GEOSPATIAL TECHNIQUES FOR STREAM RESEARCH IN THE SOUTHERN

BLUE RIDGE MOUNTAINS

by

EDWARD P. GARDINER

Under the Direction of Dr. J. L. Meyer

ABSTRACT

This research demonstrates how geographic information systems (GIS) complement stream research based on sampling habitat characteristics, water quality, and biota. The study area encompasses the Upper Little Tennessee, Tuckaseegee, Pigeon, and French Broad River basins in the Blue Ridge physiographic province in western North Carolina, U.S.A. Chapter 2 presents a methodology for constructing a spatial database of linked watershed and stream attributes and assesses the planimetric accuracy of automatically extracted watershed boundaries. Watersheds extracted from 30-m USGS Level 1 digital elevation models (DEM) larger than 250 ha have negligible errors. The mean distance between automatically derived and hand digitized watershed boundaries was comparable to the radius of a circle of the same size as a DEM pixel. In Chapter 3, a watershed sediment yield model is constructed using the Revised Universal Soil Loss Equation (RUSLE). The resolution of source data largely determines the precision of model estimates that use those data. When 90-m input data are used, 80% of the variance in model output using 30-m data are explained, but precision drops rapidly as data resolution is coarsened beyond 90 m. Modeled sediment yields (tonne yr^{-1}) are statistically related to baseflow sediment yield estimates calculated from stream discharge (m^3 s^{-1}) and total suspended solids (mg l^{-1}) measurements. Chapter 4 examines land use and physiographic factors associated with fish assemblage shifts in the Upper Little Tennessee River basin. Large and small watersheds greater and less than 39 km^2 in area have distinct fauna. Smoky sculpin proportions (*Cottus bairdi* ssp), widespread taxa, and restricted range taxa are statistically related to modeled sediment yield rate (tonne km^{-2} yr^{-1}). Chapter 5 synthesizes biological, chemical, and land use data to project the effects of future land use changes onto forecasts of stream conditions in the Upper Little Tennessee River and Cane Creek basins. This research provides a powerful suite of analytical tools and perspectives relating GIS to stream research.

INDEX WORDS: Geographic information systems (GIS), water resources geography, watershed management, water quality, simulation, land use, homogenization, stream, fish assemblage

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DEDICATION

I dedicate this work to my family, especially my wife Christine and my son Asa. Christine and Asa, you two continually give me the perspective I need. Your love and spirit have bolstered me immeasurably. Special thanks to my parents Arthur and Tish, grandmother Tish, Uncle Fred, and Aunt Joanie. This work would not have been possible without the enduring support of all my relatives.

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CHAPTER 1

INTRODUCTION AND LITERATURE REVIEW

The research presented here develops and applies geographic information systems (GIS, Burrough 1993) data and algorithms to analyze the relationship between landscape conditions and stream ecosystems. Stream ecosystem structure and function depend on land use in the catchment contributing flow to a stream (Likens et al. 1970, Hynes 1975). Use of GIS has stimulated valuable research in stream ecology because computerized GIS make it possible to examine land use and terrain information over large areas. By comparing results from field surveys and computerized watershed analysis, Roth et al. (1996) prompted a debate over whether near-stream or whole-watershed land use was a more powerful explanatory variable in describing stream biological integrity (Lammert and Allan 1999). Allan et al. (1997) convincingly argued that local conditions are more important for habitat and organic matter processes while nutrient supply, hydrology, and sediment delivery are determined by the land cover and land use in entire watersheds. Generalizations about the geomorphic form, nutrient dynamics, and communities found in streams in different landscape settings underlie the River Continuum Concept (Vannote et al. 1980), a paradigm that continues to influence stream research. A broad consensus in the past decade has called for attention to increasingly complex questions, such as examining watershed conditions throughout entire physiographic provinces, modeling effects of landscape processes on streams, and forecasting land use change and effects on streams (Naiman et al. 1995). These endeavors require novel uses of computers and GIS.

When GIS techniques and ecological research questions advance in tandem, changes in perspective in each discipline are likely to emerge rapidly (Fisher 1997). Rapid advances are needed to explore relationships between watershed conditions and ecological integrity (sensu Karr 1992, Bunn and Davies 2000) of rivers in the southeastern United States. The southeastern United States supports one of the temperate world's richest areas of endemism and diversity for invertebrates (Wallace et al. 1992) and temperate freshwater fishes (Burkhead and Jenkins 1991, Etnier and Starnes 1991), providing year-round or spawning habitats for over 660 species of fishes (Warren et al. 2000). That diversity is under considerable threat, however, with approximately 28% of those taxa endangered, threatened, or vulnerable to extirpation (Warren et al. 2000). More than half of the diversity of freshwater fishes in the southeastern US occupy mountain streams of the southern Appalachians where darters (*Percidae*) and minnows (*Cyprinidae*) are concentrated and are especially threatened. Over 20% of taxa in these families are extinct, endangered, threatened, or vulnerable (Walsh et al. 1995). Recent declines in freshwater mussel diversity and abundance have been largely attributed to the many stresses humans have imposed on southeastern rivers (Box and Mossa 1999). Having evolved in low nutrient, low temperature waters, many native endemic fish taxa are also especially vulnerable to ecosystem perturbations caused by humans, for example land use changes that increase runoff (Booth and Jackson 1997, Doyle et al. 2000) and nutrient delivery to streams (Allan 1995).

Much of the effort aimed at protecting running waters in the U.S. has focused on relatively large waterways (Meyer and Wallace 2001), despite the fact that most river miles are composed of small streams (Leopold 1994). Digital data oriented toward hydrologic applications also have been biased toward large rivers. For example, the river reach file (RF3) and hydrologic unit coverage (HUC) programs have been integrated into a seamless stream and watershed database, the National Hydrography Dataset (NHD, Roth 1998); but the NHD applies to relatively large stream channels, i.e., those that appear on maps at 1:100,000. Comparable data sets targeted at small streams are necessary to support watershed research in smaller stream settings (Meyer and Wallace 2001). When such data are available, tremendous potential can be released, allowing insights about small streams to be applied over large areas. This research is an effort to unlock some of that potential using data that are commonly available throughout the United States.

Each of the chapters presented here uses Level 1 30-m digital elevation models (DEMs, USGS 1993) to provide terrain information and stream hydrography data digitized from 1:24,000 scale topographic maps to represent streams. Comparable data sets are also available throughout the world. Watershed data and analysis methods are urgently needed to study and protect rivers in the southeastern United States, but their utility and efficacy depend upon accuracy, consistency, and precision. Therefore, Chapter 2 presents methods for building watershed databases, analyzes morphometric properties of digital watershed data, and quantifies planimetric inaccuracies in the watershed boundaries. Principal component axes based on characteristics of digital streams and watersheds are used to separate them into distinct classes. Documenting and understanding systematic errors enhance the use of automatically extracted watershed boundaries for watershed management and research.

The two chapters that follow address the application of a sediment yield model to study stream conditions in a river basin in western North Carolina. Excessive sediment is one of the most prevalent stressors to aquatic ecosystems in the United States (Judy et al. 1984, Waters 1995, US EPA 1997). Sediment also has wide-ranging deleterious effects because it alters stream geomorphology and interferes with feeding, respiration, and reproduction among invertebrates and fishes (Waters 1995).

Chapter 3 describes the development, sensitivity analyses, and application of a watershed sediment yield model. The methods used are very similar to those used in the establishment of total maximum daily load (TMDL) documents for regulation of sediment in the State of Georgia (US EPA 2000). Soil loss estimates from the Revised Universal Soil Loss Equation (RUSLE, Renard 1997) are routed to stream channels in the Upper Little Tennessee River basin, and a series of simulations demonstrates the sensitivity of results to the resolution of data used as input to the model. Quantitative measures are provided for model performance as a function of input data resolution and in comparison to annual baseflow sediment yield calculated from discharge and total suspended solids (TSS) measurements.

In Chapter 4, the sediment yield model is combined with watershed characteristics to explore physiographic and land use correlates of shifts in fish assemblage structure in the Upper Little Tennessee River basin. Excessive sediment reduces fish reproductive rates (Newcombe and MacDonald 1991, Newcombe and Jensen 1996, Montgomery et al. 1996), increases morbidity, and is associated with lower population sizes for a variety of taxa (Berkman and Rabeni 1987, Newcombe and Jensen 1996). Scott and Helfman (2001) have suggested that native fishes that evolved in mountain streams of the crystalline Blue Ridge Mountains are more vulnerable to habitat modifications than more widely distributed native taxa. Variations within those groups and individual taxa are examined relative to land use, physiographic properties, and sediment loading.

Some stream processes, such as nutrient cycling, respond and return to pre-disturbance conditions quickly; however, sediment entering headwater streams following forest clearing may remain for more than a decade (Webster et al. 1991). Land uses from past decades may have altered stream geomorphology, which in turn shapes habitat conditions for present day fauna (Harding et al. 1998). Therefore, it is important to consider the long-term impacts to stream ecosystems of current changes in land use. Chapter 5 demonstrates how land use change projections (Wear and Bolstad 1998) can be incorporated into the design of a study intended to measure and detect changes in stream ecosystem structure and function expected during the next 20 years. Rather than looking only to past and/or current land uses, this research explores likely consequences of land use change for stream form and function by relating present biological and physical patterns to land use, by projecting future land use and land cover (Wear and Bolstad 1998), and by linking projected land use to expected consequences for streams.

These efforts establish many links between stream condition and landscape patterns and processes. Simultaneously, innovative spatial information techniques are advanced that are specifically tailored to stream research. Each chapter provides technological and methodological advances for geographers and ecologists examining the effects of land use on stream ecosystems. All the techniques apply commonly available data to complex problems, so these methods are replicable in other settings. Strengths and limitations of geospatial data and GIS techniques for stream research are explored in depth to enhance future watershed-based stream research.

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CHAPTER 2

WATERSHED DATABASES FOR LARGE REGIONS[1]

[1]Gardiner, E.P., J.L. Meyer, and P.V. Bolstad. To be submitted to the Journal of the American Water Resources Association.

Abstract

Water resource professionals require higher resolution watershed data than are available commonly, so they often use automatic watershed extraction techniques. To facilitate analyses of streams and watersheds across large regions, a set of methods is presented to build a normalized relational database of streams and their watersheds using 1:24,000-scale digital line graphs (DLG) and digital elevation model (DEM) data. Watershed boundaries, stream vectors from DLGs, and DEMs were used to extract morphometric characteristics of streams, stream networks, and watersheds. Principal component axes based on those characteristics separated watersheds into distinct classes according to size, landscape position, and slope. Strahler order was a weak correlate of watershed measurements, and stream density was independent of other measures of basin and stream network geometry. These results point to inconsistent methods used in mapping streams on 1:24,000 scale maps. When compared to hand-delineated watersheds, automatically derived watersheds greater than 250 ha had minimal proportions of their total area (< 5%) incorrectly delineated. The average distance from automatically derived lines to hand digitized ones was approximately 17 m. Most of the total length (> 95%) of automatically derived watershed boundaries were within 50 m of hand digitized watershed boundaries. Documenting and understanding systematic errors enhance the use of automatically extracted watershed boundaries for watershed management and research.

KEYWORDS: Geographic information systems, water resources geography, watershed management, database, accuracy

Introduction

Digital stream and watershed boundary data support many applications in freshwater research that use geographic information systems (GIS, Burrough, 1993). These include hydrologic modeling, conservation, restoration, statistical analyses, and site selection. The National Hydrography Dataset (Roth, 1998) and Hydrologic Unit Coverage (HUC; Seaber et al., 1987) programs were designed to meet these and other needs on a national basis. The former combines attributes and digital representations of river reaches mapped at 1:100,000 scale while the latter provides watershed boundaries for streams of varying sizes throughout the country. However, researchers require watershed data for streams mapped with more detail than is represented at 1:100,000 scale. Small streams have a greater combined length than large rivers, so they harbor tremendous biological diversity, are responsible for the majority of organic matter processing and nutrient cycling that occurs in flowing water, and deliver a large proportion of the accessible freshwater supply used by humans (Meyer and Wallace 2001). Further, much stream research is conducted in streams that do not appear at a map scale of 1:100,000 (Meyer and Wallace, 2001). The lack of adequate watershed data for smaller tributaries requires researchers to build their own GIS databases to support stream research.

Ongoing research promises to enable the automatic delineation from digital elevation models (DEMs) of perennial channels and their watersheds across large regions (Dietrich et al., 1992; Helmlinger et al., 1993; Gandolfi and Bischetti, 1997; Tarboton, 1997; Wolock, 1997; Rieger, 1998; Tucker and Bras, 1998). Most automated watershed delineation techniques modify DEMs before extracting data from them, so original DEM resolution and data quality differ somewhat from those attributes of modified DEMs (Zhang and Montgomery, 1994; Band and Moore, 1995; Hodgson, 1995; Gandolfi and Bischetti, 1997; Garbrecht and Martz, 2000). Automatically derived watershed and digital stream data can be used best when organized in a carefully designed database and when accuracy limitations are understood. Such analyses will especially aid in management and policy applications (Maidment and Djokic, 2000).

The first objective of this study is to address the need for methods to archive and query watershed attributes (Band, 1989) across large areas and at higher resolution than is now possible with publicly available data. Methods are presented for building a normalized relational database of streams, watersheds, and their attributes using: (a) hydrography digitized from 1:24,000 scale topographic maps or digital line graphs (DLG); and (b) Level 1 30-m DEMs (USGS, 1993). The structure of the database facilitates further analyses of streams, stream networks, and whole watersheds. The second objective is to analyze morphometric properties extracted from the GIS database. Those data are statistically analyzed to identify systematic errors in the original data and to demonstrate the utility of a large database of streams and watersheds for regional analyses. The third objective is to quantify spatial errors in automatically derived watershed polygons relative to those that were digitized manually. These analyses enhance the usefulness of automatically extracted watersheds for regional watershed science applications.

Study Area and Methods

The study area encompasses 8600 km^2 among four basins in western North Carolina (Figure 2.1; Table 2.1). Asheville, located in the north-central portion of the

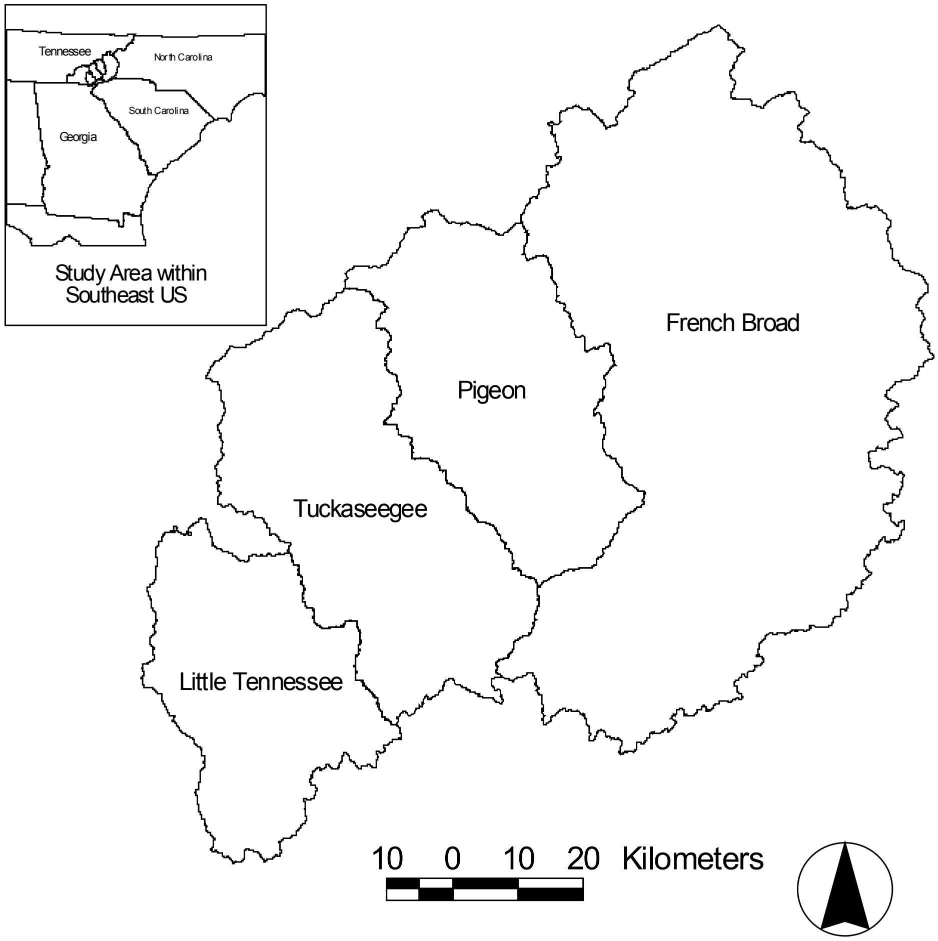

Figure 2.1. Four basins in western North Carolina examined in this study.

Table 2.1. Brief descriptions of four river basins under investigation*.

	Little Tennessee	Tuckaseegee	Pigeon	French Broad
USGS HUC	6010202	6010203	6010106	6010105
Area (km^2)	1154	1706	1403	4309
Forest* (%)	88.9	93.6	84.3	77.9
Agriculture* (%)	8.58	4.34	10.1	16.1
Developed* (%)	2.17	1.46	5.27	5.34
Strahler Order	Number of Stream Arcs			
Order 1	891	1544	1596	6012
Order 2	196	331	389	1436
Order 3	41	75	87	320
Order 4	8	17	17	61
Order 5	2	5	3	16
Order 6		2		3
Total	1775	3062	3287	11995

*Land-cover data source: Hermann 1996.

French Broad Basin, is the largest city in the region. Forests and scattered homes fill the slopes; agriculture, light industry, tourist-oriented businesses, urban, and suburban land uses predominate in the valleys. The study area ranges in elevation from 400 m to 2000 m above sea level. The Coweeta Long Term Ecological Research site is several miles north of the Georgia border in the Little Tennessee River system. The backwater reaches of Lake Fontana define the watershed outlets for the Upper Little Tennessee and Tuckaseegee River basins in this study. The North Carolina border with Tennessee defines the northern extent of the Pigeon and French Broad study areas.

Database Compilation

For each of those four basins, two data preparation paths were followed. In the first (top, Figure 2.2), DEM data were adjusted to ensure that flow routing using DEM data matched flow routing represented by stream vectors. Discrepancies between DEMs and stream vectors can arise from a variety of reasons, including insufficient resolution (Zhang and Montgomery, 1994) and errors from random or systematic factors (USGS, 1993). A modified DEM was generated from Level 1 USGS DEM data using ESRI's (1998) implementation of Hutchinson's (1989) stream burning algorithm. Stream vectors from 1:24,000 scale topographic maps were supplied to act as minimum pour points, or locations on the landscape to which drainage would be forced. Water body polygons and basin boundary coverages were also supplied to improve results (Hutchinson 1989). Processing times were recorded and regressed against watershed area. The second data preparation track (bottom, Figure 2.2) removed cyclic paths in the stream vector data, i.e., large water bodies with right and left banks or meander cut-offs. Vectors were re-oriented to point downstream as necessary, and Strahler orders (Strahler, 1957, 1964)

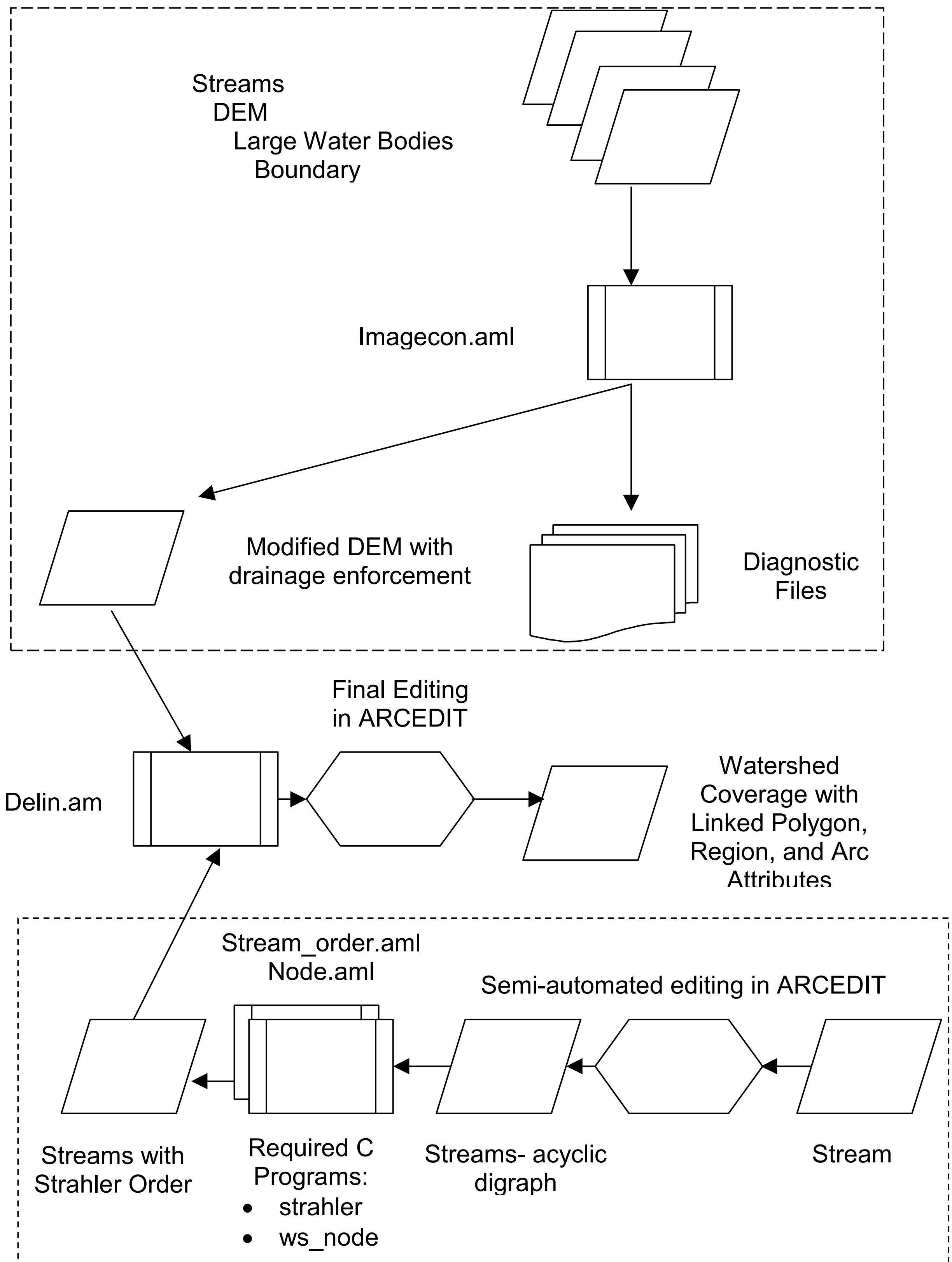

Figure 2.2. Flow chart for watershed delineation procedures. Module tasks are described in the text.

were assigned to each segment (Lampear and Lewis, 1997a). Once the DEM was modified and the stream vectors prepared, watershed boundaries were extracted for each of the vectors in the stream coverage using the watershed delineation process depicted in the center of Figure 2.2.

The watershed delineation process itself required two steps. First, a coverage was created consisting of polygons draining to, and associated with, exactly one arc representing a stream segment between upstream and downstream junctions. Stream vectors were first converted to a grid, and all pixels in the modified DEM that contributed drainage to those pixels were labeled with numbers linking the grid cells to the stream vectors (Jenson and Domingue, 1988;Tarboton et al., 1991; Martz and Garbrecht, 1993; ESRI, 1998). The resultant grid consisted of zones with identification numbers corresponding to the arcs to which they drained (Figure 2.3). The grid regions were converted to polygons, retaining the same identification numbers. Isolated polygons remained on ridges without clearly defined drainage outlets. These polygons were split with arcs representing ridges and dissolved into the neighbor with the longest adjacent border; slivers were removed, and topology was restored. Editing by hand was done to remove the few remaining isolated polygons and miscellaneous errors. Next, watershed regions were defined consisting of all polygons contributing flow to the streams in each network in the study area. For each outlet, all upstream segments were selected, polygons associated with each arc were assigned a temporary code corresponding to the identification number of the outlet arc, and attributes were stored in a table. That table also stored the watershed and stream network morphometric properties defined below.

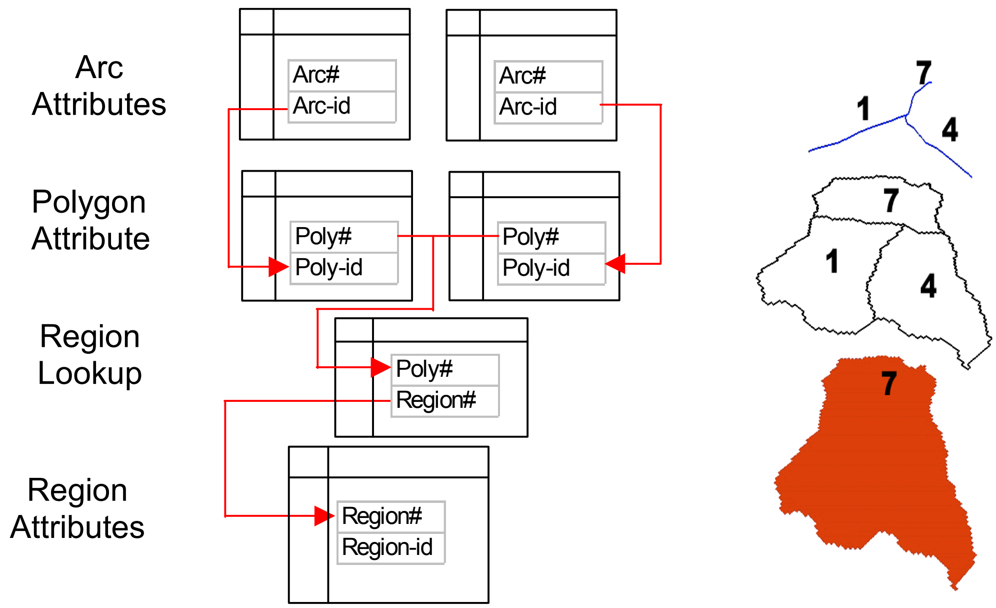

Figure 2.3. Arcs and polygons are linked via primary and foreign keys in their respective attribute tables. This is the basis of forming “regions”, or groups of polygons that contribute to the drainage of defined outlet arcs. On the left are two records representing arcs and polygons, each of which is linked to a single region. The region attributes were all stored in a table. The topological relationships among arcs, polygons, and regions are depicted on the right; three arcs are shown linked to three polygons that compose a single region.

A series of morphometric properties for each stream arc, stream network, and watershed, summarized in Table 2.2, were extracted. Stream gradients for all the arcs in each network were extracted using programs written in C (Lamphear and Lewis, 1997b). To estimate gradient within the entire stream network, stream gradients for all arcs in the watershed network were averaged. The length of each drainage basin was estimated with a surrogate, the length of the major axis of an ellipse approximating the shape of that basin (ESRI, 1998). Relief ratio was calculated as the difference between maximum and minimum elevations in the basin divided by the major axis length, as defined above. Mean hill slope within each watershed was calculated from the raw 30-m DEM. Maximum flow path length from a watershed divide to the outlet of the stream network was recorded (ESRI, 1998). The cumulative length of stream networks was determined by summing the lengths of individual stream arcs in each network.

Principal components analysis was used in the software package SAS (1990) to reduce 13 variables, including 9 watershed properties and 4 stream attributes to a smaller set of component axes that reflect watersheds with similar attributes. The 9 watershed morphometric properties were length of major axis, relief ratio, average hillside slope, average channel slope, total length of streams, maximum flow path length, stream density, area, and watershed perimeter. The 4 stream segment measurements were length, sinuosity, gradient, and outlet elevation.

