
		64	1
		65	1
		69	1
		70	1
		75	1
		85	1

SiteID	Trout Species	Total length (mm)	Trout Class
20	<i>Oncorhyncus mykiss</i>	110	1
		140	1
		150	1
		155	1
		160	1
		170	1
		180	2
		185	2
		190	2
		190	1
		200	2
		220	2
		225	2
		230	2
		240	2
		315	2
	<i>Salmo trutta</i>	210	2
		220	2
		225	2
		230	2
21	<i>Oncorhyncus mykiss</i>	202	2
		220	2
22	<i>Oncorhyncus mykiss</i>	78	2
		95	2
23	<i>Oncorhyncus mykiss</i>	59	1
		61	1
		70	1
24	<i>Oncorhyncus mykiss</i>	110	1
		130	1
		140	1
25	<i>Oncorhyncus mykiss</i>	230	2

SiteID	Trout Species	Total length (mm)	Trout Class
26	<i>Oncorhyncus mykiss</i>	46	1
		57	1
		75	1
		82	1
		180	1
		190	2
		235	2
		245	2
		250	2
		258	2
		260	2
27	<i>Oncorhyncus mykiss</i>	41	1
		202	1
29	<i>Oncorhyncus mykiss</i>	81	1
		139	1
		154	1
		158	1
		159	1
		170	1
		180	1
		221	1
30	<i>Oncorhyncus mykiss</i>	200	2

Appendix C, Table 2: Trout collected at Group 1 sites resampled in 2002.
Trout class: 1 = Naturalized and 2 = Stocked.

SiteID	Trout Species	Total length (mm)	Trout Class
5	<i>Oncorhyncus mykiss</i>	167	1
		170	2
		195	1
		215	2
		217	2
		230	2
		256	1
9	<i>Oncorhyncus mykiss</i>	49	1
		62	1
		74	1
16	<i>Oncorhyncus mykiss</i>	41	1
		45	1
		47	1
		50	1
		50	1
		52	1
		66	1
		85	1
		86	1
		92	1
		100	1
		105	1
		105	1
		110	1
		138	1
17	<i>Oncorhyncus mykiss</i>	36	1
		43	1
		45	1
		45	1
		45	1
		45	1
		51	1
		64	1

SiteID	Trout Species	Total length (mm)	Trout Class
17	<i>Oncorhyncus mykiss</i>	137	1
		150	1
20	<i>Oncorhyncus mykiss</i>	43	1
		45	1
		45	1
		45	1
		46	1
		46	1
		47	1
		49	1
		50	1
		50	1
		52	1
		55	1
		55	1
		55	1
		60	1
		60	1
		62	1
		64	1
		100	1
		125	1
		135	1
		145	2
		165	1
		175	2
		185	2
		200	2
21	<i>Oncorhyncus mykiss</i>	225	2
		245	2
		260	2
23	<i>Oncorhyncus mykiss</i>	36	1
		41	1
		45	1
		48	1
		54	2
		55	2

SiteID	Trout Species	Total length (mm)	Trout Class
23	<i>Oncorhyncus mykiss</i>	55	2
		60	1
		69	2
		120	1
		125	1
		135	1
26	<i>Oncorhyncus mykiss</i>	58	1
		60	1
		65	1
		160	2
		172	2
		175	2
		195	2
		200	2
		215	1
		250	2
29	<i>Salmo trutta</i>	78	1
		82	1
		82	1
		82	1
		84	1
		85	1
		93	1
		165	1
		175	1
		182	1
		195	1
		255	1

Appendix C, Table 3: Non-trout species collected at Group 1 sites in 2001.

SiteID	Species	# Individuals
1	<i>Cottus spp.</i>	66
	<i>Hypentelium etowanum</i>	18
	<i>Semotilus atromaculatus</i>	39
3	<i>Hypentelium etowanum</i>	6
	<i>Luxilus zonistius</i>	26
	<i>Nocomis leptcephalus interocularis</i>	5
5	<i>Cottus spp.</i>	66
	<i>Hypentelium etowanum</i>	2
	<i>Semotilus atromaculatus</i>	11
6	<i>Rhinichthys atratulus</i>	14
7	<i>Campostoma sp.</i>	7
	<i>Cottus spp.</i>	58
	<i>Etheostoma blennioides</i>	2
	<i>Hypentelium nigricans</i>	8
	<i>Lepomis macrochirus</i>	1
	<i>Luxilus coccogenis</i>	13
	<i>Nocomis micropogon</i>	20
	<i>Notropis leuciodus</i>	51
	<i>Rhinichthys cataractae</i>	4
	<i>Semotilus atromaculatus</i>	2
9	<i>Campostoma oligolepis</i>	12
	<i>Hypentelium etowanum</i>	22
	<i>Luxilus zonistius</i>	28
	<i>Nocomis leptcephalus interocularis</i>	55
	<i>Notropis leuciodus</i>	101
	<i>Percina nigrofasciata</i>	24
10	<i>Hypentelium etowanum</i>	1
	<i>Luxilus zonistius</i>	16
	<i>Nocomis leptcephalus interocularis</i>	26
	<i>Percina nigrofasciata</i>	7