Accuracy Assessment

Twenty-four watersheds throughout the study region had been digitized by hand for previous analyses of land use (Harding et al., 1998). To obtain estimates of the accuracy of these procedures for small watershed sizes, 12 additional randomly selected

Table 2.2. Morphometric properties extracted from streams, their networks, and their watersheds.

Measurement	**Measured Feature**	**Formula/Method**
Length	Each Stream Arc	Length of the stream segment at the outlet of the watershed; units m
Gradient	Each Stream Arc	Average gradient based on 30m posting interval and 30m DEM; units $m * m^{-1}$
Outlet Elevation	Each Stream Arc	Elevation of outlet downstream node; units m
Sinuosity	Each Stream Arc	(Arc length) / (Euclidean distance between upstream and downstream nodes); units $m * m^{-1}$
Network Slope	Channel Network	Average gradient of all arcs in the upstream network; units $m * m^{-1}$
Cumulative Length	Channel Network	Sum of lengths of arcs comprising stream network; units km
Watershed Area	Whole Watershed	Sum of areas of polygons comprising watershed; units m^2
Stream Density	Whole Watershed	(Cumulative Length / Watershed Area); units $km * km^{-2}$
Major axis	Whole Watershed	Length of longer axis of ellipse with same area and similar shape, compared to whole watershed; units m
Relief ratio	Whole Watershed	(Maximum elevation – Minimum elevation) / (Major axis length); units $m * m^{-1}$
Average Hillslope	Whole Watershed	Mean slope of side slopes in watershed based on 30m DEM; units $m * m^{-1}$
Maximum flow path length	Whole Watershed	Longest flow path length from a ridge to the watershed outlet; units m
Perimeter	Whole Watershed	Length of perimeter encompassing watershed; units m

stream segments were chosen, 3 in each of the four HUCs studied, and polygons that drained directly to each segment were then digitized by hand. For all thirty-six (36) hand-digitized watershed boundaries, watershed divides were traced by hand from 1:24,000 digital raster graphics (DRGs) or 7.5' topographic quadrangles mounted on a digitizer. For each hand-digitized watershed, corresponding automatically derived watersheds were identified.

Planimetric accuracy was assessed by means of area and distance measurements. "Two-dimensional error" was defined as the area bounded by hand-digitized and automatically converted lines (Figure 2.4, shaded region). Log transformed digitized area was regressed against log transformed two-dimensional error.

The linear displacement of automatically and hand-digitized lines was measured in two ways. The first was the minimum distance from digitized vertices to the automatically generated lines. The second test for linear displacement was the buffer width necessary to contain 95% of the perimeter of the automatically generated lines for each watershed boundary (Wang, 2001). Each was used as an indicator of dispersion between the two sets of bounding arcs.

Results

Database Compilation

The steps above produced a normalized relational database of watersheds and their attributes for all the hydrography vectors acquired from 1:24,000 quadrangle maps and DRGs covering the four basins in western North Carolina (Table 2.1).

Relationships among four independent stream arc variables (frequency, length, sinuosity, and gradient) and Strahler order were analyzed within each basin. Only

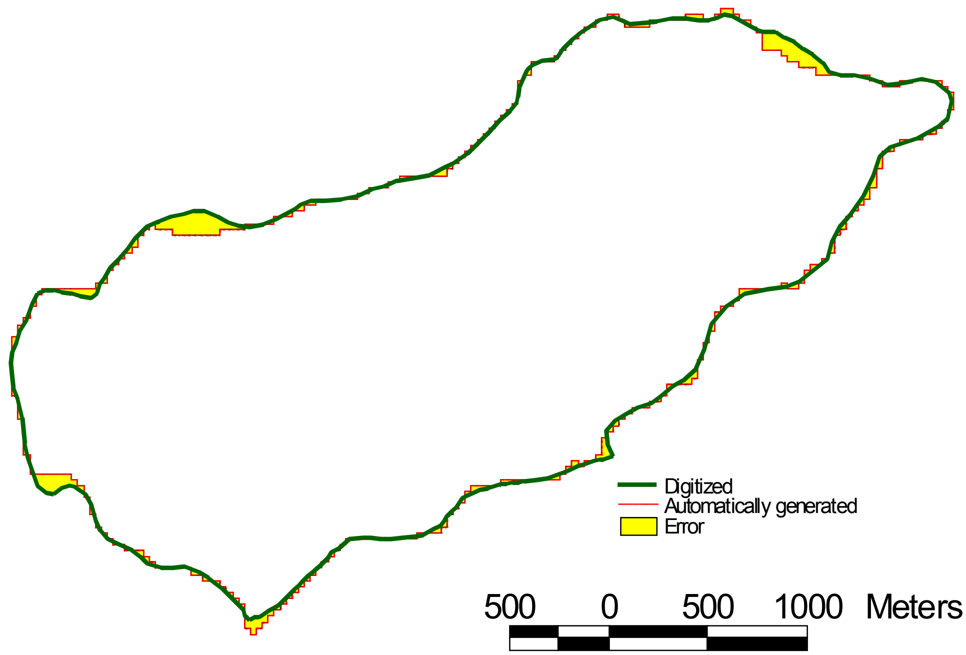

Figure 2.4. Digitized and automatically generated watershed boundaries of Ball Creek within the Coweeta Hydrologic Laboratory. The area subtended by both is the total error. This error was used in analyses of algorithm effectiveness.

frequency and elevation followed expected relationships. The number of streams was a log-linear function of Strahler order (Figure 2.5). However, bifurcation ratios were not constant, meaning the ratio of the number of first order streams: second order streams was not equal to that of second order: third order, etc. Further, the law of stream lengths was not observed, meaning the logarithm of stream length was not a function of Strahler order (Horton, 1932, 1945; Strahler, 1957; Gordon et al., 1992). Nor were sinuosity or gradient related to Strahler order.

Principal components analysis condensed 13 geomorphic variables to 5 axes which encapsulated 83% of the overall variance among the 13 variables. Additional components explained negligible amounts of variance. Component loadings revealed four axes that separated attributes of streams and watersheds (Table 2.3). Watershed perimeter, major axis length, area, cumulative length of channels, length of longest flow path, and sinuosity, in decreasing order of weight, all were positively correlated with the first component. The second component was weighted positively with relief and outlet elevation. The third component was weighted positively by average hill slope, relief ratio, and outlet elevation and negatively by gradient and average network slope.
The fourth component was characterized by sinuous, long stream segments. The fifth component is weighted exclusively on stream density.

The most computationally intensive part of the procedures presented was the generation of a new DEM from the original one. Computer time was linearly related to total basin drainage area ($R^2 = 0.92$; $p = 0.01$) by the following equation:

$$CPU = 2.36 * A_{ws} \qquad \textit{Equation 2.1}$$

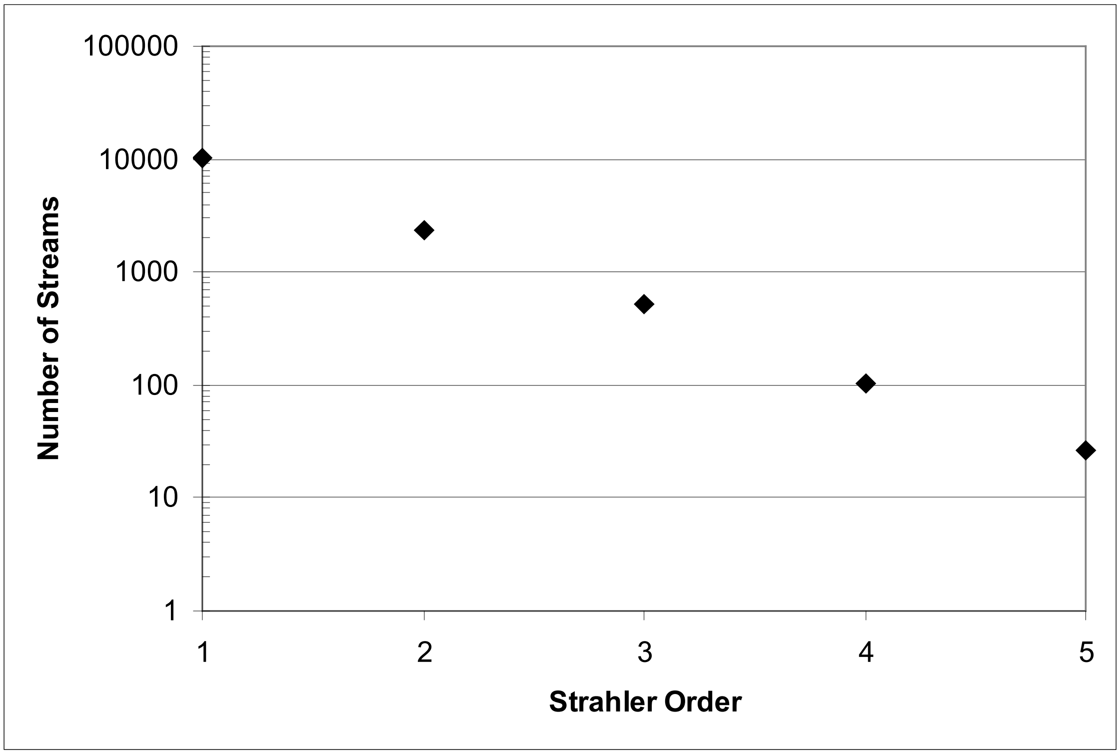

Figure 2.5. Stream frequency versus Strahler order for the entire study area.

Table 2.3. Principal component loadings for geomorphic variables extracted for streams, stream networks, and whole watersheds.

	Component 1	Component 2	Component 3	Component 4	Component 5
Length		-0.102		0.728	
Gradient		0.603	-0.359		
Outlet Elevation	-0.124	0.148	0.409	0.227	
Sinuosity	0.134		-0.154	0.624	
Network Slope		0.610	-0.350		
Cumulative Length	0.411				
Watershed Area	0.429		0.121		
Stream Density					0.996
Major axis	0.444				
Relief ratio	-0.239	0.258	0.465		
Average Hill Slope		0.365	0.538	0.110	
Maximum flow path length	0.391	0.105	0.115		
Perimeter	0.451				

In Equation 2.1, *CPU* is the number of seconds required to generate a hydrologically-corrected DEM, and A_{ws} is the watershed size (m^2) of the basin whose subwatershed boundaries were extracted.

Accuracy Assessment

The spatial accuracy of automatically derived watersheds underlies the reliability of these data for watershed analysis and modeling purposes. The log-transformed area between hand-digitized watershed boundaries and automatically derived boundaries was linearly related to log-transformed hand-digitized area (Figure 2.6; $R^2 = 0.97$, $p < 0.0001$). The regression equations relating two-dimensional error and watershed area can be rearranged to show that the proportion of a watershed that is incorrectly delineated may be expressed as a declining exponential function of watershed area (Equation 2.2a and 2.2b).

$$A_{error} = 18 * A^{0.60} \qquad \textit{Equation 2.2a}$$

$$p = \frac{A_{error}}{A} = 18 * A^{-0.40} \qquad \textit{Equation 2.2b}$$

The area between hand-digitized and automatically derived watershed boundaries is A_{error}, watershed area is A, and p is the proportion of watershed area containing two-dimensional errors due to automated processing. The average distance between automatic and digitized lines was equal to 17 m. A 50-m buffer around hand-delineated watersheds enclosed approximately 95% of the total lengths of automatically derived boundaries.

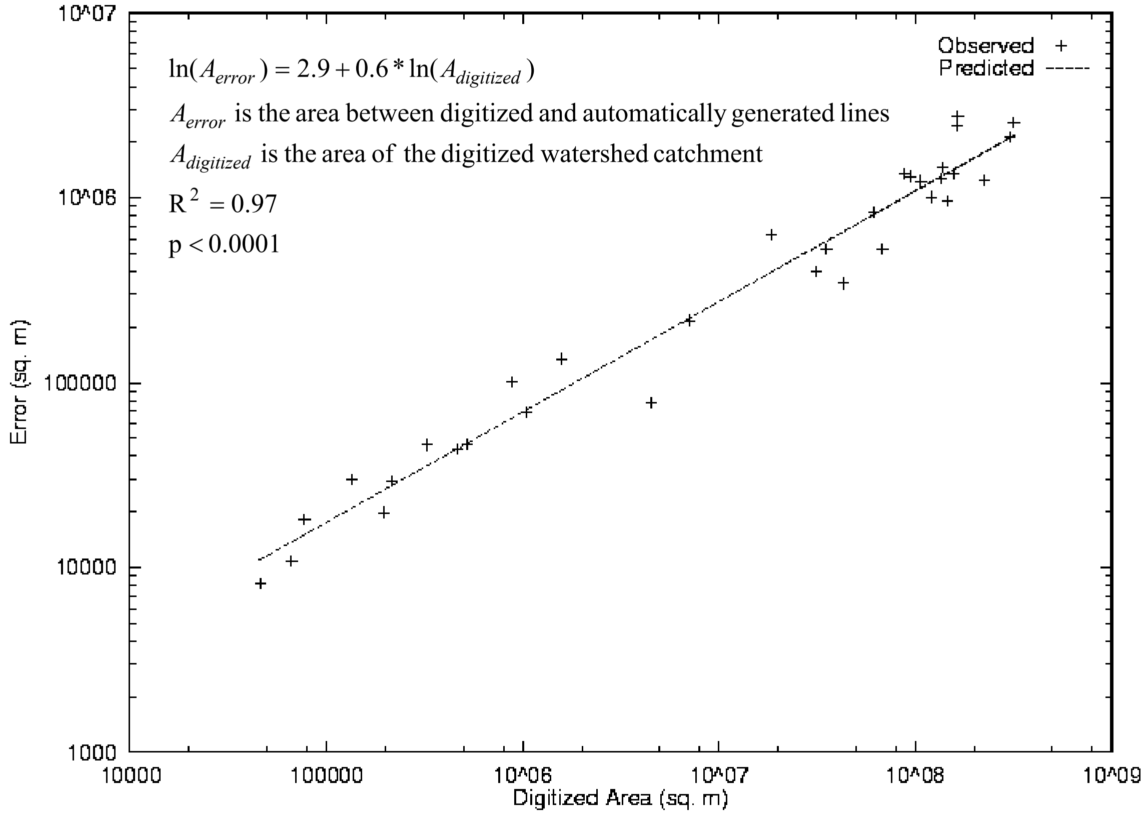

Figure 2.6. Relationship between total two-dimensional error and digitized area.

Discussion

Integrated databases of streams and automatically extracted watersheds hold much promise for stream ecological and geographical research by allowing water resource professionals to catalog and analyze streams and watersheds throughout large areas using GIS. Principal components analysis of watershed and stream attributes identified four groups expected to have distinctive habitats and biota. The first PCA axis described large streams and rivers with well-developed alluvial floodplains. The negative correlations with relief ratio and outlet elevation re-emphasize that interpretation since large watersheds would have low relief ratios and outlet elevations. The second component described high gradient streams with relatively high relief and steep hill slopes. The third component more specifically separated high elevation streams with high slopes and relief ratios but with moderate-sized watersheds and low gradients. Streams and watersheds positively correlated with Component 3 are distinct from the streams identified with Component 2 since those had steeper slopes. Component 4 separated the sinuous, long channels typical of large tributaries to the main stem of the river. The slightly positive loading with outlet elevation separated these streams from the main trunk of the stream which lies at the lowest elevations in the valley.

Because of the important role served by GIS databases of streams and watersheds, the limitations of those data deserve further explanation. When stream data are consistent, Strahler order (Strahler 1964) incorporates information about watershed area and surface drainage characteristics and is a powerful correlate for biological analyses (Naiman et al., 1991). However, inconsistent mapping methods among adjacent quadrangles compiled on different dates by different mapping analysts (Leopold, 1994) made the hydrography data from 1:24,000 quadrangle maps of limited use. Component 5 of the PCA analysis showed that stream density was not correlated with morphometric properties of streams or watersheds (see Table 2.3). Strahler (1964) emphasized that first order watersheds must be consistently identified in order for the laws of stream numbers, watershed areas (Horton, 1945; Shumm, 1956), stream lengths, and other morphometric properties to be valid. Objective, automated mapping techniques that use higher resolution elevation data could obviate deficiencies with 1:24,000 hydrography vectors observed here.

The average distance between automatically-derived and hand-digitized watershed boundaries was 17 m, which is equivalent to the radius of a circle with the same area as a single pixel with 30 m sides. The 17 m average displacement between sets of lines was determined to be acceptable since there is an inherent loss in precision stemming from using raster data to represent points and lines. While the average distance between the two sets of lines was 17 m, a buffer of 50 m around hand-digitized watersheds was sufficient to enclose 95% of the length of watershed boundaries delineated using automatic watershed delineation techniques. The 50-m buffer was required to encapsulate blunders in the automatic watershed delineation process. Two-dimensional errors were determined to be negligible for watersheds larger than about 250 ha. For a watershed with 95% of its area correctly delineated (p = 0.05, Equation 2.2b), A would be approximately 250 ha. Watersheds larger than 10 km^2 would have less than 3% two-dimensional error.

The raster data structure itself imposes some limits on DEMs and derived data. For example, slope measurements vary in discrete increments. The smallest detectable change in slope for a 30 m DEM is a 1 m change in elevation across a grid cell in the diagonal direction (Garbrecht and Martz, 2000).

$$l = 30 * \sqrt{2} = 42.4 \qquad \textit{Equation 2.3a}$$

$$\frac{1}{l} = \frac{1}{42.4} = 2.4\% \qquad \textit{Equation 2.3b}$$

where *l* is the diagonal distance across a 30 m pixel. However, 2.4% is large relative to channel slopes typically observed for streams in this study area. Slope measurements extracted from 30 m DEMs do not accurately estimate stream slope over short distances. Elevation data at 30-m resolution are not suited to analysis of local slope, which is an important correlate of fish and macroinvertebrate habitat quality (Walters et al. 2001).

Another problem introduced by using raster data for depicting elevation is that grid cells are usually large relative to a point, such as the intersection of two streams. Therefore a single point, such as the outlet of a stream where it intersects a downstream channel link, can have a contributing area in a raster representation (Band, 1989). Consequently, watershed polygons may have slivers, pixels with no clear drainage outlet, or include a large area that does not truly contribute to a stream or outlet's drainage. Evidence for these errors is given in Figure 2.6, which shows that this systematic source of error is not confined to small watersheds but rather is inherent in the automatic watershed delineation process. Further evidence for those blunders was the 50-m buffer needed to encapsulate 95% of the automatically derived watershed perimeter. Due to these topological errors, overlay operations are inadequate for relating stream arcs to watershed polygons. This problem was solved by (a) assigning pairs of arcs and polygons the same identification number, (b) selecting all stream arcs and their corresponding polygons using network functions, and (c) constructing a table consisting of properties of these selected groups of polygons. These concepts and the database design are shown in Figure 2.3.

One must consider the tradeoff between maximizing detail and maximizing accuracy in choosing the smallest watershed area to be delineated or analyzed. Maximizing detail allows one to consider small watersheds for hydrologic analyses while maximizing accuracy suggests some minimum watershed size below which analyses will be compromised by inaccuracies inherent in the watershed boundary identification process; 250 ha were necessary to obtain less than 5% two-dimensional error in this study. The average first order watershed in this database had an area of 75 ha, which translates to a two dimensional error of 8%. Using Level 1 30 m DEMs and hydrography from 1:24,000 scale topographic maps, the smallest watersheds would be expected to have two-dimensional errors on the order of 10%. To minimize error, the database of watersheds and streams could be reduced to the set with watershed areas greater than 250 ha.

Higher resolution DEM data are needed for watershed analysis of the smallest perennial streams in this study region, for they often have watershed areas below 10 ha (Swank and Crossley 1988). Much research in stream ecology has been conducted in streams that are not represented on 1:24,000 topographic maps (Meyer and Wallace 2001). For simulations, a sub-watershed area to pixel size ratio of 102 has been suggested as the minimum necessary to produce satisfactory model performance (Seybert 1996). That ratio implies a sub-watershed area of 9.2 ha if 30-m resolution DEMs are used. However, watersheds of that size would have two-dimensional errors of nearly

20% (see Equation 2b) which may be an unacceptable proportion of watershed size. For hydrologic modeling, Zhang and Montgomery (1996) suggested 10-m DEMs are needed for physically based process models, butplanimetric inaccuracies from watersheds derived from such data should also be assessed in the context of other watershed analyses.

Conclusion

Systematic methods were presented to build a database linking streams, watersheds, and watershed attributes across large areas using hydrography from 1:24,000 scale DLG and Level 1 30 m DEM (USGS, 1993) data. Principal components analysis separated four groups of watersheds corresponding to four PCA components: (1) low elevation streams with large watershed areas; (2) high gradient high elevation sites; (3) somewhat high elevation, low gradient streams with larger watersheds than those identified by Component 2; and (4) sinuous tributaries to the main stem of the river.

Hydrography data from 1:24,000 scale topographic maps were determined to be unreliable for identifying small streams and watersheds. This conclusion was supported by several lines of evidence. Many of Strahler's laws (1957, 1964) were not evident when hydrography data from quadrangle maps were compared to watershed morphometric properties. Stream density was not correlated with any morphometric characteristics such as relief ratio or average hill slope. These results underscore Leopold's (1994) observation that subjective methods in stream mapping make hydrography data from topographic maps of limited use for comparative analyses across regions spanning multiple map sheets.

Stream ecologists have called for the watershed network approach for many years (Hynes, 1975; Fisher, 1997). Data derived by the above methods will support statistical or mechanistic modeling efforts and will facilitate the whole-watershed and network perspectives in stream ecology.

Acknowledgments

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CHAPTER 3

MODELING SEDIMENT YIELD IN THE UPPER LITTLE TENNESSEE RIVER BASIN: SENSITIVITY AND APPLICATION OF THE REVISED UNIVERSAL SOIL LOSS EQUATION (RUSLE)[1]

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[1]Gardiner, E.P., J.L. Meyer, and W.O. McLarney. To be submitted to the Journal of the American Water Resources Association. An abridged version of this paper appeared as: Gardiner, E.P. and J.L. Meyer. 2001. Sensitivity of RUSLE to data resolution: modeling sediment delivery in the upper Little Tennessee River Basin. *Proceedings of the 2001 Georgia Water Resources Conference*, March 26-27, 2001, University of Georgia. Kathryn J. Hatcher, editor, Institute of Ecology, Athens, Georgia.

Abstract

A watershed sediment yield model was constructed using the Revised Universal Soil Loss Equation (RUSLE) in the upper Little Tennessee River basin in Macon County, North Carolina. When raster layers depicting elevation, land cover, and soil erodibility were coarsened in resolution from square pixels with 30 m sides to ones with 285 m sides, sub-catchment sediment yield estimates (tonne * yr^{-1}) differed across two orders of magnitude; results from the coarsest resolution captured less than 10% of the variance in model results from 30 m data. The relationship between model precision and soil data inputs was assessed using two common sources of soil data that have differing resolutions. State Soil Geographic (STATSGO) soil data provide coarser approximations of the information provided by Soil Survey Geographic (SSURGO) data. Model estimates of sub-catchment sediment yield based on STATSGO were statistically related to estimates based on SSURGO data when both sets of model results were log-transformed ($R^2 = 0.97$); however, estimates for individual sub-catchments often differed by orders of magnitude. Coarse scale land-cover data were simulated by aggregating cover factor raster layers from 45 m to 285 m in 15-m increments. When regressed against results obtained with 30-m data, model outputs that utilized coarsened land-cover data were weakly significant predictors, with R^2 values of approximately 0.50 among outputs from all resolutions. Sensitivity analyses suggested that model output is more sensitive to digital elevation model resolution than to the resolution of any other data layer. Coarsened digital elevation data produces lower estimates of sediment yield in areas with steep terrain, such as this study area. Model performance was assessed relative to calculated annual baseflow sediment yield based on measured total suspended solids (mg * l^{-1}) and discharge measurements taken at baseflow following storms. Modeled suspended sediment concentration did not agree with observed total suspended solids concentrations. Estimates of sediment yield that used SSURGO soil data were somewhat more accurate than model estimates of sediment yield that used STATSGO soil data, based on comparisons to calculated annual baseflow sediment yield (R^2 values of 0.69 and 0.54, respectively).

KEYWORDS: Hillslope sediment model, resolution, precision, water quality, geographic information systems (GIS), simulation, water resources geography, watershed management

Introduction

The Universal Soil Loss Equation (USLE) and the Revised USLE (RUSLE) are empirically-derived methods for assessing the complex process of soil erosion and loss from fields (Wischmeier and Smith, 1978; Renard, 1997). During the past two decades, variations of the USLE and RUSLE have been used to simulate soil loss and transport within watersheds and over large areas (Morgan and Nalepa, 1982; Banasik, 1986; Fernandes, 1994; McNulty and Sun, 1998). Such simulations have informed regulation of hillslope-derived sediment, such as several of the U.S. Environmental Protection Agency's sediment Total Maximum Daily Load (TMDL) documents for streams in Georgia for which model results have been generated with USLE implemented within Geographic Information Systems (GIS; Burrough, 1993) software (US EPA, 2000). The widespread use of GIS versions of the USLE and RUSLE necessitates a critical examination of their sensitivity and applicability for watershed assessments.

A debate recently emerged over whether USLE or RUSLE should be used in watershed assessments of soil loss and sediment yield (Trimble and Crosson, 2000; Nearing et al., 2000). Central to the debate were two issues: (a) application of the Soil Loss Equation to land uses or slope conditions for which it has not been extensively calibrated (Wischmeier, 1976); and (b) the inappropriate extension of a soil loss model to questions involving the transport and deposition of detached soil (Trimble and Crosson, 2000).

The first point has been addressed for field based estimation of soil loss through commentary (Wischmeier 1976) and calibration exercises (Risse et al., 1993; Liu et al., 1994) that examined plot level studies. The authors of the USLE suggested that model parameters may be applied in study areas other than locations where plot data have been collected, provided practitioners note and justify their choices (Wischmeier, 1976). The use of digital map data for input parameters rather than field assessments raises new concerns, however. The availability and structure of digital spatial data constrain the representation of USLE or RUSLE model parameters which may come from sources with varying resolutions and accuracies which may introduce systematic sources of error in model results. Quantifying the effects of varied data resolution on sediment yield estimates can assist watershed assessments of sediment yield.

The USLE and RUSLE models were originally developed for field-based estimation of soil loss, not for modeling sediment yield from watersheds using GIS (Trimble and Crosson, 2000). In this document, soil loss (tonne * yr^{-1}) refers to the process by which soil is detached and moved off a plot. Soil loss contrasts with sediment yield (tonne * yr^{-1}), the total mass of eroded soil that is delivered via a stream. The latter implies overland transport of eroded soil to waterways. In the field, the analyst may examine a specific plot, noting variations in terrain such as changes in slope or gullies that alter erosive potential or capacity for transport of sediment off the plot. Cropping patterns and conservation practices may also be noted when in the field (Renard, 1997). The USLE and RUSLE models stop at the edge of a field and have not been calibrated with whole-watershed estimates of sediment yield. The number and configuration of depositional and erosional areas throughout a watershed can be vast and difficult to ascertain; those terrain features modify overland sediment transport, so soil-loss estimates from all parcels should not be simply added to estimate sediment yield from a watershed (Trimble and Crosson, 2000).

This research examines the two issues introduced above. A watershed sediment yield model is developed that implements RUSLE in a GIS to estimate soil loss throughout entire watersheds and hillslope-derived sediment yield from streams at their outlet on an annual basis. The first objective is to quantify the variability and magnitude of model outputs as functions of input data resolution. This objective is addressed with three sets of analyses by: (a) coarsening all data layers simultaneously; (b) using two commonly used sources of soils information that differ in spatial resolution; and (c) coarsening land-cover data only. The second objective is to evaluate whether the model can be used to predict sediment yield or in-stream sediment concentrations (mg * l^{-1}). To meet that objective, model results are compared to observed sediment concentrations and calculations of annual baseflow sediment yield, i.e., mass of sediment leaving a watershed under baseflow conditions.

Study Area and Methods

Data, methods, and results described below pertain to a 1200 km^2 portion of the upper Little Tennessee River basin (Figure 3.1). Most of the basin lies within western North Carolina although a small, southerly portion lies in Rabun County, Georgia. Franklin, N.C. occupies the center of the basin. Highlands, N.C. sits in the headwaters of the Cullasaja River, a tributary of the Little Tennessee River that drains the eastern portion of the study area. Terrain varies from steep headwater reaches to a broad alluvial valley that contains the main stem of the river. Elevation varies between 550 and 1650 m and has a mean value of 880 m.

The region has been the focus of numerous studies that have examined stream ecosystems (e.g. Webster et al., 1991; Bolstad and Swank 1997), land ownership patterns

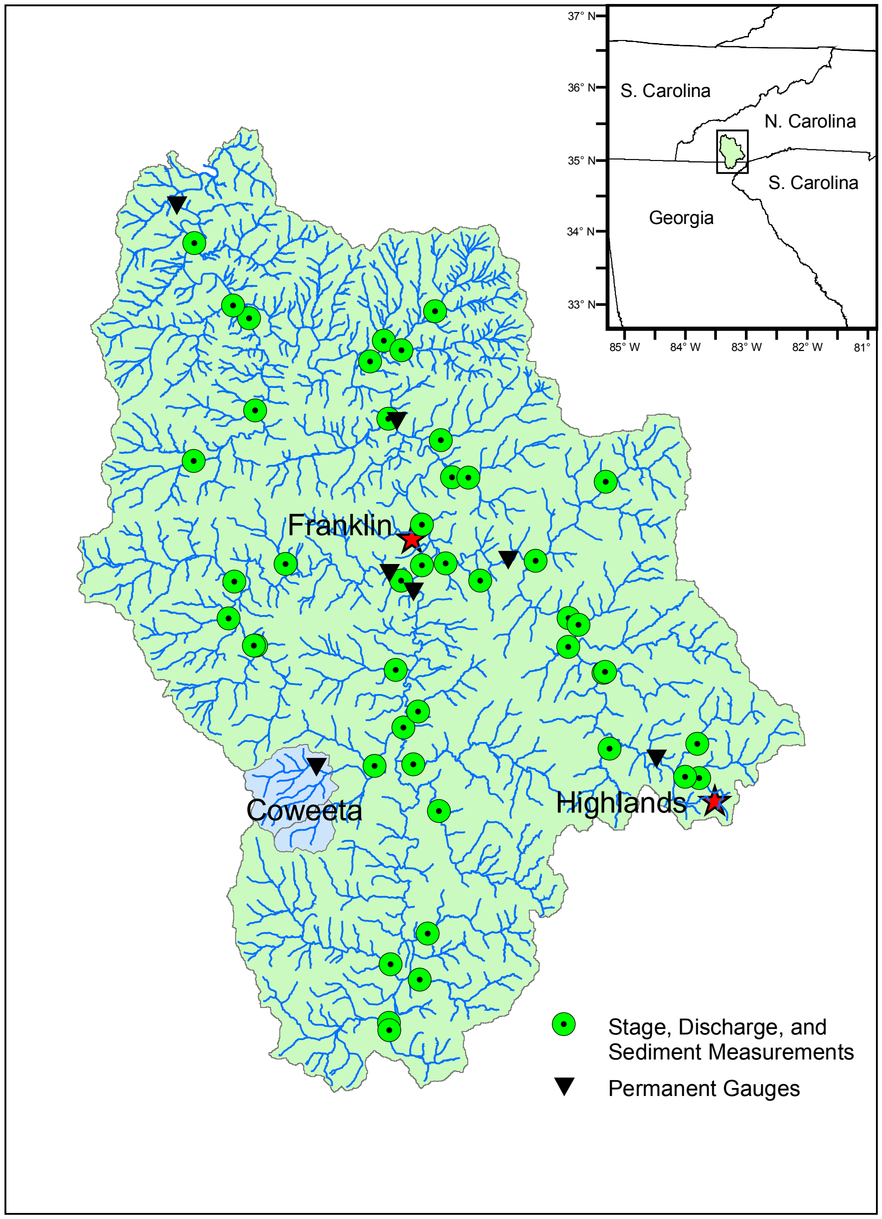

Figure 3.1. This research was conducted in the upper Little Tennessee River basin in western North Carolina. Seven permanent hydrologic gauges (closed triangles) were located throughout the study area. Stage and discharge relationships were derived for 45 stream sites (closed circles) throughout the basin. Annual baseflow sediment yields were calculated from stage and total suspended solids measurements at the same 45 sites.

(Turner et al., 1996), and land use (Wear and Bolstad, 1998). Land cover from the Southern Appalachian Assessment indicates the study area was 89% forested, 9% agriculture, and 2% developed, according to interpreted Landsat-5 imagery acquired in 1993 (Hermann, 1996). The areas that have been cleared for agriculture or other development are concentrated at lower elevations near roads, whereas headwaters are mostly forested.

The focal study basin of the Coweeta Long Term Ecological Research program lies in the southern portion of the study area. Researchers at Coweeta were among the first to link improper logging practices with geomorphological changes in small streams and hence to fish habitat (Tebo, 1955, 1957; Waters, 1995). Subsequent effort closely examined road design, use, and sediment delivery under storm conditions (Webster et al., 1983; Swift, 1984; Swift, 1986; Swift, 1988). Research at Coweeta has also linked human land use and management practices to a wide variety of modifications to terrestrial and aquatic habitats throughout the southern Appalachians.

Soil Loss and Sediment Yield Estimation

Soil loss was estimated using a modified version of RUSLE (Renard, 1997), whose formula is identical to the USLE and is presented in Equation 3.1 below.

$$A = R*K*L*S*C*P \qquad \textit{Equation 3.1}$$

Soil loss, A (ton * acre^{-1} * yr^{-1}), is a function of six factors whose interpretations and parameter ranges were described by the model developers (Wischmeier and Smith, 1978; Renard, 1997). The units for soil loss estimates, A, were converted from English to SI units and multiplied by the area of pixels to provide soil loss estimates (tonne * yr^{-1}). Erosivity is represented by the parameter R (ft * tonf * in * ac^{-1} * yr^{-1}), an erosion index that represents climatic factors; the *R* parameter was held constant at 200 for all simulations based on the isopleth map of annual soil-loss potential found in Renard (1997). The soil erodibility factor, *K* (ton * ac * $acft^{-1}$ * $tonf^{-1}$ * in^{-1}), is the annual per area soil loss per erosion index unit, *R*. The remaining dimensionless factors (*L, S, C,* and *P)* represent the effects of slope length, slope angle, land cover, and mitigation of soil erosion through conservation practices, respectively. Each is a unitless proportion representing soil loss from a land parcel relative to that observable on a unit plot, defined as continuously fallow and clean-tilled with a length of 22.1 m and a slope of 9% (Wischmeier and Smith, 1978; Renard, 1997).

Soils data came from high resolution and low resolution sources. Paper maps of the Macon County Soil Survey were digitized by the Macon County Mapping Department and attributed with *K*-values published with the survey (Devereux et al., 1929). This type of high resolution, county soil survey data are termed Soil Survey Geographic Databases, or SSURGO, of the US Department of Agriculture (USDA, 1991). Coarser scale State Soil Geographic, or STATSGO, data are spatially-weighted averages of *K*-values within polygons mapped at 1:250,000 scale (USDA, 1991). Detailed SSURGO data were available only for Macon County (courtesy Macon County Mapping Department) while STATSGO data covered the entire basin. Both SSURGO and STATSGO data (USDA, 1991) were converted to 30-m raster grids before being used to estimate soil loss and delivery to streams within the study area.

The *L* and *S* parameters were derived from Level 1 U.S. Geological Survey digital elevation models (USGS 1993). The *L* factor in RUSLE is calculated from slope length (λ), the ratio of rill:interrill erosion, and local slope. Slope length (λ) was set equal to 30 m for flow normal to the cell orientation and 42.4 m for diagonal flow. An intermediate ratio of rill:interrill erosion was assumed because most of the study area is forested with little concentrated overland flow expected. Parameter values for intermediate rill:interrill erosion rates were taken from the RUSLE manual (Renard 1997). Local slope is essential in the calculation of the *L* and *S* factors. Slope data were derived from the digital elevation models using ESRI Arc/Info software (ESRI 1998).

Cover factors were taken from Wischmeier and Smith (1978) or from the US Environmental Protection Agency's planning documents for Stekoa Creek, Georgia (US EPA 2000). Eight cover factors and *C*-values were used and are provided in Table 3.1. The *C*-factors correspond to land-cover classes extracted from classified satellite images and from other map data. Gravel roads were digitized from paper or scanned topographic maps at a scale of 1:24,000. Pixels in the land-cover image that contained gravel roads were assigned *C*-factors based on the assumption that roads were 10 m wide. Conservation practices, represented by the *P* factor, were set equal to "1" for all model simulations since data describing practice factors were not available.

Soil-loss estimates cannot be simply added in the downslope direction because decreases in slope can lead to reduced transport capacity and deposition of particles (Wischmeier and Smith, 1978; Renard, 1997). Because it was designed to estimate soil loss on agricultural plots, RUSLE does not include a function for hydrologic transport of detached soil particles. An expression for maximum transport capacity was derived from work by Moore and Wilson (1992), who found that the combined *L*S* factor in RUSLE was closely related to several formulations of dimensionless overland sediment transport capacity. The combined *L*S* factor is dimensionless, so multiplication by the erosion

Table 3.1. Land-cover classes and *C*-factors used in this version of RUSLE. *C*-factors describe the effects of land cover on RUSLE estimates of soil loss.

Cover	C Factor
Mixed Forest	0.002
Evergreen Forest	0.001
Deciduous	0.003
Rangeland	0.003
Row crop	0.12
Developed	0.003
Water	0
Gravel Roads	0.75

index, *R,* provides an estimate of maximum erosive potential (Moore and Wilson 1992). The maximum allowable soil loss for a given cell was set to the quantity $R*L*S$, an expression analogous to a soil with no resistance to erosion and cover and practice factors that do not mitigate erosive potential in any way. If the accumulated soil loss at any cell exceeded that maximum allowable soil loss, only that maximum was delivered down slope (Equation 3.2a). The relationships among soil loss, yield from direct source areas, and whole-watershed sediment yield are summarized in Equations 3.2a-c and Figure 3.2.

$$D_k = \begin{cases} (R*L*S)_k \text{ if } \sum_l U_l > (R*L*S)_k \\ \sum_l U_l \text{ if} \sum_l U_l \le (R*L*S)_k \end{cases} \qquad \textit{Equation 3.2a}$$

$$DSA_j = \sum_k D_k \qquad \textit{Equation 3.2b}$$

$$Y_i = \sum_j DSA_j = \sum_j \sum_k D_k \qquad \textit{Equation 3.2c}$$

The annual mass of soil contributed to the next pixel down slope from pixel k is D_k. The factors *R, L,* and *S* are as defined in Equation 3.1. The RUSLE soil-loss estimate upslope of and including cell k is U_l. Sub-catchment sediment yield is DSA_j, and yield within the whole watershed is denoted by Y_i.

Sediment yield was estimated for each stream segment in the drainage network taken from digitized 1:24,000 scale topographic maps and digital line graphs. Stream segments were defined as the portions of streams between upstream and downstream junctions with other river reaches (Figure 3.2). Meander cut-offs and braided channels were manually removed from the digital database. Direct source areas were defined as the portions of a watershed contributing hydrologically to each individual stream segment

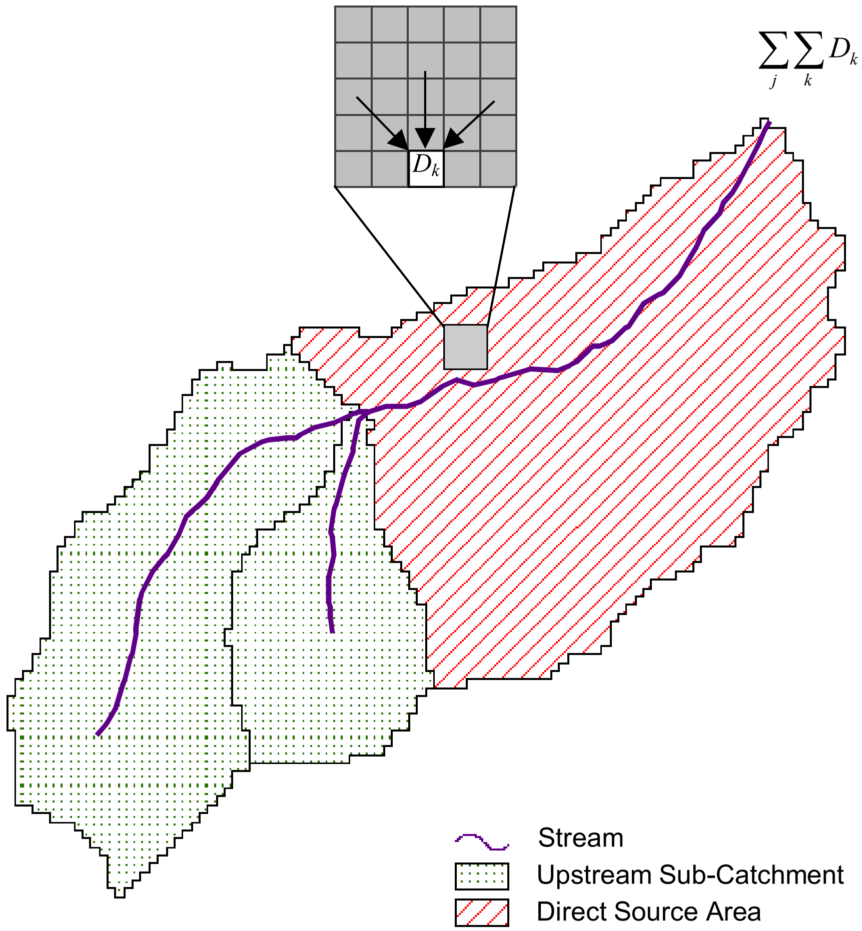

Figure 3.2. Stream segments were defined as the stream section between upstream and downstream junctions. Direct source area was defined as the area contributing to a stream section of interest but not upstream of the upstream stream junction. The entire watershed was therefore the union of all direct source areas upstream of and including the stream segment of interest. Sediment yield estimates (D_k, tonne * yr^{-1}) were calculated for every grid cell in each direct source area. An arbitrary group of 25 cells is shown at the top of the figure to demonstrate how sediment yield estimates were accumulated down slope and summed for every pixel in each direct source area. See text for details.

exclusive of the area upstream of its upstream node (Figure 3.2). Direct source areas and catchment boundaries had been partitioned using methods described in Chapter 2. Sensitivity analyses were conducted using sediment yield from all direct source areas in the study area.

Sensitivity Analyses

A series of sensitivity analyses was performed based on the modeled sediment yield to 960 direct source areas throughout the study area; see Figure 3.2 for the definition of direct source areas. One set of simulations compared model results from data represented by 30-m resolution grids to output from simulated lower resolution data. In these analyses, *K*-factors were taken from the Macon County Soil Survey (Devereux et al., 1929; digital data courtesy Macon County Mapping Department). These "reference model" results were based on the highest resolution data available and were thus considered to be the most reliable of the scenarios examined. Land-cover data for the reference model were taken from Hermann (1996). Subsequently, each data layer was simultaneously aggregated from 45 m to 285 m in 15 m increments, and the sediment yield model was run for each level of aggregation. Model outputs of sediment yield to direct source areas for each coarser resolution were regressed against the reference model outputs for those direct source areas.

Model results from coarser resolution data were compared to reference model results using linear regression and a logit transformation. From each of the 17 regressions comparing outputs from models run with successively lower resolution data, R^2 was recorded. The logit link function was used to fit a model of the following form:

$$\ln\left(\frac{R^2}{1-R^2}\right) = \beta_1 + \beta_2 * P_l \qquad \textit{Equation 3.3}$$

where P_l is the length of one side of a pixel for the dimension being analyzed and β_1 and β_2 are coefficients estimated through regression. In this way, R^2 values from each of the 17 linear regressions were expressed as a function of the pixel sizes of the data used to obtain that value. Slope and intercept parameters from the 17 independent linear regressions were also regressed against pixel sizes to quantify the changes in shape of regression functions with coarser resolution data.

The second sensitivity analysis quantified the effect of low resolution soil data on model output. Linear regression was used to compare the log-transformed model output obtained with lower resolution STATSGO data to the log-transformed model output obtained with SSURGO data. For this analysis, data layers for parameters in Equation 3.1 were represented by 30-m resolution grids.

The third sensitivity analysis examined the independent effect of land-cover resolution on model estimates. The model was run using land-cover factors aggregated from 30 m to 285 m in 15 m increments. Other layers remained at their original 30-m resolution. Linear regression was used to compare each of the coarser resolution outputs to the reference data results. Each of these sensitivity analyses quantified the loss in precision associated with degraded input data resolution.

Model Comparisons to Sediment Yield and Concentration

Model results were also compared to sediment concentration and discharge data collected in 1998. Land cover was mapped from a 1999 Landsat-7 satellite image, and *C*-factors were assigned to the resulting classes (see Table 3.1). Soil *K*-factors were taken from both SSURGO and STATSGO, and *L* and *S* factors were taken from USGS Level 1 digital elevation models, as described above. In situ measures of sediment concentration and discharge were used to estimate average baseflow sediment concentrations and annual baseflow sediment yield from 45 streams identified in Figure 3.1. Those calculated sediment concentration and yield data were compared to model estimates of watershed sediment yield and concentration. These steps are described in detail below.

Land Cover

Land-cover information was derived from a Landsat-7 image acquired on October 5, 1999. selected because clouds and cloud shadows occupied less than 1% of the study area. Processing steps included the following: (a) satellite image rectification; (b) creation of a high resolution orthoimage from aerial photographs for reference land-cover information; (c) training set identification and supervised classification; and (d) accuracy assessment of the classified Landsat image.

Rectification is the process by which spatial data are geometrically transformed so coordinates in two spatial data sets, for example imagery or vector data, refer to the same ground coordinates in a specified datum. Landsat-7 and National Aerial Photography Program (NAPP) images were rectified to the Universal Transverse Mercator Zone 17 grid, North American Datum of 1983, using ground control from 1:24,000 scale USGS topographic quadrangles and digital raster graphics. The rectified Landsat image had geometric errors of less than one pixel root mean-squred error ($RMSE_{x,y} \pm 26$ m).

Twenty-seven color infrared NAPP photographs from 1998 were scanned at 800 dpi (1.4 m nominal pixel resolution) and submitted to an aerotriangulation process in order to densify the control point network and for subsequent orthorectification. Aerotriangulation was performed using Desktop Mapping System (DMS; Welch, 1989) and PC Giant software packages (GPA Associates, 1994). The aerotriangulation provided RMSE values of 1.1 m (X), 0.68 m (Y), and 1.4 m (Z). Single photo resections of each of the 27 photos were conducted with DMS (Welch, 1989). Orthorectification of the NAPP photographs was performed in conjunction with USGS Level 1 digital elevation models. The NAPP orthoimages were characterized by RMSE values of ± 3-5 m on all photos.

Training sets for use in a supervised classification of the Landsat-7 image were digitized from the orthorectified NAPP images using ESRI Arcview or ERDAS Imagine software (ESRI, 1998; ERDAS, 1999). Seven classes of land cover (Table 3.1) were mapped using maximum likelihood supervised classification (Lillesand and Kiefer, 1994; ERDAS, 1999) in Imagine. Clouds and cloud shadows occupied a small portion of the scene and obscured mixed forest areas, so these classes were dissolved. Land-cover classes were identified with methods detailed by the National Land-Cover Data Classification System of the Multiple Resolution Land Cover Consortium (Vogelmann and Wickham, 2000).

An accuracy assessment was performed by comparing land cover in the classified Landsat-7 image to land cover observed in the orthorectified NAPPs. Imagine's random-point generator was used to locate approximately 50 points in each of the seven land-cover classes. When random points fell outside the area covered by the orthorectified NAPP imagery, those points were discarded. The reduced set included 318 stratified-random points. A 30 x 30 m square was created with its center at each point, and land cover was recorded from the orthorectified NAPPs using criteria described by Vogelmann and Wickham (2000). The land-cover image was used in subsequent RUSLE model scenarios and in regional discharge analyses.

Sediment Concentration and Yield

Instantaneous and annual sediment yield estimates were calculated based on observed river stage and observed sediment concentrations at the 45 streams in Figure 3.1 (McLarney, unpublished data). Stage and discharge sampling covered a range of discharges expected under baseflow conditions. Velocity and depth were recorded along cross sections throughout 1997 and 1998 at each stream. Velocity was multiplied by cross-sectional area to calculate instantaneous discharge. On each sampling date, river stage was recorded from the top of a semi-permanent stake. Discharge calculations were regressed against recorded river stages to derive site-specific equations relating stage and discharge using a standard formula (Gordon et al., 1992):

$$Q = \beta_3(H - H_0)^{\beta_4} \qquad \textit{Equation 3.4}$$

In this equation, Q is discharge ($m^3 * s^{-1}$), β_3 and β_4 are estimated parameters, H is river stage (m) at the time of measurement, and H_0 is the zero point for the datum used at that stream, i.e. when discharge would be 0.

Annual baseflow sediment yield and average baseflow concentrations were calculated from data collected on at least 10 dates in 1998 at the same 45 sites. Instantaneous sediment yield was calculated from total suspended solids measurements multiplied by discharge at the time of data collection. Total suspended solids (TSS) concentrations (mg/l) were measured in water samples that were transported to a laboratory, filtered onto pre-ashed, pre-weighed 47 mm diameter, 0.45 μm mesh glass fiber filters, placed in aluminium foil envelopes, dried in a 105° C oven for 24 hours, and weighed. Discharge was calculated from river stage at the time samples were collected. If stage was not recorded when grab samples were taken, discharge was assumed to be the mean of the previous and next calculated discharges. This occurred for less than 10% of the samples. Total suspended solids concentration was multiplied by discharge and the time interval between successive samples to interpolate sediment yield at baseflow by each stream between sample dates. Annual baseflow sediment yield was calculated by summing these estimates. An example is provided from Wayah Creek in the western central portion of the study area (Figure 3.3).

Calculated annual baseflow sediment yield and observed TSS concentration data were compared to model estimates of annual sediment yield and suspended solids concentration (SSC, mg/l), respectively. Hereafter, TSS will refer to measured values and SSC will refer to modeled concentrations. Model predictions of sediment yield from RUSLE were summed for stream segments upstream of the 45 sites where TSS and discharge were measured in 1998. The SSC data were estimated using the following formula:

$$SSC_i = \frac{\sum_j \sum_k D_k}{Q_{bf_i}} \qquad \textit{Equation 3.5}$$

where SSC_i represents suspended sediment concentration (mg/l) in stream segment i, D_k is the soil loss (see Equation 3.2) in all pixels (k) in all subcatchments (j) upstream of and including stream segment i. Bankfull discharge at each stream segment i, Qbf_i, was estimated using two methods. The first was based on regional curves relating flood

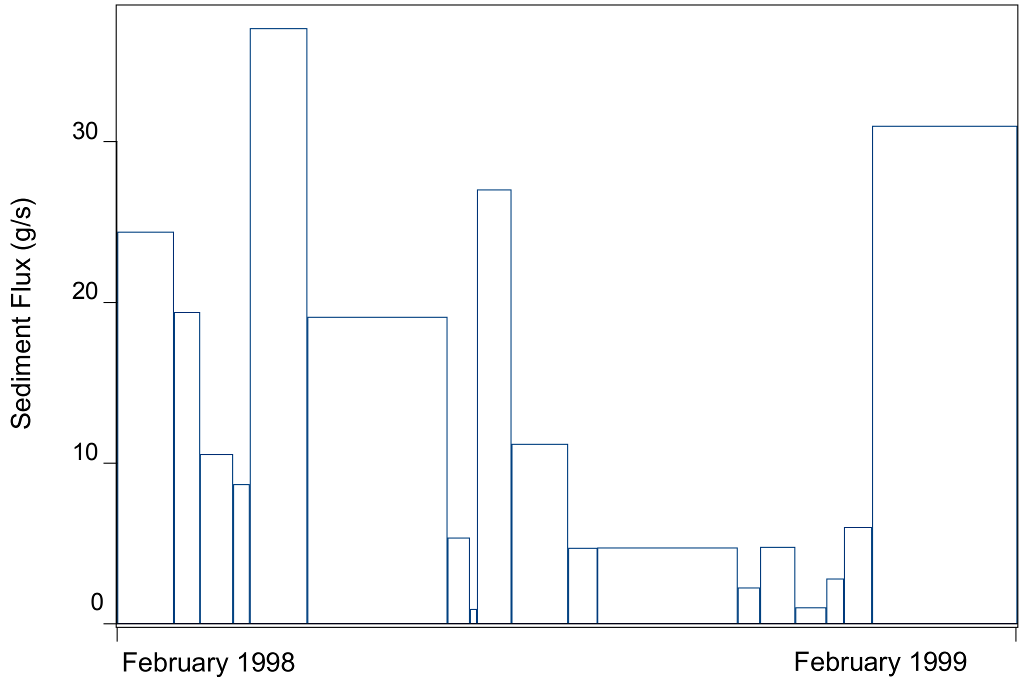

Figure 3.3. The rate of baseflow sediment yield at Wayah Creek during 1998 based on TSS concentrations measured from grab samples multiplied by discharge estimates from stage-discharge relationships. At all sites where data were sufficient, TSS was multiplied by the time interval between sampling dates and the products were summed to estimate annual baseflow sediment yield.

discharges to watershed area; the second used multiple regression to estimate discharge based on watershed area and land-use characteristics. After comparing results from the two methods, the first was used to estimate Q_{bfi} in Equation 3.5.

Regional curves were derived to express discharge as an exponential function of watershed area using methods outlined by the US Water Resources Council (1981). Peak annual flood data were taken from six US Geological Survey (USGS) gauges and from one gauge at the US Forest Service Coweeta Hydrologic Laboratory. Gauge locations are shown in Figure 3.1. The flood events analyzed were for 1.01- (annual probability of recurrence, p=0.99), 1.67- (p=0.6), 3.3- (p=0.3), and 10-year (p=0.1) recurrence intervals. Slopes among each of the five regressions were compared under the null hypothesis that they were equal. The regional curve analysis provided estimates of stream discharge based on watershed area for a variety of flow conditions.

Multiple regression was used to refine estimates of average baseflow discharge. Area-weighted dimensionless runoff coefficients were computed for each site where stage and discharge had been measured (Appendix 3A). Runoff coefficients incorporate information about the infiltration capacity of unique combinations of land cover or land-use and soil hydric potentials throughout each watershed (McCuen, 1998). Watersheds with relatively high impervious surface areas were expected to have lower observed baseflows than those predicted from regional curves. These same streams would have relatively greater quick flow in response to rain events and attendant geomorphological characteristics of urbanizing streams (Booth and Jackson, 1997). Watersheds with high soil water capacity should have higher baseflow discharge than watersheds with more impervious surface area. Runoff coefficients and watershed area were used in multiple regression to predict mean discharges observed under baseflow conditions.

Results

A sediment yield model based on RUSLE and three tests of its sensitivity and efficacy are first described. The model's sensitivity to input data was quantified by comparisons to results from a reference model based on the best available input data. Model estimates of sediment yield from 45 catchments were compared to calculated sediment yield data based on field observations. Finally, suspended sediment concentration was estimated using model results and the better of two formulae representing stream discharge. This representation of suspended sediment concentration was compared to observed total suspended solids measurements.