SiteID	Species	# Individuals
11	<i>Campostoma anomalum</i>	11
	<i>Cottus spp.</i>	64
	<i>Luxilus zonistius</i>	92
	<i>Nocomis leptocephalus interocularis</i>	29
	<i>Percina nigrofasciata</i>	6
12	<i>Cottus spp.</i>	34
	<i>Semotilus atromaculatus</i>	6
13	<i>Campostoma anomalum</i>	11
	<i>Clinostomus funduloides</i>	1
	<i>Cottus spp.</i>	135
	<i>Hypentelium nigricans</i>	8
	<i>Ichthyomyzon greeleyi</i>	16
	<i>Lepomis auritus</i>	4
	<i>Rhinichthys cataractae</i>	13
	<i>Semotilus atromaculatus</i>	3
16	<i>Cottus spp.</i>	63
	<i>Nocomis micropogon</i>	4
	<i>Semotilus atromaculatus</i>	8
17	<i>Cottus spp.</i>	85
	<i>Etheostoma blennioides</i>	6
	<i>Nocomis micropogon</i>	1
	<i>Rhinichthys cataractae</i>	5
19	<i>Cottus spp.</i>	27
	<i>Etheostoma blennioides</i>	1
	<i>Luxilus zonistius</i>	75
	<i>Nocomis leptocephalus interocularis</i>	23
	<i>Notropis lutipinnis</i>	118
	<i>Notropis sp.</i>	1
	<i>Percina nigrofasciata</i>	24
	<i>Scartomyzon rupiscartes</i>	10
	<i>Semotilus atromaculatus</i>	6

SiteID	Species	# Individuals
20	<i>Campostoma anomalum</i>	2
	<i>Cottus spp.</i>	109
	<i>Nocomis micropogon</i>	1
	<i>Rhinichthys cataractae</i>	1
21	<i>Catostomus commersoni</i>	2
	<i>Cottus spp.</i>	35
	<i>Lepomis auritus</i>	5
	<i>Lepomis macrochirus</i>	4
	<i>Micropterus salmoides</i>	1
	<i>Nocomis leptocephalus interocularis</i>	87
	<i>Notropis lutipinnis</i>	343
	<i>Rhinichthys atratulus</i>	3
	<i>Scartomyzon rupiscartes</i>	4
22	<i>Ambloplites rupestris</i>	3
	<i>Campostoma sp.</i>	15
	<i>Cottus spp.</i>	57
	<i>Etheostoma blennioides</i>	12
	<i>Etheostoma rufilineatum</i>	5
	<i>Hypentelium etowanum</i>	1
	<i>Lepomis macrochirus</i>	5
	<i>Luxilus coccogenis</i>	42
	<i>Micropterus salmoides</i>	2
	<i>Nocomis micropogon</i>	13
	<i>Notemigonus crysoleucas</i>	1
	<i>Notropis leuciodus</i>	21
23	<i>Campostoma anomalum</i>	17
	<i>Cottus spp.</i>	141
	<i>Hypentelium nigricans</i>	18
	<i>Lepomis auritus</i>	1
	<i>Lepomis cyanellus</i>	2
	<i>Luxilus coccogenis</i>	12
	<i>Micropterus dolomieu</i>	1
	<i>Nocomis leptocephalus interocularis</i>	183
	<i>Notropis lutipinnis</i>	240
	<i>Rhinichthys atratulus</i>	5
	<i>Scartomyzon rupiscartes</i>	10
	<i>Semotilus atromaculatus</i>	1

SiteID	Species	# Individuals
24	<i>Clinostomus funduloides</i>	14
25	<i>Campostoma sp.</i>	10
	<i>Clinostomus funduloides</i>	29
	<i>Cottus spp.</i>	112
	<i>Cyprinella galactura</i>	3
	<i>Hypentelium nigricans</i>	10
	<i>Ichthyomyzon greeleyi</i>	2
	<i>Luxilus coccogenis</i>	2
	<i>Nocomis micropogon</i>	1
	<i>Notropis spectrunculus</i>	53
	<i>Perca flavescens</i>	1
	<i>Percina evides</i>	2
	<i>Rhinichthys cataractae</i>	16
	<i>Semotilus atromaculatus</i>	25
26	<i>Campostoma oligolepis</i>	23
	<i>Cottus spp.</i>	169
	<i>Luxilus zonistius</i>	84
	<i>Nocomis leptcephalus interocularis</i>	18
27	<i>Cottus spp.</i>	18
	<i>Hypentelium nigricans</i>	5
	<i>Nocomis leptcephalus interocularis</i>	4
	<i>Notropis lutipinnis</i>	5
	<i>Scartomyzon rupiscartes</i>	3
	<i>Semotilus atromaculatus</i>	23
29	<i>Semotilus atromaculatus</i>	90
30	<i>Ambloplites rupestris</i>	11
	<i>Ameiurus brunneus</i>	10
	<i>Campostoma sp.</i>	2
	<i>Lepomis auritus</i>	1
	<i>Lepomis macrochirus</i>	3
	<i>Luxilus zonistius</i>	190
	<i>Nocomis leptcephalus interocularis</i>	3