Sensitivity Analyses

When pixel size was increased simultaneously for the digital elevation model, soil, and land-cover factor data layers, model predictions for sediment yield were lower (Figure 3.4). Both the intercept (intercept = 9.0 * d, $R^2 = 0.94$) and slope (slope = 0.05 * d, $R^2 = 0.80$) of the regressions of reference model output vs. coarsened model output (e.g. Figure 3.4.) increased with pixel size (d).

The strength of the relationship (R^2) between model outputs from reference vs. coarsened data grew weaker with larger pixel sizes. The relationship between R^2 and pixel size was sigmoidal, highlighting a threshold near 90 m; beyond this point there was little correspondence between model outputs based on reference data and coarser resolution data (Figure 3.5). For example, model output from cells larger than 180 m x 180 m explained less than 20% of the variation in reference model output.

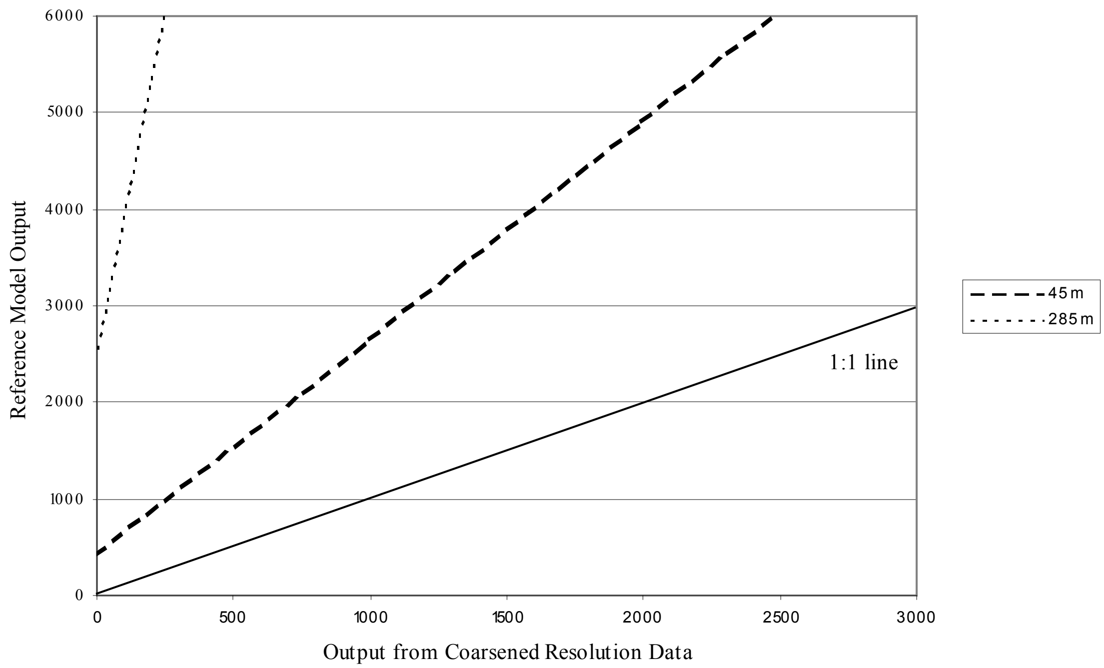

Figure 3.4. When input data resolutions were coarsened, sediment yield estimates were always lower than estimates from the reference model that used 30 m resolution data. This graph shows the regression lines obtained when 45 m data (long dashes) were used to estimate reference model output and when 285 m data were used (dotted line).

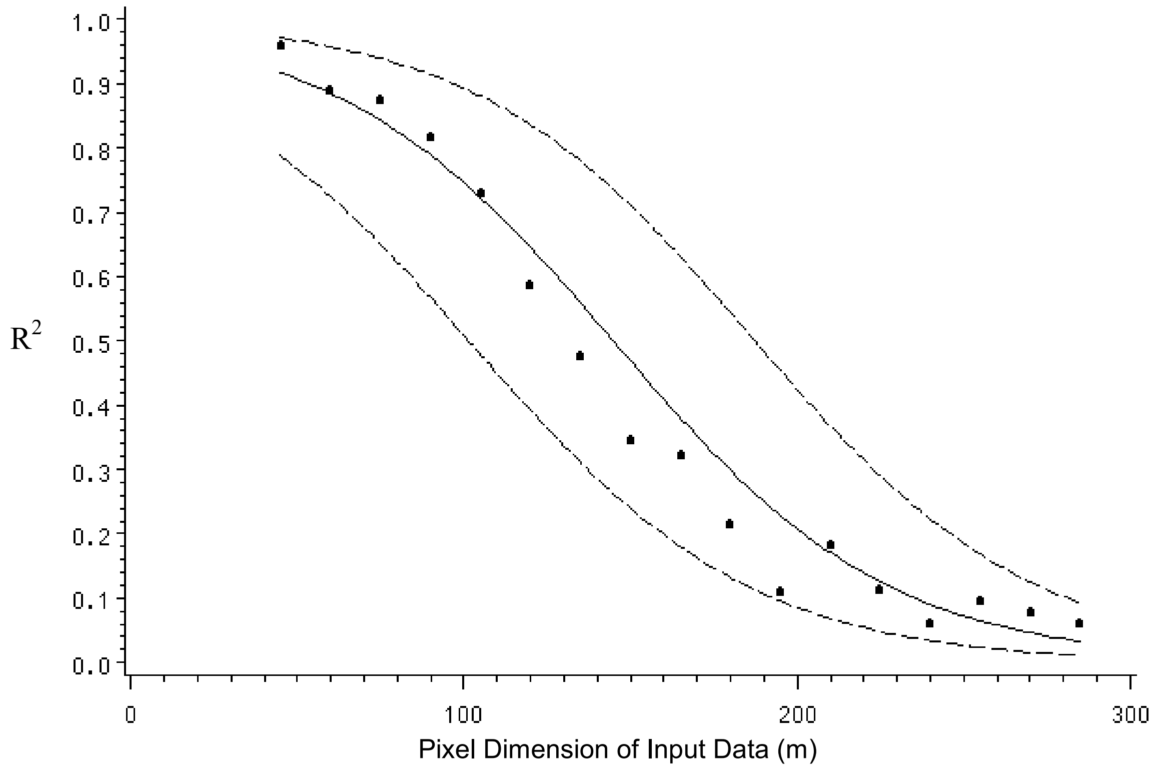

Figure 3.5. Pixel dimension explained 94% of the variance in logit-transformed R^2 values obtained by regressing direct source area sediment yield estimates based on lower-resolution input data against reference model output. Each point is the result of a regression based on 960 points. Confidence intervals, ±95%, are shown with dashed lines. This sigmoidal function illustrates a threshold near 90 m beyond which explanatory power dropped rapidly.

Contrary to the trend observed when the resolutions of all input data were simultaneously decreased, lower resolution soil and land-cover data produced higher predictions of sediment delivery to streams, relative to reference model results. On a pixel by pixel basis, STATSGO*K* -values were generally higher than SSURGO data when each was represented by 30 m pixels; therefore the model using STATSGO data produced higher soil-loss and sediment yield values than the reference model. Those two sets of model output were significantly related ($R^2 = 0.97$) on log-log scales, but there was considerable scatter in the relationship, as was evident when data were back-transformed and displayed on plots with linear axes (Figure 3.6).

Land cover analyses provided a similar set of results. Aggregated land-cover factors (*C* in Equation 3.1) were larger, on average, than cover factors mapped with 30 m pixels. These aggregated land-cover factors led to sediment yield predictions that were higher and that had a great deal of unexplained variance relative to the reference model output. Lower resolution model output was a weak predictor of reference model results: explanatory power (R^2) values ranged from 0.48 to 0.50, and slope coefficients ranged from 0.31 to 0.32 for all sets of comparisons. Loss in precision was considerable in aggregating land cover from 30 m to 45 m but was not substantially worsened by further aggregation of land-cover factors.

The simultaneous aggregation of digital elevation, soils, and land-cover data layers produced lower predictions of sediment yield despite the fact that lower resolution soil and land-cover data each provided higher estimates of sediment yield. By inference, coarsened digital elevation model data decreased model predictions to a larger extent than the observed increases associated with land-cover and soil information.

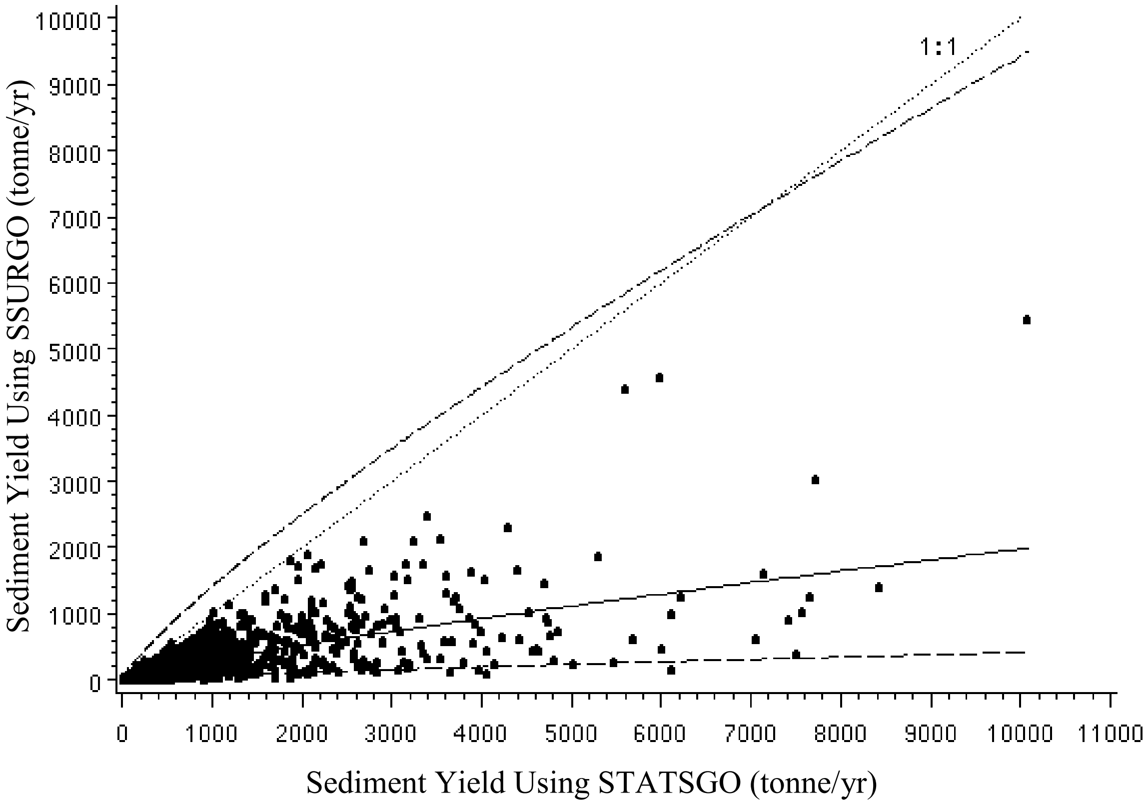

Figure 3.6. Log-log regression results were highly significant and powerful when sediment yield estimates based on STATSGO soils data were compared to estimates that used SSURGO data ($R^2 = 0.97$; $p < 0.0001$; $n = 960$). Dashed lines represent the upper and lower 95% confidence intervals for the regression. The expected 1:1 relationship is shown as a fine dotted line. Data were back-transformed and plotted on linear axes to emphasize that individual estimates of sediment yield differ dramatically when STATSGO soils data are used instead of SSURGO soils data.

Sediment Concentration and Yield

Before considering the model's ability to describe observed stream sediment concentration and yield, a brief description of the land-cover data derived for 1999 is required. The classified image had an overall accuracy of 77% for the seven land-cover classes (Table 3.2). Much of the confusion in the classification was attributable to the three forest classes. The deciduous class had producer's and user's accuracies of less than 60% (Lillesand and Kiefer, 1994). Higher accuracy may have been obtained with multidate imagery that would have highlighted phenological changes in deciduous vegetation. On the date this image was acquired, leaves had fallen from species whose leave fall early, e.g. tulip poplar (*Liriodendron tulipifera*). Other species of trees were in various stages of leaf fall and color change. The minimum producer or user accuracy within 30 m x 30 m analysis windows was 56% for the deciduous forest category (Table 3.2a). Deciduous cover was frequently confused with evergreen (28%) and mixed (12%) forest types. Aggregating land-cover classes improved the overall accuracy of the map. By collapsing the forest categories into one and by aggregating crop and pasture/grass classes, overall accuracy was boosted to 92% (Table 3.2b).

Sediment yield estimates were compared to annual baseflow sediment yields calculated for each stream where stage, discharge, and TSS had been measured in 1998. To calculate values for annual baseflow sediment yield required stage-discharge curves. River stage was a good predictor of discharge under a range of typical baseflow conditions at most of the 45 sites where stage and discharge were measured simultaneously. All but two regressions were highly significant. Of the remaining 43 significant regressions, only four had R^2 values below 0.8. For 14 streams, R^2 values

Table 3.2. Accuracy assessment results for 1999 Landsat-7 image interpretation. The first part of the table (a) shows the error matrix obtained by recording the observed land cover (columns) at random points within each land-cover class in the classified image (rows). Results are summarized according to producer's (proportion correct among number observed in each class) and user's (proportion correct among number classified in each class) accuracy. The second part of the table (b) provides producer's and user's accuracies obtained by merging the three forested categories and the two agricultural and grass categories.

(a)

			Observed Land Cover							Row	User's
			1	**2**	**3**	**4**	**5**	**6**	**7**	Total	Accuracy
Classified Land Cover	Mixed Forest	**1**	36	4	5	2	1	2	0	50	0.72
	Evergreen	**2**	6	27	9	0	0	0	0	42	0.64
	Deciduous	**3**	5	12	24	0	0	2	0	43	0.56
	Pasture	**4**	0	0	2	30	2	3	0	37	0.81
	Crop	**5**	0	0	1	5	33	7	0	46	0.72
	Developed	**6**	0	1	1	2	0	46	0	50	0.92
	Water	**7**	0	0	0	0	0	0	50	50	1.00
	Column		47	44	42	39	36	60	50	318	
	Producer's Accuracy		0.77	0.61	0.57	0.77	0.92	0.77	1.00		Overall 0.77

(b)

Super Groups	**Producer's**	**User's**
Forest	0.96	0.95
Crop, Pasture, and Grass	0.93	0.84
Developed	0.77	0.92
Water	1.00	1.00
Overall	0.92	

ranged from 0.80 to 0.90. The remaining 25 streams had stage-discharge relationships with R^2 values between 0.90 and 1.00 (Table 3.3). River stage recorded in the field was a dependable predictor of baseflow discharge.

Discharge-area regressions for 1.01-, 1.67-, 2-, 3.3-, and 10-year recurrence interval floods were statistically significant and powerful ($0.0002 \leq p \leq 0.0004$, $0.93 \leq R^2 \leq 0.95$). Each regression was performed on log-transformed data, yielding power law relationships similar to Equation 3.6a for each recurrence interval examined.

$$Q = \beta_5 * WS^{0.9} \qquad \textit{Equation 3.6a}$$
$$Q_{1.01} = 1.49 * WS^{0.9} \qquad \textit{Equation 3.6b}$$
$$Q_{1.67} = 2.09 * WS^{0.9} \qquad \textit{Equation 3.6c}$$
$$Q_2 = 2.20 * WS^{0.9} \qquad \textit{Equation 3.6d}$$
$$Q_{3.3} = 2.46 * WS^{0.9} \qquad \textit{Equation 3.6e}$$
$$Q_{10} = 2.95 * WS^{0.9} \qquad \textit{Equation 3.6f}$$

In these equations, Q represents discharge in $m^{3*}s^{-1}$, and WS represents watershed area in km^2. Equations *3.6b* through *3.6f* represent recurrence intervals of 1.01-, 1.67-, 2-, 3.3-, and 10-years, respectively. Slopes were not statistically different ($p > 0.5$) among the regressions representing each of the five different recurrence intervals, so discharges for each recurrence interval were all expressed as a function of watershed area raised to the 0.90 power by methods outlined by Sokal and Rohlf (1995). Mean baseflow discharge calculated from stage measurements was also a function of watershed area raised to the 0.9 power ($\beta_{5,baseflow} = 0.06$, $R^2 = 0.88$, $p < 0.0001$).

Additional spatial data did not add explanatory power to the regional curve analyses. Area-weighted runoff coefficients explained approximately 68% of the variance observed in mean baseflow discharge at each of the 45 sites ($R^2 = 0.68$, $p < 0.0001$), but multiple regression of baseflow discharge vs. runoff coefficients and

Table 3.3. Results of stage-discharge regressions for 45 sites where grab samples were collected. In all but two cases (italics), results were statistically significant.

Creek	p	R^2	Watershed Area- km^2
Allison Creek	0.0001	0.983	15.4
Beasley Creek	0.005	0.887	15.9
Betty Creek	0.0117	0.829	43
Big Creek	0.0055	0.881	12.2
Blacks Creek	0.0001	0.976	11
Brush Creek at Lower Needmore Road	0.0025	0.806	19.4
Brush Creek Upstream of Highlands Road	0.0001	0.99	11.4
Buck Creek	0.0001	0.97	6.7
Burningtown Creek	0.0029	0.855	68.2
Cartoogechaye Creek at Recreation Park	0.0001	0.974	148.1
Caler Fork	0.0012	0.795	30.4
Cat Creek	0.0033	0.847	9.8
Cowee Creek	0.0005	0.843	66.2
Coweeta Creek at 441	0.0001	0.942	44.2
Crawford Branch	0.0126	0.673	5.7
Cullasaja River above Brush Creek	0.0001	0.977	76.1
Cullasaja River at River Rock Inn	0.0001	0.967	146.2
Cullasaja River at Wells Grove	0.0001	0.996	236.8
Cullasaja River between Reservoirs in Highlands	0.0043	0.895	16.6
Darnell Creek	0.0001	0.958	14.7
Dryman Fork	0.0002	0.918	31
Ellijay Creek	0.0002	0.978	52.2
Hickory Knoll Creek	0.0109	0.834	9.8
Iotla Creek	0.0001	0.931	26.1
Jones Creek	0.0006	0.879	18.1
Left Prong Burningtown Creek	0.0004	0.931	14.7
Little Tennessee River at 441 Bypass	0.0004	0.931	512.7
Little Tennessee River at Wolf Fork	0.0119	0.749	29.5
Mashburn	*0.948*	*0.001*	*3.8*
Matlock Creek	0.0003	0.866	13.0
Middle Creek	0.0021	0.926	31.9
Mill Creek, Tributary to Cartoogechaye	0.0008	0.914	10
Mill Creek Upstream of Mirror Lake	0.0009	0.907	3.9
Mud Creek	0.0017	0.881	17.8
N. Prong Ellijay Creek	*0.1403*	*0.739*	*17.5*
Poplar Cove Creek	0.019	0.784	10.3
Rabbit Creek	0.0001	0.93	23.6
Skeenah Creek	0.0012	0.943	17.4
Tellico Creek	0.0009	0.952	31.7
Tessentee Creek	0.0001	0.952	38.5
Turtle Pond Creek	0.0007	0.917	14.1
Walnut Creek	0.0018	0.974	16.9
Watauga Creek	0.0002	0.878	20.4
Wayah Creek	0.0008	0.866	36.5
Younce Creek	0.0001	0.974	6.7

watershed area was not significant. Nor were the following watershed descriptors significantly related to mean baseflow discharge when included as covariates of watershed area: road density, building density, % agricultural area, and % developed area. Runoff coefficients, land-cover and land-use data did not improve discharge predictions based on area alone, so regional curves were used to estimate discharge as a function of watershed area in the denominator of Equation 3.5.

Using SSURGO for K-factors, model estimates of annual sediment yield were statistically significant predictors of the annual baseflow sediment yield calculated from total suspended solids and discharge measurements (Figure 3.7, $R^2 = 0.69$, $p < 0.0001$). This analysis included only the 38 sites whose entire drainage basins were contained within Macon County because SSURGO data were not available outside its boundaries. Explanatory power was somewhat less, though still statistically signficant, when STATSGO data were used to represent soil *K*-factors ($R^2 = 0.54$, $p < 0.0001$); that analysis included all 45 sites where stage and discharge and total suspended solids had been measured on multiple dates in 1998. Additional land-cover information, namely building density, impervious cover, road density, and whole-watershed runoff coefficients, were not significant independent variables in a multiple regression model predicting baseflow sediment yield.

Sediment yield estimates were divided by the 1.67-year recurrence interval discharge from Equation 3.6c to estimate suspended sediment concentration. This estimate of suspended sediment concentration was not a statistically significant predictor of grab sample total suspended solids measurements (Figure 3.8).

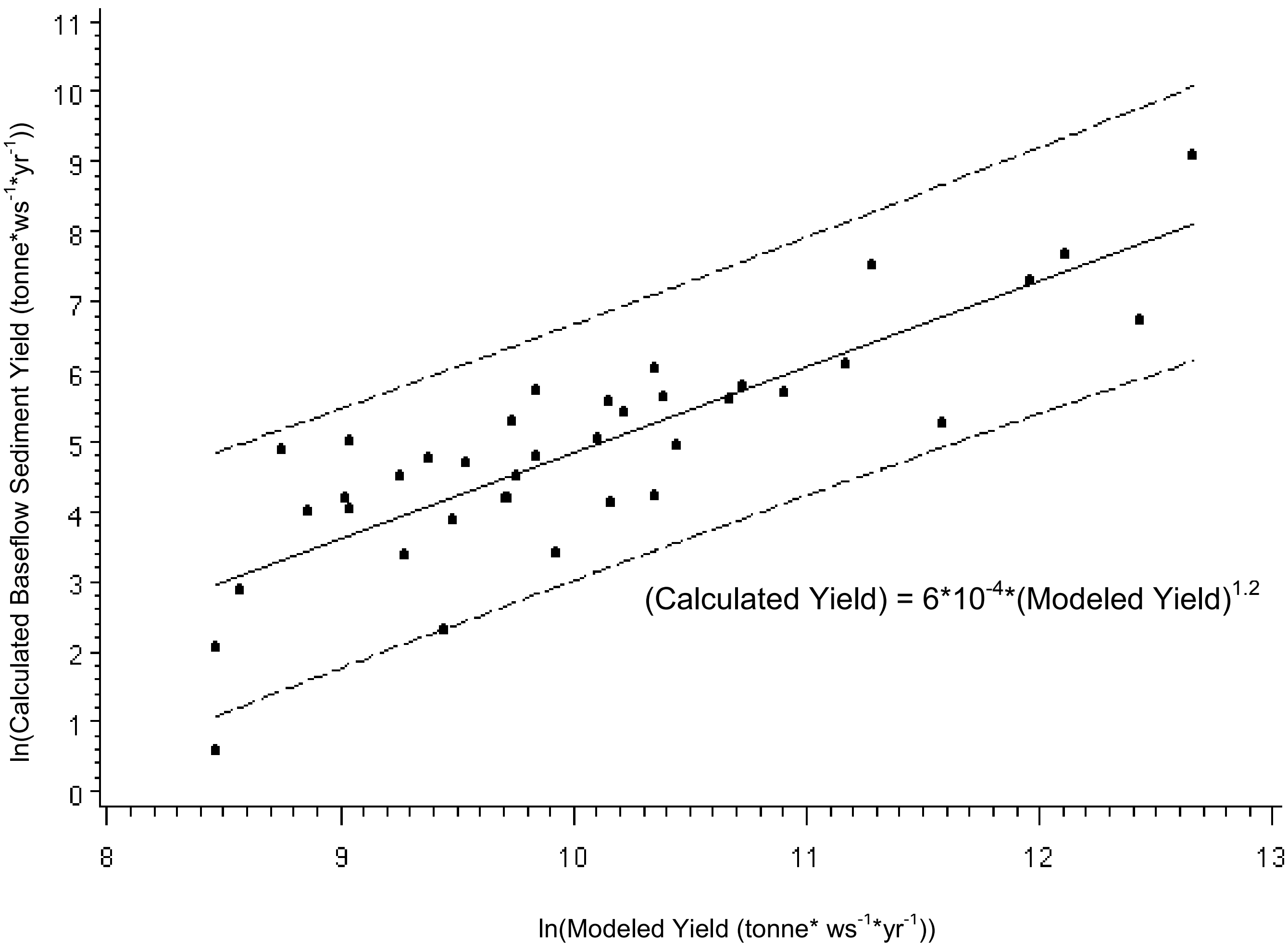

Figure 3.7. Modeled sediment yield based on SSURGO soil data were good predictors of calculated sediment yield. Regression estimates are shown with a sold line; 95% confidence intervals are shown with dotted lines.

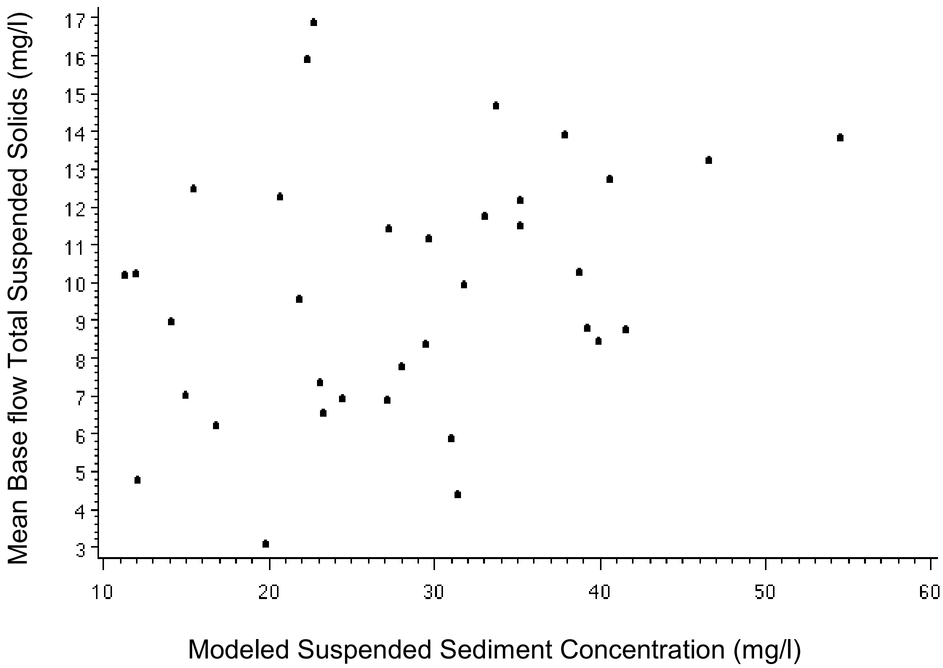

Figure 3.8. Modeled suspended sediment concentration was not a good predictor of observed baseflow sediment concentration.

Discussion

Previous research has examined the sensitivity of RUSLE to its input parameters (Renard and Ferreira, 1993; Ferreira et al., 1995; Yoder et al., 1998). Yoder et al. (1998) pointed out that RUSLE is most sensitive to the land-cover factor since that variable can assume values that span multiple orders of magnitude (see Table 3.1). While previous assessments of RUSLE's sensitivity are pertinent to sediment yield models that use it, those guidelines must be augmented with an understanding of the effects of modifications used to calculate sediment yield from soil-loss estimates.

Digital elevation model resolution appeared to be the most critical factor controlling the magnitude of model output. The aggregation of all data layers led to lower sediment yield predictions relative to predictions based on the highest resolution data available. However, aggregated soil K-factors and land-cover C-factor data produced larger estimates of sediment yield. By inference, the combined $L*S$ factor decreased substantially when pixel size was increased, and that change produced more dramatic changes in sediment yield estimates than were apparent when either the soil or land-cover data were degraded.

For all the comparisons, lower resolution input data introduced tremendous scatter in sediment yield estimates relative to reference model estimates. This loss of precision, graphically presented in Figures 3.4 and 3.5, serves as a caveat for those who use this type of simulation model to compare the possible effects of different land-use scenarios on potential sediment yield due to non-point source erosion. If land-cover data are derived from sources of information with differing resolutions, there will likely be a concomitant loss in thereplicability of model results.

Land-cover data are available at a number of resolutions, and the resolution of land-cover data alters sediment yield predictions. If cover factors are mapped using field surveys, aerial photography, or high resolution satellite imagery, lower estimates of sediment yield will be obtained in this study basin than with satellite imagery of moderate to low resolution, i.e. with pixel resolution equal to or greater than 30 m. Aggregation of land-cover factors from 30 m to 45 m produced approximately the same amount of uncertainty as aggregations up to 450 m. When isolated patches of forest, water bodies, or grass were aggregated with surrounding land uses whose *C* factors were larger (see Table 3.1), sediment yield estimates for watersheds were substantially altered.

The confusion within forested or pasture/grass categories described in Table 3.3 would not adversely affect either the RUSLE or runoff coefficient calculations since *C* factors (Table 3.1), and runoff coefficients (Appendix 3A) were very similar for the land-cover classes that were combined. Rather than continue to refine training sets for supervised classification, the image was deemed acceptable for the modeling exercises at hand, and the sediment yield model was run using the land-cover image described above.

The model's sensitivity to the resolution of raster data describing *K* factors underscores caveats issued by the US Department of Agriculture. Their documentation states that STATSGO data may not be appropriate for analyses for which site-specific information is required (USDA, 1991). In STATSGO, map units are aggregated, so soil K-values are not spatially explicit. In contrast, SSURGO data portray map unit composition based on field traverses and photo interpretation at scales from 1:12,000 to 1:63,600. Despite resolution limitations of STATSGO, it is often used in modeling exercises because the higher quality, SSURGO data are not commonly available in digital form. If STATSGO data must be used, sediment yield estimates will generally be greater than estimates made using the same methods and SSURGO data. Although there is a statistically significant (log-log) relationship between model outputs based on SSURGO and STATSGO, sediment yield estimates within any given sub-catchment may differ by an order of magnitude (Figure 3.6). Analysts must therefore be aware of the variability in the relationship between model output derived from these two sources of soil K-values. Despite the observed increase in unexplained variance when STATSGO soil data were used, model outputs that used those lower resolution soil data were significantly related to calculated annual baseflow sediment yield.

This and other sediment yield simulations based on USLE or RUSLE call for the deposition of detached soil when slope decreases during downslope transport (see Morgan and Nalepa, 1982; Banasik, 1986; Fernandes, 1994; McNulty and Sun, 1998). This study employed Moore and Wilson's observed relationship between maximum dimensionless transport capacity and the combined $L*S$ factor in RUSLE (1992). This theoretical maximum transport capacity function is an improvement over previously used methods that presume sediment deposition upon a change in slope, regardless of the mass of sediment annually delivered to a given pixel. Sediment delivery from a cell should be a function of local slope, surface roughness, soil erodibility, and the mass of sediment entering the cell, not a simple function of slope as has been used in the past.

On an annual basis, streams transport most sediment during storms (Pitlick and Van Steeter, 1998; Knighton, 1998), so suspended sediment concentration was estimated by annual upland sediment delivery divided by bankfull discharge, the channel forming flow (Knighton, 1998). However, baseflow total suspended solids concentrations could not be predicted with the modeled estimate of suspended sediment concentration using this methodology and the best available spatial data.

Suspended sediment concentration (mg * l^{-1}) is equivalent to sediment yield expressed on a per unit area basis (tonne * km^{-2} * yr^{-1}) according to the following argument that puts this study in context of other GIS models of erosion and sediment yield. Sediment yield is widely recognized as a function of watershed area and sediment delivery, so sediment delivery ratios are used to quantify the observation that larger watersheds tend to have larger floodplains and hence more sediment storage due to deposition (Roehl, 1962; Walling, 1983). The simplest sediment delivery ratios express the proportion of sediment delivered as an exponential function of watershed area. In the region of this study area, sediment delivery ratio exponents range from -0.15 to -0.20 (Roehl, 1962; Renfro, 1975; Walling, 1983; US EPA, 2000), so sediment delivery ratio (SDR) is expressed as follows:

$$SDR = \beta_6 * A^{-0.2} \qquad \textit{Equation 3.7}$$

where β_6 is a constant, and A is watershed area. Suspended sediment concentration (SSC) is expressed below:

$$SSC = \alpha \frac{SDR * Y}{Q} \qquad \textit{Equation 3.8}$$

Parameters in Equation 3.8 are sediment delivery ratio (SDR), sediment yield (Y, tonne * yr^{-1}), and discharge (Q, m^3 * s^{-1}), which is a function of catchment area as shown in equations 3.6a through 3.6f above, and α is a constant that incorporates the conversion of tonne * m^{-3} to mg * l^{-1} as well as the constant β_6. Therefore,

$$SSC = \delta \frac{S}{A^{1.1}} \approx \delta \frac{S}{A} \qquad \textit{Equation 3.9}$$

where δ is a constant. If the sites considered have watersheds whose areas span a narrow range over which $A^{1.1}$ is approximately linear, model estimates of *SSC* should be linearly related to sediment yield divided by watershed area. Equation 3.9 has the additional simplicity of expressing the sediment yield on a per area basis (tonne * km^{-2} * yr^{-1}) which is a formulation that can be easily applied to the regulation and management of land use. However, modeled suspended sediment concentration and mean total suspended solids concentrations observed at baseflow were not statistically related. Since most sediment moves through streams during storms (Pitlick and Van Steeter, 1998) and since stormflow sediment concentration is highly variable (Sutherland et al., in press), it is not surprising that it was not possible to predict total suspended solids concentrations. These analyses suggest that watershed sediment yield models that use RUSLE are better suited to predicting annual basin-wide sediment yield (tonne * yr^{-1}) than to predicting suspended sediment concentration.

Conclusion

Four main themes emerged from the sensitivity analyses of this sediment yield model. First, simultaneous coarsening of digital elevation, soil, and land-cover data layers led to lower predictions of sediment yield. Second, modeled sediment yields based on STATSGO were generally higher than yields that used SSURGO soil erodibility (K) factors. Third, land-cover data from imagery with pixels larger than 30 m x 30 m may result in higher sediment yields in forested, Blue Ridge Province watersheds. Fourth, all simulations produced unexplained scatter when comparing outputs to the reference model output. Therefore, model estimates of sediment yield are not comparable if different source data are used.

The sediment yield model provided statistically powerful estimates of annual baseflow sediment yield calculated from total suspended solids and discharge measured in the field. The same model, however, could not be used to predict total suspended solids concentrations. This lack of fit between modeled sediment concentration and observed total suspended solids concentrations was likely due to a number of factors, including the following: (a) the lack of precision in discharge estimates (see Equation 3.6); (b) the fact that the model provided annual sediment yield estimates rather than sediment yield from events or shorter time intervals; (c) the many other sources of sediment that a stream carries (Knighton, 1998); and (d) the spatial and temporal variability in total suspended solids concentrations in a stream.

The empirical soil-loss models, RUSLE and USLE, are relatively simple to implement within GIS software, and they provide objective means of comparing the potential impact of land-use management on stream sedimentation. However, management decisions should not be based solely on such modeling because of the many potential sources of error and variability identified here. Management decisions should be supported by local observations of stream channel conditions and processes in addition to watershed land use.

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Appendix

Appendix 3A. Runoff coefficients used to estimate relative contributions to excess runoff. Each coefficient is a unique combination of soil hydric class, local slope, and land cover (McCuen 1998).

Hydric Class	A	A	A	B	B	B	C	C	C	D	D	D
Slope Class	0-2%	2-6%	>6%	0-2%	2-6%	>6%	0-2%	2-6%	>6%	0-2%	2-6%	>6%
Mixed forest	0.05	0.08	0.11	0.08	0.11	0.14	0.1	0.13	0.16	0.12	0.16	0.2
Evergreen forest	0.05	0.08	0.11	0.08	0.11	0.14	0.1	0.13	0.16	0.12	0.16	0.2
Deciduous forest	0.05	0.08	0.11	0.08	0.11	0.14	0.1	0.13	0.16	0.12	0.16	0.2
Pasture and Grass	0.12	0.2	0.3	0.18	0.28	0.37	0.24	0.34	0.44	0.3	0.4	0.5
Cultivated	0.08	0.13	0.16	0.11	0.15	0.21	0.14	0.19	0.26	0.18	0.23	0.31
Impervious	0.85	0.86	0.87	0.85	0.86	0.87	0.85	0.86	0.87	0.85	0.86	0.87
Water	1	1	1	1	1	1	1	1	1	1	1	1

CHAPTER 4

INFLUENCE OF WATERSHED SETTING AND HILLSLOPE-DERIVED SEDIMENT ON FISHES IN THE UPPER LITTLE TENNESSEE RIVER, N.C.[1]

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[1]Gardiner, E. P., J. L. Meyer, and W.O. McLarney. To be submitted to Ecological Applications.

Abstract

This study examines land use and physiographic factors associated with fish assemblage shifts in a largely forested river basin in the southern Appalachian mountains. It explores the effects of hillslope-derived sediment on three groupings of fishes: (1) whole fish assemblages, (2) one widespread yet sediment-sensitive taxon, and (3) two groups of native fishes distinguished on the basis of their evolutionary history, distribution ranges, and preferences for distinct habitat types. A simulation model that implemented the Revised Universal Soil Loss Equation (RUSLE) within a geographic information system (GIS) was used to predict sediment loading rates in watersheds throughout the Upper Little Tennessee River basin. Model results were combined with physiographic descriptors of streams and watersheds to examine basin-wide shifts in assemblage structure associated with a range of natural and anthropogenic factors. Large and small watersheds greater and less than 39 km^2 in area had distinct faunal characteristics, and watershed area was identified as the principal correlate of this shift. Within small watersheds 23% of the variance in species abundances were encapsulated in an ordination using canonical correspondence analysis (CCA). Average hillslope and % forested area within the basin were the most important factors correlated with whole assemblage shifts among small watersheds. In large watersheds, for which CCA explained 52% of all fish occurrences, outlet elevation, basin relief ratio, and average stream slope in the stream network were correlated with large shifts in assemblage structure. Modeled sediment loading rate was not strongly correlated with any of the ordination axes in any of the analyses, but it was used to predict the relative abundance (proportion) of smoky sculpin (*Cottus bairdi* ssp) throughout the study area. Multiple

regression with modeled sediment yield rate (tonne km^{-2} yr^{-1}) and 1993 % forested area in the catchment explained 50% of the variation of sculpin among small watersheds (area < 39 km^2) and 72% of their variation among large watersheds (area > 39 km^2). Proportions of two groups of native taxa were also analyzed: (1) "highland endemics" with affinity for upland habitats and restricted distributions in the southern Appalachian highlands; and (2) "cosmopolitan species" that are widespread and can tolerate warmer temperatures, higher nutrient levels, and sandier substrate. In large watersheds, both groups' proportions could be explained using multiple regression with modeled sediment yield rate and % forested area as independent regressors. Sculpin comprised a large proportion of highland endemics, so explanatory power (R^2 = 0.74) was similar and somewhat redundant with the sculpin analysis; building density added some explanatory power to the analysis of cosmopolitan native taxa in large watersheds (R^2 = 0.60). Regression results suggested hillslope-derived sediment and forested area are important correlates of sculpin, highland endemic, and cosmopolitan proportions, but whole assemblages did not appear to vary with those factors. Whole assemblages, sensitive taxa, and groups of taxa provided complementary information about the effects of watershed land use on biotic assemblages.

KEYWORDS: Land use, watershed, geographic information systems (GIS), landscape, homogenization.

Introduction

The southeastern United States is one of the world's richest areas of endemism and diversity for temperate freshwater fishes (Burkhead and Jenkins 1991, Etnier and Starnes 1991), providing year-round or spawning habitats for over 660 species of fishes (Warren et al. 2000). That diversity is under considerable threat, however, with approximately 28% of those taxa of special concern, whether endangered, threatened, or vulnerable to extirpation (Warren et al. 2000). More than half of the diversity of freshwater fishes in the southeastern US occupy mountain streams of the southern Appalachians where darters (*Percidae*) and minnows (*Cyprinidae*) are concentrated and are especially threatened. Over 20% of taxa in these families are extinct, endangered, threatened, or vulnerable (Walsh et al. 1995).

Recent work has highlighted the susceptibility of native fishes of the southeastern Appalachian highlands to anthropogenic water quality and habitat modifications (Burkhead et al. 1997, Scott 2001, Scott and Helfman 2001). Humans' actions can directly and indirectly increase stream water temperature, elevate nutrient concentrations (Lowrance et al. 1984), or increase sediment loading to streams (Tebo 1955, 1957, Gurtz et al. 1980). Each of these has been shown to degrade aquatic habitats and/or biological integrity (Waters 1995, Hunsaker and Levine 1995). For example, excessive sediment reduces fish reproductive rates (Newcombe and MacDonald 1991, Newcombe and Jensen 1996, Montgomery et al. 1996), increases morbidity, and is associated with lower population sizes for a variety of taxa (Berkman and Rabeni 1987, Newcombe and Jensen 1996). Scott and Helfman (2001) have suggested that native fishes that evolved in mountain streams of the crystalline Blue Ridge Mountains are more vulnerable to habitat modifications than more widely distributed native taxa. The former group evolved in habitats with cool temperatures, low nutrient concentrations, bedrock-constrained stream bottoms, and coarse, clean substrate. The latter evolved to use larger river habitats typified by finer substrates and often higher temperatures and nutrient concentrations (Mayden 1987). Below, these two groups of taxa are termed "highland endemics" and "cosmopolitan", respectively. As cosmopolitan fishes exploit modified habitats, their ranges may expand, leading to replacement of highland endemics (Scott and Helfman 2001). This process is an example of faunal homogenization (Rahel 2000) by native taxa (Scott and Helfman 2001).

Important gaps in knowledge must be addressed to elucidate how human land use affects whole fish assemblages, highland endemics, cosmopolitans, and individual taxa of interest. One such gap is that land use factors are often intercorrelated, or redundant, with other factors examined (Angermeier and Winston 1998). Hence multiple regression models that use different sets of independent data may yield similar explanatory power (see Harding et al. 1998) without identifying a specific cause and effect relationship. In extreme cases, it may be difficult to establish a mechanism for the effect on fishes of an observed land use or physiographic pattern if two very different environmental indicators provide similar explanatory power. Simulation modeling can be used to address that gap in knowledge by integrating land use, terrain geomorphology, hydrology, or additional stream processes to a specific stressor of interest such as sediment. Model output, rather than land use metrics alone, can then be related to measures of biological integrity such as proportions of taxa, biological metrics, or whole assemblages.

Excessive sediment has consistently been among the most prevalent impairments to water bodies in the United States in recent decades (Judy et al. 1984, Waters 1995, US EPA 1997) and is a widely recognized yet complex problem for water resource managers. Although biological consequences are understood, they must be measured *in situ*. The same is true for the hydraulic and geomorphic factors that control the timing and quantity of sediment moving in stream channels which also must be measured in the field (Walling 1983, Doyle et al. 2000). Sources of sediment are difficult to trace. Geomorphologists and hydrologists have long recognized the monotonic relationship between hillslope erosion rates and sediment yield from watersheds (Roehl 1962, Renfro 1975, Walling 1983). A large proportion of a watershed's sediment budget is the movement of soil from upland areas into waterways and valley bottoms (Knighton 1998, Trimble 1999). Floodplain deposits can later be carried in suspension by streams and rivers through bank erosion, lateral scouring, or resuspension of formerly deposited sediments (Walling 1983, Meade et al. 1990, Gordon et al. 1992, Trimble 1997, Knighton 1998). Throughout the United States, upland erosion has been linked to excessive sediment delivery to streams and subsequent declines in stream habitat and water quality (Berkman and Rabeni 1987, Pimentel and Skidmore 1999).

The potential flux of sediment to water bodies can be examined through simulation modeling using geographic information systems (GIS, Burrough 1993). The Universal Soil Loss Equation (USLE, Wischmeier and Smith 1978) or the Revised USLE (RUSLE, Renard 1997) are often implemented within GIS software to provide estimates of sediment loading to water bodies (Fernandes 1994, McNulty et al. 1995, Pilotti and Bacchi 1997, De Roo 1998, Molnar and Julien 1998, Renschler et al. 1999, Yitayew et al.

1999, James and Hewitt III 1992). The chief advantage of USLE or RUSLE is their simplicity because there are relatively few parameters to fit, and thousands of plot years of data support their application (Risse et al. 1993, Nearing et al. 2000). Reasonable estimates of soil loss and sediment delivery can be obtained from USLE or RUSLE computer simulations when the purpose is to assess relative rates of soil loss and sediment export from watersheds (Nearing et al. 2000).

Faced with the rapid loss of native fish diversity due to chronic anthropogenic stresses, there is increasing need for methods to identify watersheds and streams degraded by anthropogenic land use change. The first objective of this study is to determine whether fish assemblages are distributed across continuous environmental gradients throughout the basin or, alternatively, if physiographic properties of watersheds are associated with distinct groupings of taxa. The second objective is to investigate whether hillslope-derived sediment has influenced fish assemblages in the Upper Little Tennessee River basin in the southern Blue Ridge physiographic province. This is done by examining whole assemblages, highland endemics, cosmopolitan fishes, and a subspecies of sculpin endemic to the study region.

Study Area and Methods

Study Area

The study area is a 1200 km^2 portion of the Upper Little Tennessee River (Figure 4.1), a north-flowing tributary of the highly diverse Tennessee River system (Etnier and Starnes 1991, Warren and Burr 1994, Warren et al. 2000). The majority of this area, nearly 1000 km^2, lies within Macon County, North Carolina. The northern, downstream section of the basin is in Swain County, N.C., and the southern, upstream

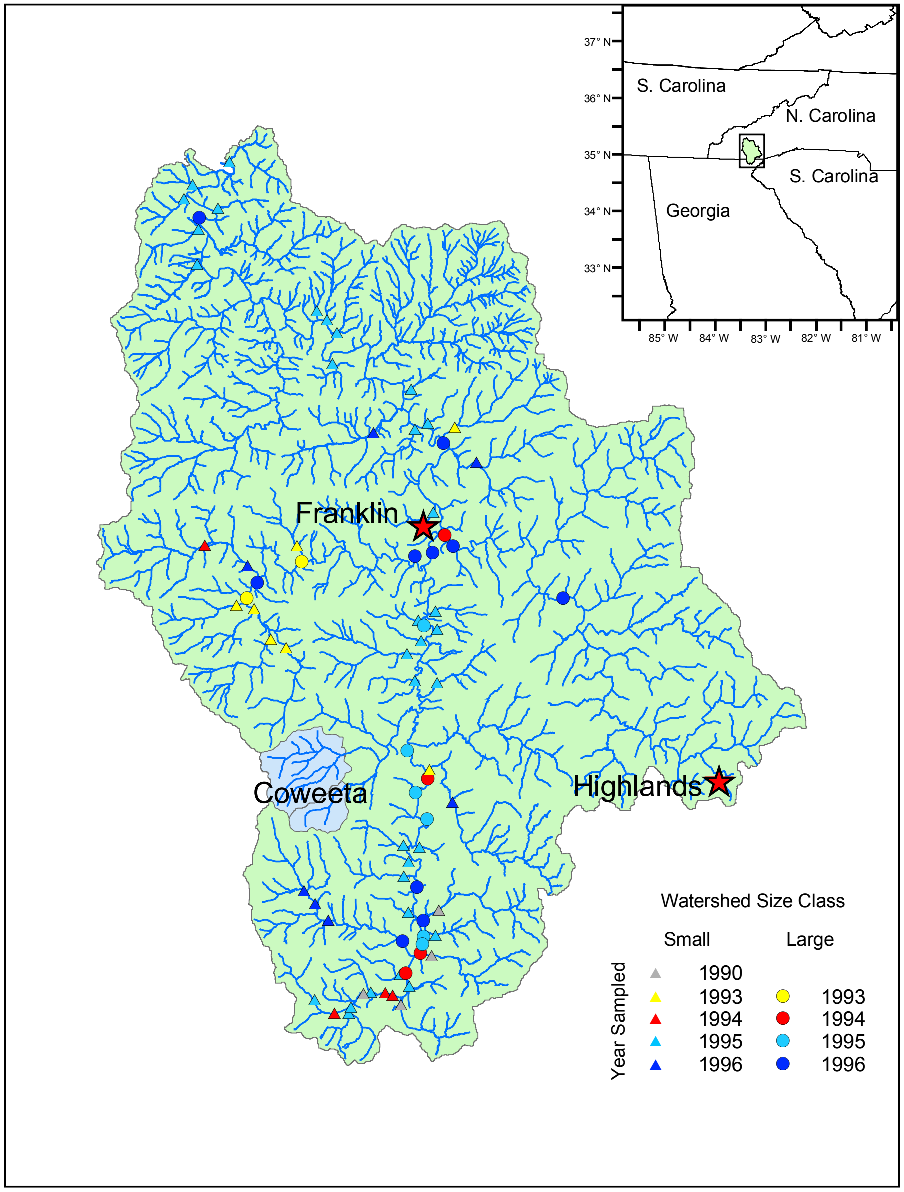

Figure 4.1. Fishes were collected at 76 sites throughout the study area between 1990 and 1996. Colors represent years in which data were collected: gray, 1990; yellow, 1993; red, 1994; light blue, 1995; dark blue, 1996. Symbols represent Watershed size classes: triangles, small watersheds (area < 39 km^2); circles, large watersheds (area > 39 km^2).

portion is in Rabun County, Georgia. Elevation varies between 1650 and 550 m and has a mean value of 880 m. Headwaters are mostly forested and have steep slopes shaped from complex folds, contacts, and thrusts with gneiss and schist bedrock of Cambrian and Precambrian age (Hatcher 1988). The main stem of the river meanders through a broad valley with unconsolidated quaternary colluvium and alluvium and where the predominant land use is agricultural land used for grazing and light crop production. In 1993, the basin was 89% forested, 9% agriculture, and 2% developed, according to a published land cover mosaic derived from Landsat-5 imagery (Hermann 1996). Human settlement is closely tied to topographic conditions throughout the basin (Wear and Bolstad 1998, Wear et al. 1998). Rural land uses with scattered homes at low density typify upper and side slopes while cleared valley land is used for light agricultural and commercial activities. The city of Franklin, N.C., the Macon County seat, is the largest population center in the basin. The next largest town is Highlands in the headwaters of the Cullasaja, a high gradient river in the southeastern portion of the study area.

Research in this study area has linked land use, sediment generation, aquatic habitat degradation and changes in aquatic biological organization. The focal site of much of that research has been the Coweta Hydrologic Laboratory, home of the Coweeta Long Term Ecological Research program (Swank and Crossley 1988), located in the southern portion of the study area (Figure 4.1). Almost 50 years ago, Tebo (1955, 1957) showed that sediment originating from logging roads can homogenize the substrate of high gradient eastern mountain streams thus negatively affecting macroinvertebrate habitats. Stream studies have been organized in conjunction with experimental forest harvesting to examine the timing and quantity of sediment, nutrient, and organic matter fluxes from watersheds following forest clearing (Webster et al. 1991). Nutrient concentrations increase dramatically following timber harvest in small watersheds but rapidly return to pre-disturbance levels. Streams require longer periods before sediment and coarse woody debris regimes return to pre-disturbance conditions.

Fish Collection

Fishes were collected at 76 sites throughout the basin in the spring or summer between 1990 and 1996 (McLarney 1993; McLarney 1994, McLarney 1995a; McLarney 1995b; McLarney 1996a; McLarney 1996b; McLarney 1997). Sites and years for fish collections are each noted in Figure 4.1. Backpack electric shockers were used to collect individuals greater than one year of age for all species present in distinct habitats at each site. Upon counting, individuals were revived and released. Species names and codes used in figures throughout this text are given in Table 4.1. Prior to sampling, land use conditions were surveyed for a minimum of 1 km upstream of each sampling site to ensure that no point sources of pollution or drastic changes in land use were evident within that larger span of river. Sampling effort was allocated to riffles, runs, pools, stream edge, and point bar habitats in proportion to the amount of each habitat present in the stream reach. Collection continued until no new species were discovered at a site and covered a minimum of 100 m.

Data describing natural and anthropogenic factors that were expected to vary with fish abundances and distributions were extracted from a GIS built for analysis of streams in the study region (see Chapter 2). All catchments contributing to each of the 76 sites were delineated. Watershed area (km^2) and outlet elevation (m) were recorded because they were each expected to influence both fish assemblage structure and modeled

Table 4.1. Species list, distribution class (Scott and Helfman 2001), number of sites out of 76 total where the species was found, total numbers of individuals collected, and species codes used in plotting ordinations.

Linnaean Name	Common Name	Code	Distribution Class*	Number of Sites Where Found	Total # Indivs. Collected
Ambloplites rupestris	Rock Bass	amru	C	43	201
Ameiurus nebulosus	Brown Bullhead	amne		4	4
Ameiurus platycephalus	Flat Bullhead	ampl		3	3
Campostoma anomalum	Central Stoneroller	caan		59	1471
Catostomus commersoni	White Sucker	caco	C	25	253
Clinostomus funduloides	Rosyside Dace	clfu	H	47	1390
Cottus bairdi	Smoky Sculpin	coba	H	71	11497
Cyprinella galactura	Whitetail Shiner	cyga		30	364
Cyprinus carpio	Common Carp	cyca		3	100
Dorosoma cepadianum	Gizzard Shad	doce		1	1
Erimonax monachus	Spotfin Chub	ermo		1	12
Etheostoma blennioides gutselli	Tuckasegee Darter	etblgu	H	17	87
Etheostoma chlorobranchium	Greenfin Darter	etch	H	12	134
Etheostoma vulneratum	Wounded Darter	etvu	H	2	22
Etheostoma zonale	Banded Darter	etzo	C	3	5
Hypentelium nigricans	Northern Hogsucker	hyni	C	57	609
Ichthyomyzon greeleyi	Mountain Brook Lamprey	icgr		46	579
Ictalurus punctatus	Channel Catfish	icpu		1	1
Lepomis auritus	Redbreast Sunfish	leau	C	45	362
Lepomis cyanellus	Green Sunfish	lecy	C	37	139
Lepomis gulosus	Warmouth	legu		6	9
Lepomis macrochirus	Bluegill	lema	C	21	160
Luxilus chrysocephalus	Striped Shiner	luch		1	1
Luxilus coccogenis	Warpaint Shiner	luco	H	58	1223
Micropterus dolomieu	Smallmouth Bass	mido		5	9
Micropterus salmoides	Largemouth Bass	misa	C	16	31
Moxostoma anisurum	Silver Redhorse	moan		1	1
Moxostoma carinatum	River Redhorse	moca		1	9
Moxostoma duquesnei	Black Redhorse	modu	C	14	50
Moxostoma erythrurum	Golden Redhorse	moer	C	29	147
Moxostoma macrolepidotum	Shorthead Redhorse	moma		1	1
Nocomis micropogon	River Chub	nomi	C	60	1615
Notemigonus crysoleucas	Golden Shiner	nocr	C	10	49
Notropis leuciodus	Tennessee Shiner	nole	H	48	1294
Notropis lutipinnis	Yellowfin Shiner	nolu	C	39	1239
Notropis photogenis	Silver Shiner	noph	C	2	10
Notropis rubellus	Rosyface Shiner	noru		1	13
Notropis spectrunculus	Mirror Shiner	nosp	H	26	274
Notropis telescopus	Telescope Shiner	note		4	27
Oncorhynchus mykiss	Rainbow Trout	onmy		29	183
Perca flavescens	Yellow Perch	pefl	C	3	4
Percina aurantiaca	Tangerine Darter	peau		1	4
Percina evides	Gilt Darter	peev	H	31	363
Percina squamata	Olive Darter	pesq	H	1	1
Phenacobius crassilabrum	Fatlips Minnow	phcr	H	18	57
Pylodictis olivaris	Flathead Catfish	pyol		1	1
Rhinichthys atratulus	Blacknose Dace	rhat	C	32	759
Rhinichthys cataractae	Longnose Dace	rhca		30	341
Salmo trutta	Brown Trout	satr		14	57
Salvelinus fontinalis	Brook Trout	safo	H	4	12
Semotilus atromaculatus	Creek Chub	seat	C	59	793
Stizostedion vitreum	Walleye	stvi		1	2

* "H" refers to highland endemics; "C" refers to cosmopolitan taxa identified by Scott and Helfman (2001).

annual whole-watershed sediment yields. Watershed and stream geomorphic characteristics were extracted using catchment boundaries, elevation data, and vectors recorded from 1:24,000 scale quadrangle maps for the study area. Sinuosity (unitless), maximum flow length for overland flow (km), basin major axis length (km), relief ratio (unitless), average hillslope (unitless), and average stream gradient within each basin (unitless) were all extracted. These factors have long been associated with physical processes in streams (Knighton 1998). For example, high gradient streams might be expected in catchments with high average hillslopes and relief ratios. Such streams would be expected to have higher sediment transport capacities than those with lower gradients.