Appendix D, Table 1: 2001, 2002, and 2003 M7DAM temperatures (°C) for Group 1 sites.

SiteID	M7DAM temperature (°C)			Date of M7DAM temperature		
	2001	2002	2003	2001	2002	2003
1	19.6	21.3	18.2	8/8	8/5	8/25
2	23.2	24.2	21.5	8/8	8/3	8/25
3	20.5	21.3	19.8	8/9	7/30	8/31
4	23.7	24.6	23.5	8/9	7/29	8/25
5	20.4	21.3	19.6	8/8	7/30	8/25
6	17.7	18.8	16.9	8/10	7/30	8/27
7	19.4	21.0	20.0	8/17	7/30	8/30
8	22.3	23.1		8/9	7/31	
9	20.5	22.6	20.7	7/11	7/31	8/25
10	20.1	20.9	19.0	8/9	7/31	8/28
11	19.9	20.7		8/9	8/19	
12		21.2	19.8		7/30	8/28
13	21.0	22.5	20.2	8/8	7/29	8/26
14	19.1	19.9	18.5	8/10	8/22	8/25
16	17.6	18.3	17.5	7/28	7/30	8/30
17	18.4	19.0	17.8	8/8	7/30	8/31
19	21.3	22.7	20.4	8/9	7/31	8/25
20	18.3	18.8		8/10	7/29	
21	22.6	23.0	20.1	8/10	7/7	8/27
22	22.0	23.2	21.8	8/8	8/1	8/28
23	21.7	22.8	20.7	8/10	7/30	8/25
24	19.7	20.3	19.1	8/10	7/30	8/25
25	22.4	23.0	22.0	8/9	8/22	8/26
26	21.7	22.8	20.7	8/9	7/30	8/25
27	19.9	20.0	18.9	8/10	7/31	8/26
28	22.1	22.8	21.0	8/8	7/31	8/31
29	20.1	20.9	19.4	8/10	8/22	8/25
30	24.5	25.1	23.2	7/11	8/1	8/27

Appendix D, Table 2: 2003 M7DAM temperatures (°C) for Group 2 sites.

SiteID	2003 M7DAM temperature (°C)	Date of M7DAM temperature
2	21.5	8/25
21	20.1	8/27
22	21.8	8/28
47	19.4	8/24
48	22.1	8/26
49	23.4	8/26
50	18.9	8/25
51	22.4	8/27
52		
53	20.8	8/26
54	18.8	9/1
56	21.7	8/28
57		
58	21	8/26
59	21.7	8/17
61	22.6	8/25
62	21.8	8/24
63	20.2	7/28

Appendix D, Table 3: Habitat data collected at Group 1 sites in 2001.

SiteID	Riffle Embeddedness Score	% Coarse Substrate	Local Stream Slope	Maximum Velocity (m/second)	Maximum Depth (cm)
1	41	25	0.59	0.96	89.5
2	43	36	1.30	0.93	65.5
3	31	77	3.33	1.43	77
4	46	18	0.35	0.52	49
5	36	33	1.80	0.73	31
6	38	43	1.04	0.85	75
7	39	33	0.55	0.94	79
8	43	21	1.01	0.78	49
9	39	39	0.81	0.96	58
10	34	43	0.46	0.75	60
11	37	50	2.05	1.4	123
12	41	50	2.33	1.19	48
13	34	42	0.56	1.07	84.5
14	36	52	1.07	1.4	60
16	35	60	2.48	1.69	72
17	40	52	1.25	1.33	71.5
19		66	1.39	1.61	64.5
20	42	39	1.59	0.95	32
21		25	0.31	1.56	66
22		78	2.01	2.2	64
23	40	23	0.36	0.76	61.5
24	38	44	1.21	1.13	67
25	36	21	0.29	1.41	42.5
26	32	78	0.86	1.24	64.5
27	36	32	0.57	0.7	82
28	47	20	0.31	0.97	35
29	42	30	0.45	0.44	47.5
30	43	37	0.45	1.17	67