The timing and quantity of suspended and saltating sediment in a stream channel are primary determinants of its geomorphic structure (Leopold 1994) and of the biotic community that resides there (Waters 1995). Annual whole-basin sediment yield was simulated for all streams in the study area by accumulating RUSLE soil loss estimates down slope to stream outlets. The details of this model are described in Chapter 3, in which model estimates of annual sediment yield were shown statistically to be related to sediment yields calculated from grab sample sediment concentrations and river discharge at baseflow ($R^2 = 0.54$, $p < 0.0001$, see Chapter 3). Model estimates of sediment loading increased with watershed area, so modeled annual loading was divided by watershed area to provide modeled sediment loading rates (tonne km^{-2} yr^{-1}). This normalizes each prediction, allowing comparison of relative rates of hillslope-derived sediment yield among watersheds with different areas.

Land use metrics were extracted from catchment boundaries (Chapter 2), land cover information, and digitized buildings and roads from 1:24,000 quadrangle maps. Percent forest in 1993 was calculated by overlaying catchment boundaries on land cover from the Southern Appalachian Assessment (Hermann 1996). Forest cover in 1970 was assessed using data developed by Wear and Bolstad (1998). Density of buildings (# km^{-2}) was calculated using digitized buildings and catchment boundaries. Densities of unimproved and improved roads (km km^{-2}) were calculated for each watershed upstream from each fish sampling site. Regression was used to examine which land cover variables explained modeled sediment loading estimates most effectively. Correlation analysis was used to select watershed metrics that most effectively captured important land use characteristics while minimizing redundancy. Variables, their descriptions, and transformations used to approximate normality for each are provided in Table 4.2.

Canonical correspondence analysis (CCA) was used in the software package PC-ORD (McCune and Mefford 1999) to explore multivariate relationships among watershed metrics and fish collection data. This is a method of direct gradient analysis analagous to multiple regression in which species data are expressed as a function of multiple environmental descriptors (Jongman et al. 1995). Rather than examining a single species, CCA expresses the variation within an entire matrix of species abundances as a function of multiple environmental parameters. Abundance data were raised to the 0.25 power prior to CCA, and environmental parameters were transformed as described in Table 4.2 to better approximate normal distributions. Weighted averages of species scores for each site and of site scores for each species are computed iteratively until values change little in subsequent calculations (Jongman et al. 1995). Site scores are the weighted averages

Table 4.2. Variables extracted from the GIS database and used in analyses. The top portion of the table is a correlation matrix. Excessive correlations are in bold. The bottom row provides variable descriptions and transformations used to approximate normal distributions. Correlations exceeding 0.7 are in **bold face**; see text for details regarding high correlations among variables.

	sqkm	MAJ-AXIS	RH	SW	avgrad	sinuosity	slope	elev	pvden	grden	loadarea	tf70	tf93	tblds
sqkm	1.000													
majaxis	**0.920**	1.000												
RH	-0.521	**-0.702**	1.000											
SW	0.000	0.031	0.358	1.000										
avgrad	0.028	-0.017	0.030	-0.129	1.000									
sinuosity	0.328	0.322	-0.241	-0.012	-0.025	1.000								
slope	-0.073	-0.084	0.041	-0.085	**0.806**	-0.093	1.000							
elev	-0.263	-0.148	0.035	0.137	-0.045	-0.118	-0.111	1.000						
pvden	0.028	0.005	-0.195	-0.640	0.109	0.051	0.143	-0.165	1.000					
grden	-0.042	-0.110	0.145	-0.507	0.200	-0.033	0.191	-0.160	**0.716**	1.000				
loadarea	-0.035	-0.090	0.188	-0.335	0.158	-0.031	0.168	-0.154	0.665	**0.939**	1.000			
tf70	0.020	0.063	0.308	**0.875**	-0.196	-0.030	-0.246	0.153	-0.603	-0.407	-0.265	1.000		
tf93	0.010	0.040	0.334	**0.855**	-0.303	-0.012	-0.323	0.161	-0.579	-0.382	-0.251	**0.952**	1.000	
tblds	0.109	0.124	-0.406	**-0.775**	0.091	0.082	0.181	-0.032	**0.791**	0.565	0.434	**-0.747**	**-0.712**	1.000
Descriptions	Watershed Area (km^2)	Length of Major Axis (m)	Relief Ratio (unitless)	Average Hillslope (unitless)	Average Gradient in Stream Network (unitless)	Sinuosity of Stream Reach (unitless)	Gradient of Stream Reach (unitless)	Elevation (m)	Improved Road Density ($km\ km^{-2}$)	Unimproved Road Density ($km\ km^{-2}$)	Modeled Sediment Loading Rate ($tonne\ km^{-2}\ yr^{-1}$)	Catchment Percent Forest in 1970	Catchment Percent Forest in 1993	Building density ($\#\ km^{-2}$)
Trans-formation Used	ln	ln	none	none	none	none	none	none	ln	ln	ln	Arcsin (sqrt (%F70/100))	Arcsin (sqrt (%F93/100))	ln

of species numbers multiplied by each species' score, and species scores are the weighted average of site scores. This process is iterated with updated species and site scores until neither set of scores changes in subsequent calculations. The method is robust to initial site or species scores. Final scores maximize the dispersion among sites based on the abundances and species of fishes present at each site. Ordination plots portray similarity and dissimilarity among site scores, species scores, or both. Site scores were scaled using Hill's method (Jongman et al. 1995) and normalized to standard deviation units so that sites that have an absolute difference of 4 s.d. units have essentially no taxa in common. In CCA, the weighted averaging is restricted to account for dispersion of sites due to environmental descriptors. The amount of variation in species data explained by each orthogonal axis is reported along with correlations of environmental data with each axis. These methods are described in detail by Jongman et al. (1995).

Three CCA were performed. The first showed the relationship between fish abundances at all sites and watershed elevation and area. The second and third were performed on data stratified by watershed size since the first CCA suggested that streams with catchments larger and smaller than 39 km^2 had biologically distinct assemblages. For each of these analyses, watershed descriptors were first subjected to correlation analysis in order to remove redundant information from the matrices of explanatory data. This step also facilitated interpretation of each CCA. Multiple regression was used in each watershed size class to relate proportions of individual taxa, highland endemics, and cosmopolitan taxa to modeled sediment loading rate and other land use variables.

Results

Some general observations about the correlations among independent variables (Table 4.2) are necessary before presenting results from the ordinations. Many of the independent data were normalized by watershed area so that comparisons could be made between watersheds irrespective of size. Two exceptions in Table 4.2 are watershed area (sqkm) and major axis length (majaxis) which were tightly correlated. Watershed area was used in the first ordination; major axis, a description of watershed size, was dropped from all analyses since it was highly correlated with watershed area and relief ratio. The latter metric is an indicator of overall change in elevation, so that metric was used in ordinations 2 and 3. Reach slope and average stream gradient in the basin were highly correlated, so it was dropped from further analyses. Local slope can be a very important determinant of habitat quality in a stream reach (Walters et al. 2001). However, experience had suggested that the measurement of slope from 30 m digital elevation data and stream segment data from topographic maps does not agree with *in situ* measurements (see Chapter 2). As measured here, gradient is an indicator of general changes in relief. Average stream gradient, also extracted from digital elevation models, is an indicator of overall basin morphometry and was retained for the ordinations. Average hillslope was an efficient indicator of development intensity: percent forest in 1970 and in 1993 and building density were all very highly correlated with average basin hillslope. Steep slopes are difficult places to build homes, so buildings tend to be in flatter areas. Average hillslope (SW) conveys geomorphic and hydrologic information and is correlated with land use. Building density was highly correlated with improved road density (pvden). Linear regression suggested that unimproved road density (km km^{-2}) is an adequate surrogate for model estimates of sediment loading rate (tonne km^{-2} yr^{-1}; Figure 4.2; $R^2 = 0.88$, $p < 0.0001$). Alternatively, the total sediment yield (tonne yr^{-1}) from a given watershed is a function of the length of unimproved roads in that watershed. Past research has shown the link between logging roads and sediment load in forested watersheds (Tebo 1955, 1957, Gurtz et al. 1980, Swift 1984, 1988). Sediment loading rate (loadarea) and average basin hillslope (SW) were used in ordinations 2 and 3 as indicators of sediment contribution to streams and human settlement intensity.

Ordination of All Sites

Elevation and watershed area explained 22% of the total variance in fish assemblage structure when all 54 species at all 76 sites were analyzed using CCA (Table 4.3, Figure 4.3). Elevation and area were not substantially correlated with one another ($r = -0.24$). Two distinct groups, separated by watershed size, are evident in the graph showing the two CCA ordination axes. Saylor and Ahlstedt (1990) and McLarney (1995) have noted that distinct biological communities exist above and below a watershed size of 39 km^2. That threshold corresponded well with results from the ordination of all sites. The first group of sites comprises watersheds greater than 39 km^2 in area and is arrayed along axis 1 of the CCA, which was highly correlated with watershed area and explained 18% of the variance in the matrix of all fishes collected. The second group of sites had catchments smaller than 39 km^2 and were arrayed along axis 2. This axis was most highly correlated with outlet elevation ($r = 0.66$). However, axis 2 only explained 4% of species observations and was thus deemed unimportant. After examining this ordination, small and large watersheds with areas less and greater than 39 km^2 were analyzed independently since they were characterized by distinctive fish assemblages. Given the

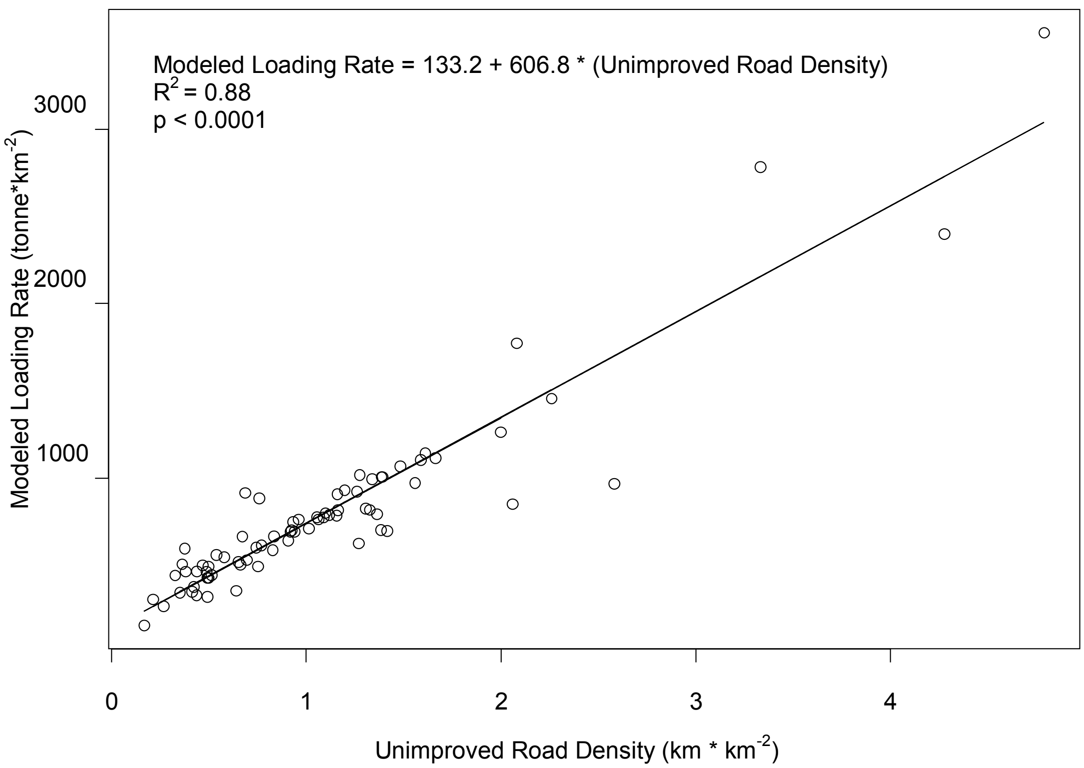

Figure 4.2. The per area rate of sediment yield from watersheds (tonne km^{-2} yr^{-1}) was linearly related to unimproved road density (km km^{-2}).

Table 4.3. Results of CCA when all 76 sites were analyzed and compared to watershed elevation and area.

	Axis 1	Axis 2
Variance in Species Data		
% Variance Explained	17.9	4.2
Cumulative % Explained	17.9	22.1
Correlations		
Elevation	0.52	0.66
Basin Area	-0.95	0.10

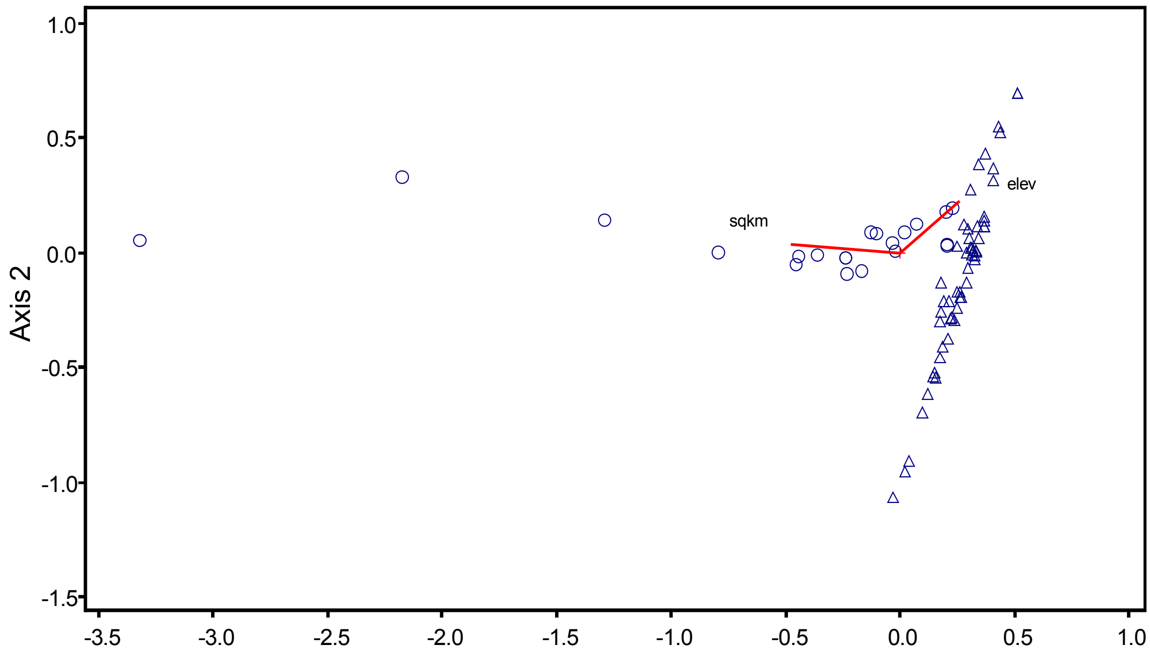

Figure 4.3. Canonical correspondence analysis ordination of all fishes collected at all 77 sites. Site scores were arranged to maximize variation explained by watershed area and outlet elevation. Watershed area was correlated with axis 1 while elevation was correlated with axis 2, pointing to the biological distinctiveness of assemblages found in streams above (circles) and below (triangles) 39 km^2 in area.

low explanatory power of axis 2, elevation was not used to stratify sites. However, elevation was retained as a covariate with the other watershed metrics and measures of anthropogenic impacts.

Small Watersheds

Among the small watersheds, there were very high intercorrelations among many of the watershed and site descriptors extracted from the GIS, so several variables were dropped from further analyses. As noted above, modeled sediment delivery rate (tonne km^{-2}) was explained by the density of unimproved roads in watersheds (km km^{-2}; r = 0.94). The percentage of each watershed that was forested in 1993 was nearly perfectly correlated with % forested area in 1970 (r = 0.95). Building density (# km^{-2}) and improved road density (km km^{-2}) were highly correlated (r = 0.80). However, building density was not correlated with relief ratio, as it was when all sites were combined (Table 4.2). Average stream gradient and stream reach gradient were essentially redundant (r = 0.88). Elevation was not correlated with any of these factors. For ease of interpretation, data volume was reduced by retaining only the following variables for CCA: modeled sediment delivery rate (tonne km^{-2} yr^{-1}), relief ratio, average hillslope in the basin, average stream gradient in the basin, outlet elevation, stream reach sinuosity, watershed area, percentage of watershed area forested in 1993, and building density. Data were transformed using log- and arcsine-transformations to approximate normal distributions (see Table 4.2).

These variables explained 23% of the variance structure in the fish data for small catchments (Table 4.4, Figure 4.4). Three multivariate axes explained 11%, 8%, and 4% of the variation in sites scores computed from fish assemblage data. Average catchment

Table 4.4. Summary results of CCA for 37 species and 55 sites with small watersheds (Area < 39 km^2). Explained variance in species observations is followed by inter-set correlations among explanatory variables and Axes 1 through 3.

	Axis 1	**Axis 2**	**Axis 3**
Variance in Species Data			
% Variance Explained	11.1	7.5	4.5
Cumulative % Explained	11.1	18.6	23.2
Correlations			
Modeled Loading Rate	-0.33	-0.13	-0.05
Basin 1993 % Forest	0.63	-0.00	0.44
Basin Building Density	-0.20	0.18	-0.32
Basin Area	0.41	0.13	-0.14
Basin Relief Ratio	0.17	-0.05	0.15
Basin Average Hillslope	0.75	-0.13	0.21
Basin Average Gradient	0.09	-0.36	-0.54
Outlet Elevation	0.41	0.72	-0.12
Reach Sinuosity	0.13	0.14	-0.12

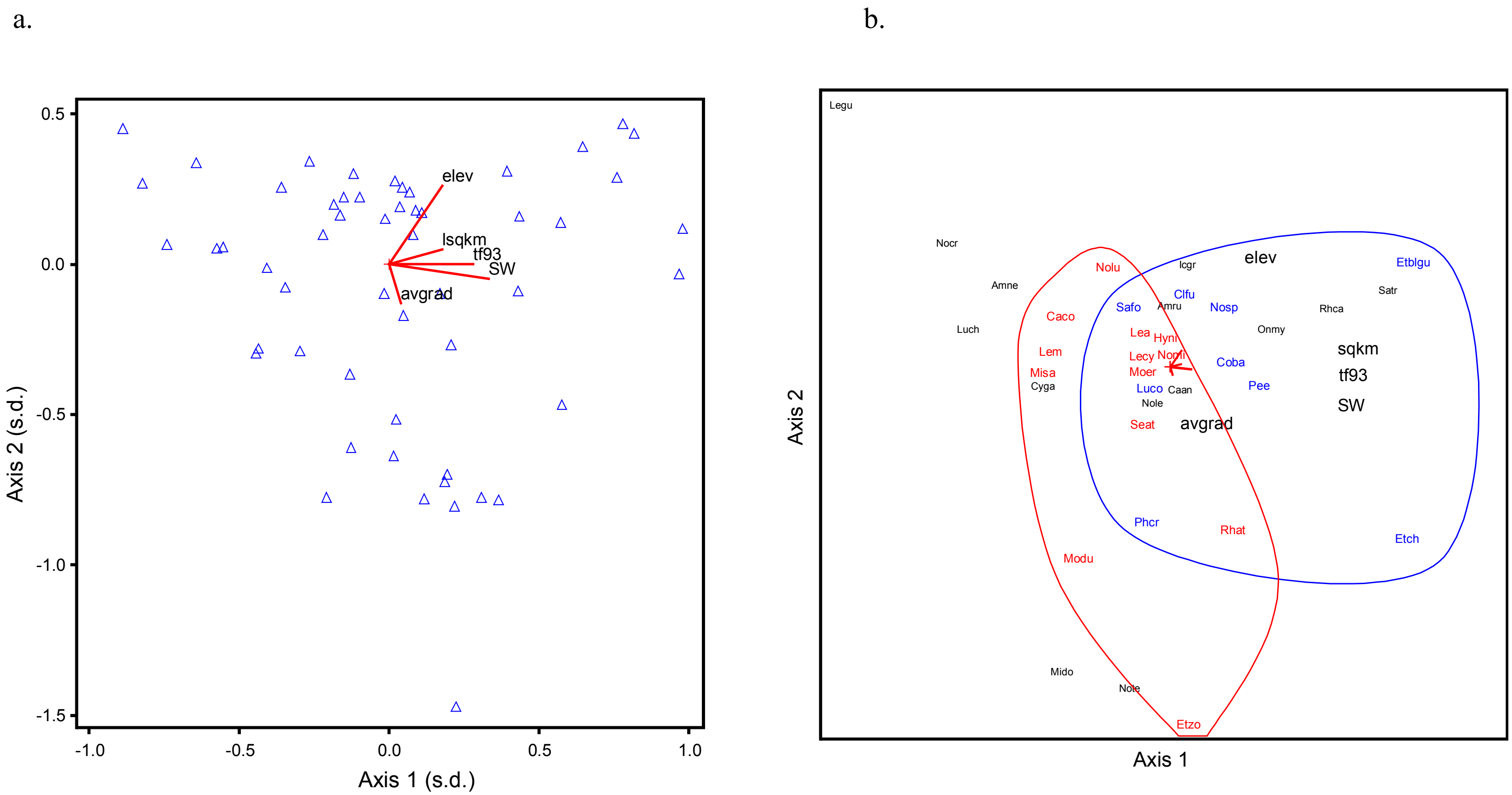

Figure 4.4. Canonical correspondence analysis results for small watersheds (area < 39 km^2). (a) The first two canonical axes (Hill's scaling) explained 19% of variations in species abundance and distribution. Abbreviations for environmental correlates are given in Table 4.2. (b) Relative values of species scores demonstrate three regions: cosmopolitan (red) taxa are concentrated in the lower left, highland endemics (blue) in the upper right, and a central area with overlapping species scores. Species codes are included in Table 4.1. Highland endemic taxa are shifted toward higher elevations and greater forested area. Red vectors in (b) are the same as in (a).

hillslope had the highest correlation with axis 1, followed by percent forest in 1993, elevation, and watershed area (Table 4.4). Modeled sediment loading rate had a small amount of correlation with axis 1. Elevation was strongly correlated with axis 2. None of the other descriptors had correlations above 0.4 on axis 2. Axis 3 was deemed unimportant because it explained only 4% of the total variation in species data.

Small catchments were characterized by less biological variability than the large sites. Sites with small catchments spanned a range of about 2 s.d. units along Axes 1 and 2 in the first ordination, suggesting substantial biological similarity, even among sites with very different elevations. The most distinct sites on axis 1 were Jerry Branch and Wayah Creek. Wayah Creek had 11 species while Jerry Branch had 24, and they shared 7 taxa. Sites at either end of axis 2 were Betty Creek at Messer Creek Road (axis 2 score 0.7) and Sawmill Creek (axis 2 score -1.1). Betty Creek is a high elevation site (elevation 716 m) in the headwaters of the Little Tennessee whereas Sawmill Creek is at the downstream terminus of the study area and drains directly into the mainstem of the river (elevation 524 m). This substantial difference in outlet elevation was accompanied by approximately 50% species turnover. Betty Creek at Messer Creek Road supported 16 taxa, Sawmill Creek supported 17, and there were 8 species common to each.

Cosmopolitan and highland endemic taxa are enclosed by red and blue polygons, respectively, in the plot of species scores (see Figure 4.4b). Three regions are notable: a region with cosmopolitan taxa dominating, a region of overlap between cosmopolitan and highland endemic taxa, and a region where scores of highland endemic taxa are concentrated. The observed separation supports the contention that the two groups of taxa have differing habitat preferences and requirements (Mayden 1987, Scott and Helfman 2001) and illustrates the availability of habitat suitable for highland endemics.

Elevation and percent forest explained 49% of the variance in smoky sculpin (*Cottus bairdi*) proportions among sites with small catchments ($p < 0.0001$ after Bonferroni-correction). Both factors had positive coefficients, indicating a preference by sculpin for forested, high elevation sites. Proportions of no other common taxa among small streams were statistically related to the watershed and stream descriptors analyzed.

Large Watersheds

The correlation structure of independent variables describing characteristics of larger watersheds (area > 39 km^2) was somewhat different than that for small watersheds. Improved road density, unimproved road density, building density, and modeled sediment delivery rate were all correlated with Pearson's r values of 0.8 or more. Watershed area was highly correlated with both relief ratio and outlet elevation. As noted for small watersheds, 1970 and 1993 % forest were highly correlated. Variables used in the third CCA are listed in Table 4.5.

Approximately 52% of the variation in site scores based on species abundances were explained with the first three CCA axes (Figure 4.5). Axes 1, 2, and 3 explained 30%, 13%, and 8% of the total variation in site scores, respectively. Elevation, average basin gradient, and relief ratio were most correlated with axis 1 (Table 4.5). Basin % forest in 1993, relief ratio, and average basin hillslope all had correlations greater than 0.6 with axis 2. Modeled sediment loading rate was negatively correlated with axis 1 but positively correlated with axis 2; however, it was not associated with a large proportion

Table 4.5. Summary results of CCA for 50 species and 21 sites with large watersheds (Area > 39 km^2). Explained variance in species observations is followed by inter-set correlations among explanatory variables and Axes 1 through 3.

	Axis 1	**Axis 2**	**Axis 3**
Variance in Species Data			
% Variance Explained	30.5	13.4	7.7
Cumulative % Explained	30.5	43.9	51.6
Correlations			
Modeled Loading Rate	-0.28	0.35	-0.15
Basin 1993 % Forest	0.16	-0.63	-0.50
Basin Relief Ratio	0.62	-0.60	0.13
Average Basin Hillslope	0.05	-0.65	-0.43
Average Gradient in Basin	-0.64	-0.35	-0.49
Outlet Elevation	0.92	-0.23	0.11
Reach Sinuosity	-0.30	-0.15	-0.23

a. b.

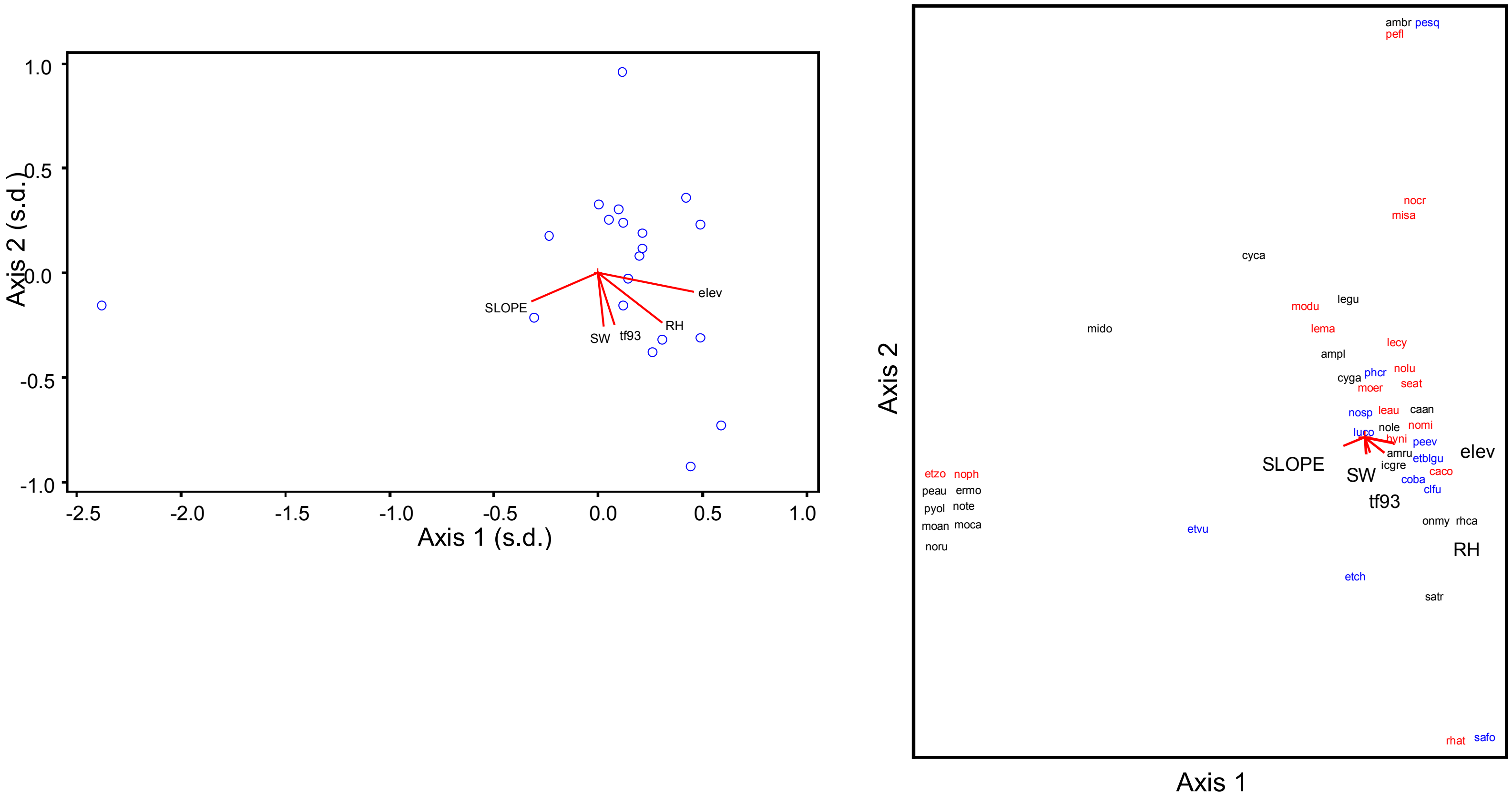

Figure 4.5. Ordination of fish assemblages and environmental descriptors found in streams with watersheds greater than 39 km^2. (a) Site scores (Hill's scaling) encapsulated 30% and 13% of variance in species data on Axes 1 and 2, respectively. The first axis spans about 3 s.d. units, representing substantial species turnover. See Table 4.2 for environmental descriptor definitions. (b) Species scores showed that cosmopolitan (red) and highland endemic taxa (blue) had broadly overlapping environmental tolerances. See Table 4.1 for species abbreviations. The Needmore (middle left) site was quite distinct from other large elevation sites.

of the overall variation in fish abundances. The third axis is not interpreted here due to its marginal explanatory significance.

Although it was not strongly correlated with the canonical axes, modeled sediment loading rate was a statistically powerful independent variable in several regression analyses. Modeled loading rate and 1993 percent forested area accounted for 72% ($p < 0.01$ after Bonferroni-correction) of the variation in smoky sculpin (*Cottus bairdi* ssp.) proportions in large watersheds (Figure 4.6). As indicated in the equation in Figure 4.6, sediment loading rate was negatively correlated and % forested area was positively correlated with sculpin proportions. Similarly, 74% of the variation in highland endemic proportions were explained through regression against modeled sediment loading rate, transformed 1993 % forest, and transformed building density. Those variables explained 60% of the variation in proportions of cosmopolitan taxa. As expected, coefficients were negative for building density and loading rate but positive for % forested area. Modeled sediment loading rate was not a useful predictor for other taxa found in large watersheds.

Discussion

This study revealed three distinctions among small and large watersheds in the Upper Little Tennessee River basin. (1) They support distinct fish assemblages with differing degrees of homogenization. (2) Variations in taxa abundances and distributions were correlated with different factors in each watershed size class. (3) Regression analyses had different explanatory power within each set of watersheds; proportions of highland endemics, cosmopolitans, and sculpin were more accurately predicted among large watersheds than among small ones. To some degree, these results confirm what is

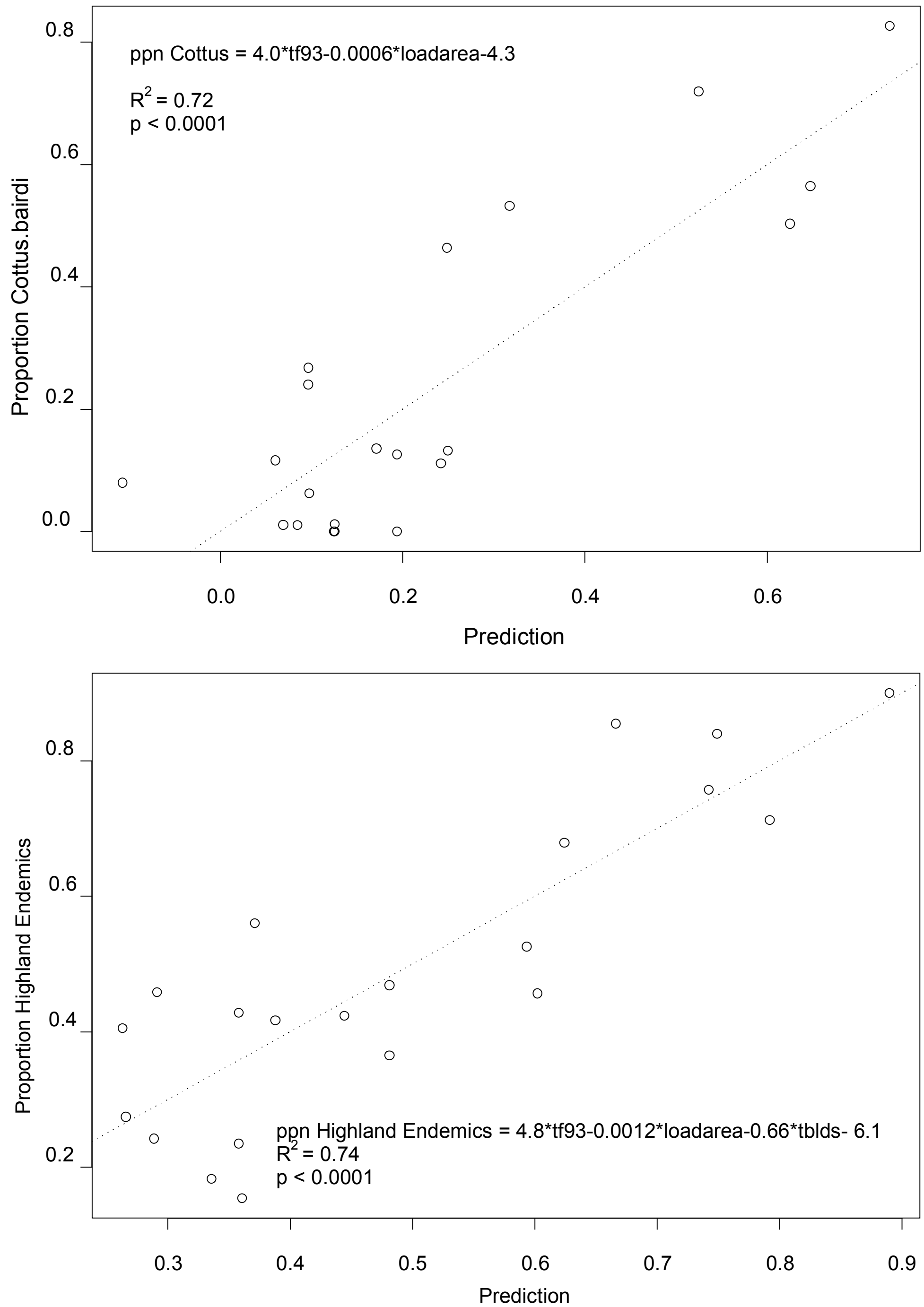

Figure 4.6. Regressions from sites with large watersheds (area > 39 km^2). (a) Proportions of smoky sculpin (*Cottus bairdi* ssp.) were tightly related to modeled sediment loading rate (loadarea, tonne km^{-2} yr^{-1}) and 1993 %forested area (arcsine-transformed) in the catchment (tf93). (b) Building density (log-transformed) added explanatory power in predicting proportions of highland endemic taxa.

known about the natural history of fishes native to the region, but they also point to important distinctions between large and small catchments in their landscape setting and assemblage structure.

Whole Assemblage Analysis with CCA

The first ordination (see Table 4.3, Figure 4.3) showed that sites with catchments smaller than 39 km^2 had almost no biological similarity to the sites with the largest catchments. Their separation along axis 1 by almost 4 s.d. units indicated almost complete lack of similarity between sites on opposite ends (Jongman et al. 1995).

In the second ordination (see Figure 4.4, Table 4.4), the smaller catchments were analyzed alone, and average hillslope and 1993 % forested area were the most significant correlates with axis 1. Brown trout (*Salmo trutta*), Tuckaseegee darter (*Etheostoma blennioides gutselli*), greenfin darter (*E. chlorobranchium*) and longnose dace (*Rhinichthys cataractae*) were all weighted positively on this axis. These taxa prefer cool water temperatures and require clean, coarse substrate for spawning and for feeding on macroinvertebrates. Taxa with negative loadings on axis 1 included warmouth (*Lepomis gulosus*), brown bullhead (*Ameiurus nebulosus*), and golden shiner (*Notemigonus crysoleucas*). Negative values on axis 1 were associated with taxa favoring backwater areas and pools that tolerate a wide range of temperatures and turbidity levels, and that have generalized diets. For example, warmouth are often found among macrophytes in slack water, and golden shiners avoid uplands (Jenkins and Burkhead 1994).

The larger biological disparity in the ordination of sites with large watersheds is due to a sample from the main stem of the Little Tennessee River near Needmore, N.C. This sampling site differs from upstream sections of the river due to the abundant exposed bedrock in the main channel. Needmore hosted many taxa found rarely in large watersheds and the following 6 species that were found only at that site: (1) spotfin chub (*Erimonax monachus*); (2) silver redhorse (*Moxostoma anisurum*); (3) river redhorse (*M. carinatum*); (4) rosyface shiner (*Notropis rubellus*); (5) tangerine darter (*Percina aurantiaca*); and (6) flathead catfish (*Pylodictus olivaris*). Due to this biological diversity and uniqueness, Needmore was weighted negatively and appears far to the left on axis 1 in Figure 4.5a. Also in the ordination of large sites, axis 2 separated sites that were located in the main stem of the Little Tennessee River from more upland sites. Needmore shared 12 of 18 taxa found at Cartoogechaye Creek near Muskrat Road, a sampling site whose score on axis 1 in Figure 4.5a is to the far right. With the exception of Needmore, the biological variation within large watersheds was comparable to that observed within the small watersheds.

Homogenization, Sediment, and Sculpin

Qualitative examination of species scores for small watersheds revealed some separation of highland endemic and cosmopolitan taxa (see Figure 4.4b). A large number of very common taxa, including highland endemics and cosmopolitans, occupies the center of the ordination space. Those taxa are found throughout the watershed. In contrast, Tuckasegee darters (*Etheostoma blennioides gutselli*), brown trout (*Salmo trutta*), greenfin darters (*E. chlorobranchium*), longnose dace (*Rhinichthys cataractae*), rainbow trout (*Oncorhynchus mykiss*), gilt darter (*Percina evides*), smoky sculpin (*Cottus bairdi* ssp), mirror shiner (*Notripis spectrunculus*), and rosyside dace (*Clinostoums funduloides*) were all clustered together in the upper right corner of Figure 4.4b in a region distinct from the cosmopolitan taxa scores. These upland restricted taxa appeared to favor streams in watersheds with steep slopes (SW) and mostly forested conditions (tf93).

Ordination plots suggested a greater degree of homogenization in large watersheds than in small ones (cf. Figure 4.4b, 4.5b). Species scores for highland endemics and cosmopolitan taxa spanned the full range of values on both axes 1 and 2 in the ordination of sites with large watersheds, so it was not possible to delineate distinct regions where highland endemics were weighted more heavily than cosmopolitan taxa. Nor could cosmopolitan species scores be separated from those of highland endemics. This suggests that geographic ranges for both highland endemics and cosmopolitan fishes were coincident, an indication of homogenization within the basin by native taxa (Scott and Helfman 2001). If native cosmopolitan taxa are displacing native highland endemic taxa in the Upper Little Tennessee, they would colonize larger, lower elevation sites first (Scott and Helfman 2001). These ordinations support that explanation.

Patterns for cosmopolitan or highland endemic taxa among small watersheds could not be related to land use variables using regression analysis. One possible interpretation is that minor differences in land use that did not greatly alter forest cover, modeled sediment loading rate, or other measured variables were associated with the shifts in assemblage structure noted in Figure 4.4. As a result, regression did not reveal the patterns that were more evident by examining whole assemblages directly. The converse can be argued with respect to regression models in large watersheds where sculpin proportions, highland endemic proportions, and cosmopolitan proportions were effectively predicted using linear regression against % forested area and modeled sediment loading rate. The contrast between regression results in small vs. large watersheds implies that modeled sediment loading rate, forest cover, and building density characteristics varied over a wider range of values in large watersheds, where more agriculture and development are concentrated, than in small watersheds. The greater variability in land use conditions in large watersheds was associated with larger shifts in fish assemblage structure than in small watersheds and therefore had more explanatory power.

Sculpin comprised a majority of the total number of individuals in the highland endemic group, so results for highland endemics, cosmopolitans, and sculpin are partially redundant. The correlation structure among proportions of each is provided in Table 4.6. Although these measures are not independent, each suggests that sediment loading rate has a significant influence on fish assemblages in the Upper Little Tennessee River basin, especially in large watersheds. In small catchments, sculpin proportions were statistically related to modeled sediment loading rate and % forested area, but neither highland endemic nor cosmopolitan taxa varied with those factors. This is a very different conclusion than is reached by analyzing the ordination results, and it points to the limitations of whole assemblage analysis for examining the effects of sediment on fishes in this river basin.

One reason for the equivocal ordination results with respect to the influence of sediment on fish assemblages is inherent in using CCA for direct gradient analysis. Taxa that are tolerant of a wide range of environmental conditions, for example suspended sediment or bed load, receive the same weight as more sensitive taxa. The 0.25-power transformation used also diminished the importance of abundance information in the weighted averaging. Although smoky sculpin (*Cottus bairdi*) appeared to be more

Table 4.6. Correlation structure among two categories of fishes and the most abundant taxon in the study.

	Endemic Proportion	Cosmopolitan Proportion
Endemic Proportion		
Cosmopolitan Proportion	-0.89	
Sculpin Proportion	0.89	-0.75

sensitive to sediment than other taxa, the fact that it was widespread in this basin led to sculpin having scores very near the mean for Axes 1 and 2 in each of the ordinations. This point about ubiquitous yet potentially sensitive taxa is further illustrated by the large group of nest associates that spawn over mounds constructed by river chubs (*Nocomis micropgon*). These taxa had species scores near the centroid of the ordination of sites with large watersheds. These "brood hiders" (Simon 1999) deposit their eggs in the well-oxygenated coarse substrate prepared by chubs to benefit developing embryos (Etnier and Starnes 1993, Jenkins and Burkhead 1994, Simon 1999). Because these taxa were common, their individual species scores were near the center of the ordination plot in Figures 4.4b and 4.5b. A final point about these CCA results is that species absence data do not influence CCA. Because fish collections continued until no new species were found, absence of a species from a site in this study reflects meaningful ecological information that is excluded by using CCA.

Whole fish assemblages in small and large watershed size categories appeared to be more influenced by elevation, watershed area, relief ratio, average hillslope, and average gradient within watersheds than by anthropogenic stressors. Ordination of all sites did not uncover relationships between whole assemblages and sediment loading that were as strong as those for sculpins, highland endemics, or cosmopolitan taxa. Analysis of assemblages by biogeographic category (Mayden 1987, Scott 2001) was necessary to reveal the influence of sediment on fish assemblages in the Upper Little Tennessee River basin.

In this study, the smallest unit of analysis was the stream reach and its watershed. Ordination and correlation analyses showed that watershed area, elevation, land use, and stream habitat types covary. Steep, rocky, upland headwaters contrast with larger, lower elevation, gently sloping, and sandier streams. These types of habitats support distinct faunal assemblages due to evolutionary adaptations; highland endemics prefer cool temperatures, low nutrient concentrations, and clean, coarse substrate for spawning while cosmopolitan taxa are suited to a wide range of temperatures and sandier substrates (Mayden 1987, Scott and Helfman 2001). Wear et al. (1998) also highlighted the contrast in this basin between (1) low elevation streams with large catchments and cleared floodplains with relatively high building and road densities near streams and (2) higher elevation streams with smaller catchments and minimal human settlement and development impacts. Those two distinct classes of streams and watersheds also carry distinct biological signatures, evidenced by an ordination that separated small and large watersheds based on the abundance and distribution of taxa within the basin (Figure 4.3). Elevation was correlated with major shifts in assemblage structure within both small (see Table 4.4) and large (see Table 4.5) catchments.

Modeled sediment loading rate was a statistically significant predictor of proportions of smoky sculpin (*Cottus bairdi*) in both watershed size classes. This species prefers coarse gravel or cobble substrates for spawning (Etnier and Starnes 1993, Jenkins and Burkhead 1994, Simon 1999) and has been shown to be sensitive to excessive sediment in the channel (Sutherland et al. in press). Sculpin are the most abundant fish species in the basin. Modeled sediment yield rates also increased monotonically with unimproved road density in a statistically significant way (see Figure 4.2). The correlation reported here is not surprising because RUSLE, on which the sediment yield model was built, assigns a larger proportion of soil loss to unimproved roads than to other land uses (see Table 3.1). Nearly fifty years of research on sediment in forested catchments in this study area have suggested that forest access roads built for logging operations are the major sources of sediment to high elevation streams with steep slopes and forested catchments (Tebo 1955, 1957, Swift 1984, 1988). When a forest was clearcut, roads and skid trails were the major source of sediment reaching streams (Gurtz et al. 1980). Because unimproved road density (km km^{-2}) is easy to measure using digital map information, it could serve as a surrogate for separating forested watersheds in this study area according to expected sediment yield rates. This study suggests that unimproved road density can be a useful indicator of watersheds where sculpin populations may be depressed due to hillslope-derived sediment.

Conclusion

The first objective of this study was to determine whether basin physiography separated fish assemblages into distinctive groups. Distinctive fish assemblages were noted in small (< 39 km^2) and large (> 39 km^2) watershed size categories. The most important watershed descriptors associated with broad shifts in fish assemblage structure in either size category were watershed area, watershed relief ratio, average hillslope in the basin, outlet elevation, and average stream gradient in the stream network above sampling sites. The second objective was to identify patterns in fish assemblage structure associated with an important anthropogenic stressor, hillslope-derived sediment. Those patterns were more evident by analyzing sensitive and insensitive taxa independent of whole assemblages.

Although small and large watersheds supported distinctive faunas, more continuous variation of species abundance vs. environmental descriptors was observed within both size categories. In small watersheds, forest cover, average hillslope, and outlet elevation were the primary correlates of shifts in assemblage structure. In large watersheds, outlet elevation, relief ratio, and average stream gradient were associated with shifts in species abundance and distribution. Large watersheds and small watersheds had different biological and physiographic characteristics, which suggested different influences on fishes in each watershed size category. Modeled sediment loading rates had consistently low correlations with any of the CCA axes examined among either small or large watersheds.

In small watersheds, ordination separated sites into those dominated by cosmopolitans, those having both cosmopolitans and highland endemics, and sites dominated by highland endemics. Characteristics of sites in each group can be summarized as follows. Low elevations, low relief ratios, and average slope characterized the sites dominated by cosmopolitan taxa. The area of overlap was characterized by average values for all the watershed metrics. Areas where highland endemics dominated had higher elevations and higher relief ratios than other sites. In large watersheds, where building and agricultural activities are concentrated near streams (Wear and Bolstad 1998), ordination did not separate sites with greater proportions of cosmopolitans from sites with greater proportions of highland endemics. Modeled sediment loading rates were negatively related to sculpin and highland endemic proportions and positively related to cosmopolitan proportions. These observations are consistent with Scott and Helfman's (2001) postulate that highland endemics flourish in steep, cool-water, low-nutrient streams with forested catchments and that native cosmopolitan taxa can displace highland endemics when anthropogenic disturbance increases temperatures, nutrients, or sediment reaching streams of the southeastern Appalachian highlands.

In both watershed size classes examined, hillslope-derived sediment loading rate was a strong predictor of the relative abundance of smoky sculpin (*Cottus bairdi*), the most common species in the basin. This species is also the most prevalent taxon among the highland endemics, which are thought to be most sensitive to a broad range of anthropogenic disturbances (Scott and Helfman 2001). Cosmopolitans are thought to displace highland endemics in disturbed stream habitats. In large watersheds, but not in small ones, regression showed that relative abundances of highland endemics and cosmopolitan fishes could be predicted using modeled sediment loading rate and % forest cover with building density as an additional statistically significant predictor for highland endemic and cosmopolitan fishes.

Sculpin, highland endemics, and cosmopolitans were statistically related to modeled sediment loading rates, but modeled loading rates were not associated with broad shifts of whole assemblages. Sediment modeling, or even simple indicators of sediment impacts such as unimproved road density, can be used to identify the sites and catchments most likely to have chronic stresses to fishes and their habitats due to hillslope-derived sediment.

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CHAPTER 5

LINKING STREAM ECOSYSTEMS AND LANDSCAPE TRAJECTORIES IN THE SOUTHERN APPALACHIANS[1]

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Abstract.

Land use projections were made within two watersheds, the Upper Little Tennessee River and Cane Creek, in western North Carolina using a previously-published model, and eight stream sites were chosen for long-term research to identify the consequences of predicted land use changes. The eight sites are drawn from three distinct groups with different land use trajectories: forested (n=2), rural (n=3), and suburban (n=3). The *a priori* groupings were based on land cover, which differed in both 1970 and 1993. Algal biomass (Chlorophyll-*a* $\mu g\ m^{-2}$ and ash-free dry mass $mg\ m^{-2}$), median substrate particle size, sediment core particle size distributions, and total suspended solids ($mg\ l^{-1}$) measures did not differ among watershed land use categories. Fish collections and water chemistry and temperature data from the eight sites were combined with data collected at eight other sites in the study area in previous years. Four categories of watersheds were extracted from combined fish assemblage data using non-metric multi-dimensional scaling ordination methods. They were labeled “forested”, “rural”, “suburbanizing farmland”, and “suburban/urban” to capture the past and projected land uses in each set of watersheds. Water chemistry and fish guild structure data were compared using nested ANOVA. Stream chemistry was statistically different among watershed categories. NO_3, K, Ca, and Mg each increased from forested to rural to suburbanizing farmland to suburban/urban. Na was much higher in the less forested catchments than in the more forested watersheds; within those groups, Na was somewhat higher in forested than rural and somewhat higher in suburbanizing farmland than suburban/urban watersheds. Stream temperature generally decreased from suburban/urban to suburbanizing farmland to rural to forested catchments, although rural

and suburbanizing farmland catchments had very similar temperatures. Percentage of omnivores increased from forested to rural to suburbanizing farmland to suburban/urban watersheds. Invertivores and widespread taxa decreased in percentage across that set of categories. Lithophilic nest spawners and endemics decreased in percentage across the four sets. Compared to forested and rural streams, suburbanizing farmland and suburban/urban watersheds have higher nutrient concentrations, lower percentages of invertivores, specialized insectivores, and lithophilic nest spawners; and they have higher percentages of omnivores, pelagophils, polyphilic nest spawners and widespread taxa. Biological integrity is maximal in rural watersheds and minimal in urban ones. Land use data were extracted for 1970 and 1993 for all watersheds in the larger study area meeting size and elevation constraints. Those 149 watersheds were classified into the four groups identified through biological and physical analyses. Two trajectories of stream ecosystem response to predicted future land use were identified. If building and road densities increase, streams in forested watersheds are likely to be warmer, higher in nutrients, and support more omnivores and widespread taxa; proportions of invertivores, lithophilic nest spawners, and endemics would decrease. In suburbanizing farmland watersheds that become more like suburban/urban ones, the same predictions hold. This study classifies land use patterns and associated stream ecosystem states, thereby providing insights into past, present, and future land use effects on streams in western North Carolina.

KEYWORDS: Stream, fish assemblage, nutrients, land use, Geographic Information Systems, socioeconomic modeling

Introduction

There are many consequences for stream processes when land use shifts from forested to agricultural or to urban. Documented examples include heightened sediment loading rates (Judy et al. 1984), higher nutrient concentrations (Hampson et al. 2000), increased stream temperatures, and geomorphic changes such as channel incision (Doyle et al. 2000) and habitat homogenization (Berkman and Rabeni 1987). Ecologists also have linked socioeconomic patterns to those same physical processes (Carpenter et al. 1999, Strange et al. 1999, Wilson and Carpenter 1999, Grimm et al. 2000). This literature documents human impacts on streams at multiple scales. Human social and economic processes operate over large spatial extents (i.e. cities or regions). Those social processes affect land use ownership patterns that result in land use change (Turner et al. 1996), and land use change influences stream water (Hampson et al. 2000), geomorphic (Doyle et al. 2000), and habitat (Berkman and Rabeni 1987) qualities. Understanding the processes that yield landscape transformations over large areas, such as human land use patterns or climate change (Meyer and Pulliam 1992, Vitousek 1994), helps in understanding patterns and processes in specific streams.

The first objective of this research is to introduce a unique synthesis and approach to studying the many influences of increased watershed building density on stream ecosystems. This objective requires integration of many approaches: (a) land use change modeling and analysis in the context of social and economic patterns; (b) site selection using geographic information systems (GIS); (c) biological and physical monitoring of streams; and (d) forecasting stream ecosystem attributes based on land use change projections. The second objective is to set these analyses in the context of land cover and land use conditions throughout the study area. Results are extrapolated to possible stream ecosystem states in a large region using widely available spatial information. These objectives, methods, and results synthesize socioeconomic modeling, GIS science, and the study of land use effects on stream ecosystems. This research applies site-specific information to issues in a larger region and, conversely, regional information is used to guide analyses of individual sites.

Study Area and Methods

The entire study area, two focal watersheds, and the study sites used in these analyses are shown in Figure 5.1. The study spans four sub-basins of the Upper Tennessee River system: the Upper Little Tennessee (LT), Tuckaseegee, Pigeon, and Upper French Broad (FB) basins. The backwater reaches of Lake Fontana define the downstream limits of the Upper Little Tennessee and Tuckaseegee River basins studied here. The North Carolina border with Tennessee defines the northern extent of the Pigeon and French Broad study areas. These are western North Carolina's major river systems in the Blue Ridge physiographic province (Wallace et al. 1992). Elevations in this entire study area range from 400 to 2000 m above sea level. Land use is largely rural with a few urban centers. Asheville, located in the north-central portion of the French Broad Basin, is the largest city in the region. Franklin is a small urban center in the Little Tennessee basin. Canton is an industrial center in the Pigeon River basin. Watershed area and site elevation are each correlated with shifts in assemblage structure for fish (see Chapter 4), so analyses were restricted to watersheds that varied in size between 10 and 40 km^2 and whose outlet elevations were between 550 and 720 m above sea level. Forests and scattered homes fill the slopes whereas agriculture, light industry,

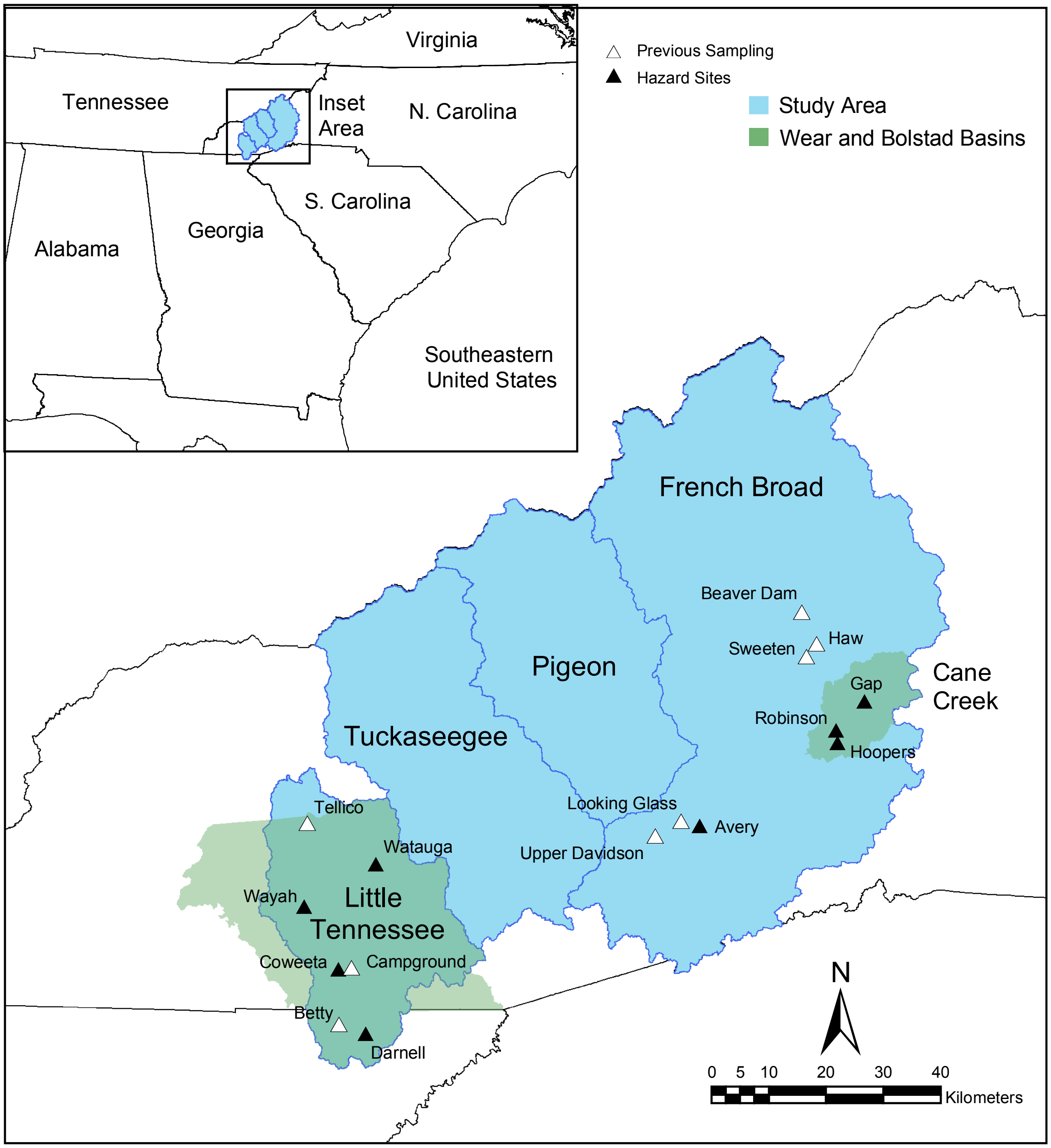

Figure 5.1. The study area encompasses the four largest drainages in the southern crystalline Blue Ridge in western North Carolina. Stream data are drawn from 16 sites, six of which were chosen based on projections of land use change that is expected to alter stream ecosystem conditions. Area in green depicts study areas of Wear and Bolstad (1998) for which future land use predictions had been generated. Triangles are sites sampled for fish, water chemsitry, and stream temperature by Scott (2001). Black triangles are sites selected for long term analysis based on projected land use.

tourist-oriented businesses, small urban centers, and suburban land uses predominate in the valleys. The Coweeta Long Term Ecological Research site is several miles north of the Georgia border in the Little Tennessee River system.

Likely future land use was examined for two watersheds: the Upper Little Tennessee River basin south of Swain County, North Carolina and Cane Creek, a tributary of the French Broad River. The following descriptions are interpreted from conclusions drawn by Wear and Bolstad (1998) for these basins. In the Upper Little Tennessee River basin, most of the land use/land cover changes between 1950 and 1990 were the conversion of non-forested to forested land cover and from forest without buildings to forest with buildings. The social and economic factors that underlie that transition suggest that land use has shifted from agricultural and forested land uses toward rural second home development. The socioeconomic setting in the Cane Creek study basin contrasts with that of the Upper Little Tennessee. There land use transitions between 1950 and 1990 were more evenly split among forest clearing and reforestation. Roughly equal proportions of cleared and reforested areas supported increased housing density between 1950 and 1990. The Cane Creek watershed had higher proportions of agricultural land use in both 1950 and 1990 as well as substantially more areas with higher housing densities (>8 buildings ha^{-1}) than the Little Tennessee River watershed. The two basins, then, can be characterized as undergoing (a) second home development, in the case of the Upper Little Tennessee system and (b) suburbanization to supply the housing needs of Asheville, in the case of Cane Creek.

Forecasting Assessment

Studies of development impacts on streams usually make statements *post priori* regarding the presumed effects of human activities on stream ecosystems because research is conducted after development has occurred and after its impacts have reshaped the geomorphic, nutrient, productivity, and faunal states of a stream ecosystem. This study used projections of land use (Wear and Bolstad 1998) to examine the locations of likely future building activities and subsequently established a set of stream research sites where ecosystem changes will presumably occur in coming years. The site selection process is outlined in Figure 5.2. The greatest increases in sediment flux, nutrient supply, and runoff were expected for land use/cover changes from “undeveloped” to “developed” conditions, i.e. the conversion of forested land with no buildings to non-forested land with buildings. The definition of “undeveloped” used hereafter, and in the lower right hand portion of Figure 5.2, is a land parcel that had been forested since 1950 and that had fewer than 1 building per hectare in both 1950 and 1990.

A primary concern in selecting sites was to limit the confounding influence of the “ghost of land use past” (Harding et al. 1998). Forested sites were defined as having been forested and free of building development over the past 50 years. Disturbed sites could include locations that were forested in 1990 but that had been deforested in 1950. Much of the forest cover that is present in the Upper Little Tennessee River and Cane Creek basins had been deforested in 1950.

Two methods were used to examine probable building development at all undeveloped pixels. The first method measured the difference between predicted building density for 1990 and observed building density in 1990 (Wear and Bolstad

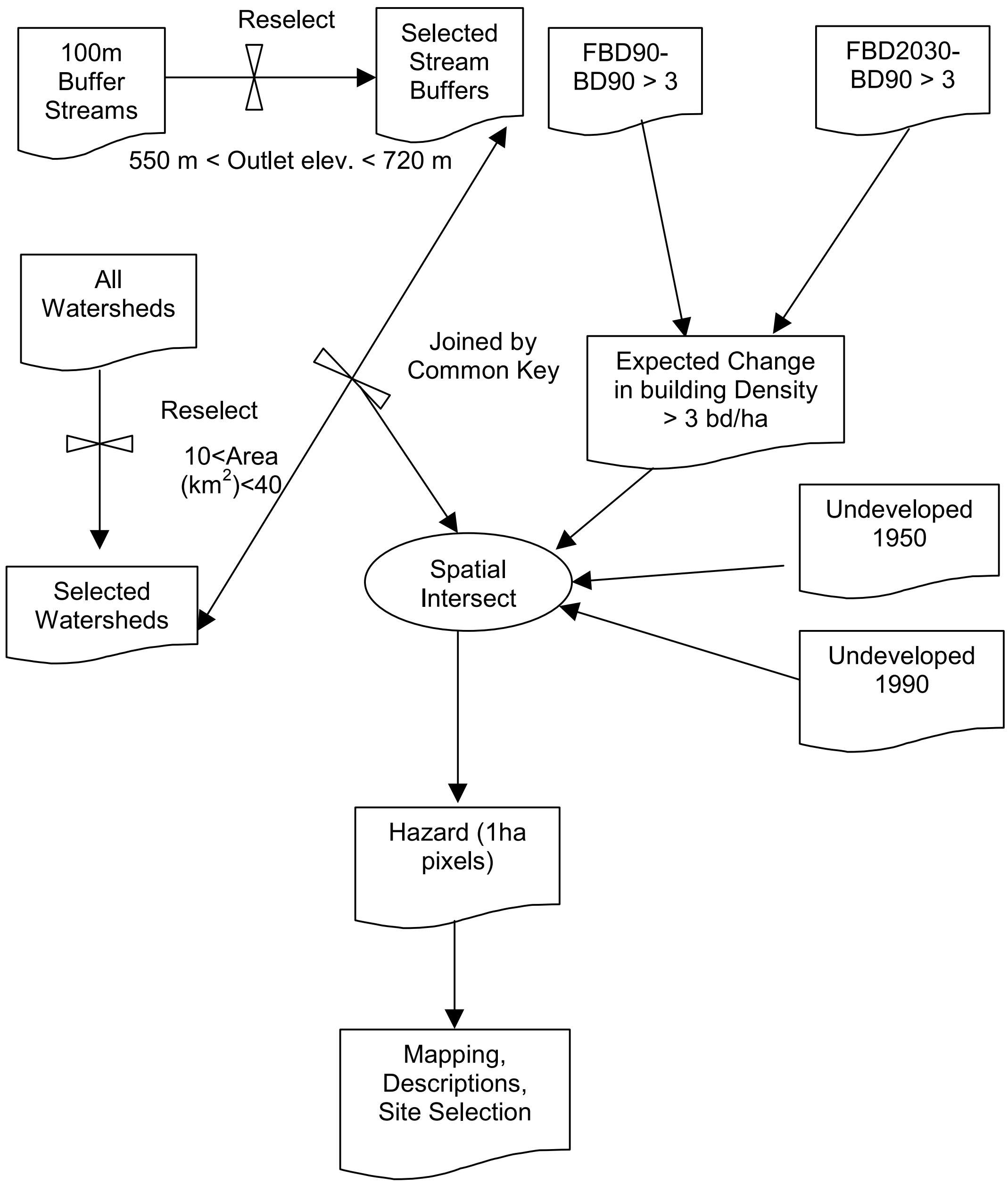

Figure 5.2. The composite hazard index was composed of (a) watershed size and outlet elevation criteria, (b) two projections of large building density increases, and (c) identification of land parcels that had been undeveloped for the past 50 years. The difference between observed building density (BD) and forecast building density (FBD) was measured using 1990 projections and 2030 projections (Wear and Bolstad 1998). See text for details.

1998). This was a measure of the untapped profitability inherent in land parcels that were accessible and in desirable locations in 1990 but where new buildings had not yet been built. The second index of development hazard was the difference between forecast building density for 2030 (Wear and Bolstad 1998) and observed building density for 1990. This was a measure of the expected change in building density at each site. Where each index exceeded 3 buildings ha^{-1}, it was assumed that there was a very high likelihood that building developments were going to proceed in the coming 20 years. These measures of likely future building development were incorporated into a composite "hazard index" which served to delineate locations that had been undeveloped for at least 50 years and where large increases in building density were predicted in coming decades. Greatest changes to streams were expected when building activities were to be concentrated near streams, so hazard indices were examined within 100 m of streams. In summary, the following characteristics, which are depicted in Figure 5.2, defined the composite hazard index: (a) forested and fewer than 1 building ha^{-1} in 1950; (b) forested and fewer than 1 building ha^{-1} in 1990; (c) 1990 forecast building density - 1990 observed building density > 3 ha^{-1}; (d) 2030 forecast building density - 1990 observed building density > 3 ha^{-1}; and (e) within 100 m of streams mapped on 1:24,000 scale topographic maps. All pixels were mapped where all of these conditions were met.

Watersheds containing large numbers of pixels meeting the composite hazard index criteria were identified and visited. Buildings were already being constructed in many of those watersheds. Of the watersheds identified through GIS analysis as meeting criteria in Figure 5.2, six had not yet experienced significant increases in building density near streams and were selected for long term study (Figure 5.1; Table 5.1). The three LT

Table 5.1. Land use, % composite hazard index pixels within 100 m of streams, algal, and geomorphic factors among the eight sites chosen for long term research. Composite hazard indices are described in the text and in Figure 5.2. Quantities in the bottom half of the table are ash-free dry mass (AFDM); chlorophyll-*a* (chl-*a*) concentration; phi (ϕ) is the $-\log_2$ of the mean medial axis size of substrate; fines are defined as particles < 2 mm in sediment cores; and total suspended solids (TSS) concentrations.

River Basin	***Upper Little Tennessee River Basin***				***Upper French Broad River Basin***			
Land Use Category	**Rural Home Development**			**Reference**		**Suburbanization**		
Site Name	*Darnell*	*Wayah*	*Watauga*	*Coweeta*	*Avery*	*Gap*	*Robinson*	*Hoopers*
Land Use and Projected Development								
% Forest 1993	98.4	97.5	84.9	98.5	99.5	71.5	65.5	71.6
% Developed 1993	0	0.11	5.26	0	0.02	0.86	6.15	0.63
% Agriculture 1993	1.53	2.22	8.81	1.19	0.44	27.47	27.88	27.57
% Forest 1970	98.9	98.2	84.0	99.5	99.1	70.3	64.4	72.7
% Forest 1950	97.4	98.0	76.4	98.4	100	64.3	63.9	67.5
% Composite Hazard Index in 100 m Stream Buffer	< 1	2.17	2.05	<1	<1	7.68	7.62	3.40
Algal and Geomorphic Conditions								
AFDM (mg m^{-2}) (s.d.; n=3)	1.1 (1.3)	4.8 (11)	1.7 (0.84)	0.85 (1.1)	0.88 (0.29)	2.2 (0.90)	0.8 (0.84)	1.2 (0.98)
chl-a (μg m^{-2}) (s.d.; n=3)	115.7 (112.4)	89.1 (42.8)	109.1 (119.7)	35.1 (21.2)	60.4 (47.4)	299.6 (226.9)	42.4 (29.6)	48.7 (18.1)
Average ϕ (n=100)	-4.4	-5.3	-3.8	-4.6	-5.5	-5.1	-2.9	-2.6
% Fines (n=3)	9.6	10.0	15.5	3.3	13.7	13.1	18.4	36.1
TSS (mg l^{-1}) (s.d.; n=3)	14.7 (22.7)	13.3 (6.1)	132.3 (14.0)	6.1 (4.7)	3.2 (0.8)	14.7 (2.8)	8.0 (2.8)	19.2 (7.6)

sites are in watersheds experiencing rural home development while the three Cane Creek sites are in an agricultural area that is shifting toward suburban land use (Wear and Bolstad 1998). Two additional reference streams were located on US Forest Service (USFS) research facilities, where future land use is predicted to change little from present land use. The first reference stream is Coweeta Creek, within the USFS Coweeta Hydrologic Laboratory in the LT basin. The second reference stream is Avery Creek, near the Bent Creek USFS Experiment Station in the FB system.

An accuracy assessment was conducted to test assumptions underlying land cover change forecasts. Land cover and building density had been measured in 1990, so changes between 1990 and 1993 were measured to ensure that buildings had not already been erected where increases in building density were predicted by the hazard index described above. Land cover and building density were interpreted from a high-resolution orthophoto mosaic built from aerial photographs taken in 1993 (see Chapter 3). The digital orthophotographs had a nominal pixel resolution of 1.4 m and a spatial root mean squared error ($RMSE_{xy}$) of less than ±5 m. The software package Imagine (ERDAS 1999) was used to select 100 1-ha pixels that met the composite hazard criteria defined above (Figure 5.2). Wear and Bolstad (1998) had reported very high (> 95%) land cover accuracies for both 1950 and 1990. Land cover and building density (in 1993) were recorded in each randomly-selected cell based on interpretation of the 1993 CIR orthophotographs. “Forested” was defined as having > 75% of the 1-ha pixel covered with forest vegetation. “Non-forested” was the inverse. Changes in building density or forest between 1990 and 1993 cover were then recorded.

Ecosystem Assessments

Algal biomass, geomorphic, fish assemblage, water chemistry, and stream temperature data were collected once during the summer of 2000 at the eight hazard sites and eight additional sites examined in previous research associated with the Coweeta LTER stream research program (Harding et al. 1998, Jones et al. 1999, Scott 2001). The eight additional sites included 5 less developed and 3 urbanized watersheds that met the size and elevation criteria described above. The less developed watersheds were Upper Davidson, Looking Glass, Betty, Campground, and Tellico. Two of these streams, Upper Davidson and Looking Glass, are on US Forest Service land and are tributaries of the Davidson River in the FB basin. Betty, Campground, and Tellico are each tributaries of the LT. The developed watersheds were all near the urban center of Asheville, North Carolina and were at Beaverdam, Haw, and Sweeten creeks. Together, these 16 sites exemplified a wide range of watershed land uses, including forested, agricultural, and developed conditions. Below are methods descriptions for (a) algal and geomorphic data collected only at the eight hazard sites and (b) fish and water quality sampling methods for all 16 sites.

Hazard sites

Algae, suspended and deposited sediment, and substrate size were measured at the eight hazard sites to assess current conditions and to provide a reference point for future research in these streams. Benthic periphyton was collected with a device similar to that described by Loeb (1981). The apparatus consisted of a 20 cm plexiglass cylinder with a 4.3 cm diameter (area 14.7 cm^2). A rubber gasket, fitted to the end of the sampler, was pressed against the substrate to ensure a water-tight seal between the sampler and substrate. A round scrub-brush inside the sampler was used to dislodge algae from the substrate. Three transects were chosen randomly from 10 possible transects at 10-m intervals above the downstream end of the sampling site. Three algae samples were collected from the dominant substrate along each transect and combined; three sets of transect data were analyzed for each site. Composite samples were placed on ice and transported to a lab for processing. Algae samples were sub-sampled to determine chlorophyll-*a* and ash-free dry mass (AFDM) concentrations. Pigments were extracted using 90% acetone buffered with ammonium hydroxide; chlorophyll-*a* concentration was measured using standard fluorometric methods (Wetzel and Likens 1991). Different sub-samples were filtered through pre-weighed 47 mm, 0.45 μm Whatman® glass fiber filters. Filters were then dried in a 105° C oven for 24 hours and weighed to determine ash-free dry mass (AFDM mg m^{-2}).

Sediment core samples were collected using a 60 cm height x 25 cm diameter stainless steel cylinder. Sediment cores were located in three riffles and three pools at each site. The coring device was inserted into the streambed, and all sediment was removed to a depth of 10 cm wherever bedrock or boulders were not reached first. Large sediment (> 64 mm) was removed first and weighed in the field. All remaining sediment was transported to a laboratory for processing. Samples were dried at 105°C for three weeks, or until completely dry. Samples were then sieved into coarse (2 – 64 mm) and fine (< 2 mm) size fractions and weighed. Thus, % coarse (> 2mm) and % fine (< 2mm) size fractions were determined.

Pebble counts were conducted to measure the particle size distribution of the stream bed. One hundred particles were picked up and measured while traversing a ‘zig-zag' pattern at each site. Particles were measured in riffle-run habitat; pool habitat was avoided. This affords direct comparison of similar habitats among sites. Medial axes of particles were measured to the nearest millimeter. Measurements were then converted to phi size (ϕ; negative base two logarithm of medial axis). Average phi was calculated for comparison among sites.

Total suspended solids (TSS) samples were collected at baseflow at each site. Three 125 ml water samples, to be used for determining water chemistry, were collected from the thalweg of 'run' habitat. Water samples were filtered with a hand pump in the field onto pre-ashed, pre-weighed 47 mm, 0.45 μm Whatman® glass fiber filters. These were placed in aluminium foil envelopes, dried in a 105° C oven for 24 hours, and weighed. Filtrates were placed on ice and transported to lab for water chemistry analyses described below.

Sites in Larger Region

At hazard sites, fish were collected between April 16th and July 6th 2000 using a backpack electroshocker, seines and dip nets. Scott (2001) collected fish using the same methods but in 1995 and 1996. At each site, a quantitative sample was taken during one thorough pass within a representative 50 m reach. An attempt was made to equalize electroshocking time per area sampled to ensure comparable catch per unit effort at all sites. Fishes were identified to species and assigned to feeding and reproductive guilds (Appendix 5A) and to distributional categories to facilitate comparisons of sites. Feeding guilds included invertivores, specialized insectivores, herbivores, omnivores, and piscivores. Reproductive guilds followed Simon's (1999) scheme: pelagophils, lithophilic broadcast spawners, lithophilic brood hiders, lithophilic nest spawners, and polyphilic nest spawners. Distributional traits included two categories (Scott and Helfman 2001): widespread native taxa (cosmopolitan) and highland endemics (Mayden 1987) were determined for each site. Percentages of individuals in each of the above groups were analyzed.

Physical assessments focused on water chemistry and temperature. At all 16 sites, filtered water samples (0.47 μm; see TSS sampling above) were analyzed for NH_4, NO_3, SO_4, PO_4, K, Na, Ca, and Mg using methods outlined in Deal (2001). Water chemistry samples were collected concurrently with fishes. Stream temperature data were collected at hazard sites from August 3rd to September 15th, 2000 and from August 3rd to September 15th, 1999 for Scott and Helfman's sites. Temperature was logged automatically every 2 hours during those periods. Mean late summer temperature was determined for each site by averaging bi-hourly data over the six-week period for which these data were collected at each site.

Statistics

Ordination was used to extract patterns from fish collections among the 16 sites. Sorensen dissimilarities, also known as Bray and Curtis dissimilarities (McCune and Mefford 1999), were calculated among sites based on the fourth root of the abundance of each species. This distance measure has been found to preserve important information about assemblage structure while relieving bias introduced when a few species represent most of the catch (Clarke 1993, Faith et al. 1987). Ordination was conducted in PC-ORD (McCune and Mefford 1999) using nonmetric multidimensional scaling (NMS) on the dissimilarity matrix. This technique is non-parametric and has been shown to be more robust than other common methods of ordination (Minchin 1987). The ordination provided a set of watershed classes based on distinct biological signatures among the 16 sites examined. These classes were labeled and related to broader regional conditions and were the basis of subsequent analyses. Water chemistry, temperature, and fish assemblage structure observations were analyzed using a nested ANOVA design that is explained below, after describing the ordination results. Some of the fish guild data had high inter-correlations. For each guild, Bonferonni-corrections were used based on the number of other guilds with which a guild had correlations exceeding 0.7.

Watershed Classification

All watersheds in the study area meeting size and elevation requirements were classified using land cover data. Digital elevation models (Level 1, USGS 1993) were used in conjunction with digital line graphs of 1:24,000 scale topographic maps to extract watershed boundaries for every mapped stream in the study area using Arc/Info (ESRI 1998; see Chapter 2). Land cover data from 1970 (Wear and Bolstad 1998) and 1993 (Hermann 1996) were used to extract % forested area in 1970, % forested area in 1993, and % agricultural area in 1993. These land use data descriptions were extracted for entire watersheds and within 100 m of mapped streams within each watershed. Improved and unimproved road densities (km km^{-2}) were recorded because they are indicators of potential hydrologic change induced by impervious surfaces (Booth and Jackson 1997, Forman and Alexander 1998) and sediment input (Swift 1988, Swift 1984), respectively. These land cover and land use data were extracted for all watersheds in the entire study area (n = 149) meeting elevation (550 m < elevation at outlet < 720 m) and size criteria (10 km^2 < watershed size < 40 km^2). Data were transformed to approximate normal distributions (angular transformations for percentages; log transformation otherwise;

Sokal and Rohlf 1995). Watershed data were then clustered using the k-means algorithm (Hartigan and Wong 1979) in S-plus (Mathsoft 1999). Four classes of streams were identified based on water quality and fish assemblage structure data.

Results from physical, biological, and land use assessments were synthesized to describe current watershed conditions and possible consequences of land use changes on stream ecosystem structure and function throughout the entire study region.

Results

Forecasting

Land use changed measurably between 1990 and 1993 with respect to forested area and building density. Of the randomly chosen 1-ha pixels with a high hazard index value (n = 100), 15% had buildings in 1993 that were not observed in 1990 by Wear and Bolstad (1998). Most of the pixels that had been classified as forest in 1990 appeared to be forested in 1993, but 11% witnessed a decline in forested area during that 3-year period. A small percentage of area that experienced deforestation (4% of sampled pixels) had no buildings in 1993. Approximately 2/3 of the area that was deforested between 1990 and 1993 supported buildings in 1993. In total, 17% of sampled pixels were either deforested, supported new buildings, or had both by 1993. Areas that had been forested since the 1950's and that Wear and Bolstad (1998) had suggested would be the sites of future development were indeed under development pressure during the period between 1990 and 1993.

Ecosystem Assessments

Hazard Sites

Algal biomass and geomorphic conditions did not differ greatly among the eight hazard sites (Table 5.1). Ash-free dry mass (AFDM, mg m^{-2}), chlorophyll-*a* ($\mu g\ m^{-2}$), substrate size (ϕ), % fines (< 2mm) by weight in sediment cores, and TSS were measured at the hazard sites only. Neither AFDM nor chlorophyll-*a* differed among sites. The two were linearly related by a factor of roughly 8:1 (AFDM: chlorophyll-*a*) with the exception of Wayah Creek, for which the ratio was closer to 50:1. Average ϕ was greatest (particle size was smallest) and the percentage of fine sediments in cores was greatest at Hoopers Creek in the Cane Creek sub-basin. Watauga Creek appeared to have the highest TSS values. Observed TSS data were similar to storm flow TSS values reported by Sutherland (1998). TSS is highly dependent on antecedent conditions, so useful comparisons are usually gained when these data are collected under calibrated conditions, for example following the falling hydrograph limb of storms of similar magnitude or during baseflow following a storm. As with periphyton, core particle sizes, and benthic substrate measurements, TSS data were too variable to make statistically valid comparisons among reference sites and the Little Tennessee or Cane Creek study sites.

Regional Sites

While algal or geomorphic assessments were not distinct when comparing the LT, Cane Creek, and reference sites, fish assemblages separated groups in the larger region into four distinct sets of sites (Figure 5.3). A two-axis solution explained 88% of the information in the Sorensen dissimilarity matrix for the fishes collected at all 16 sites

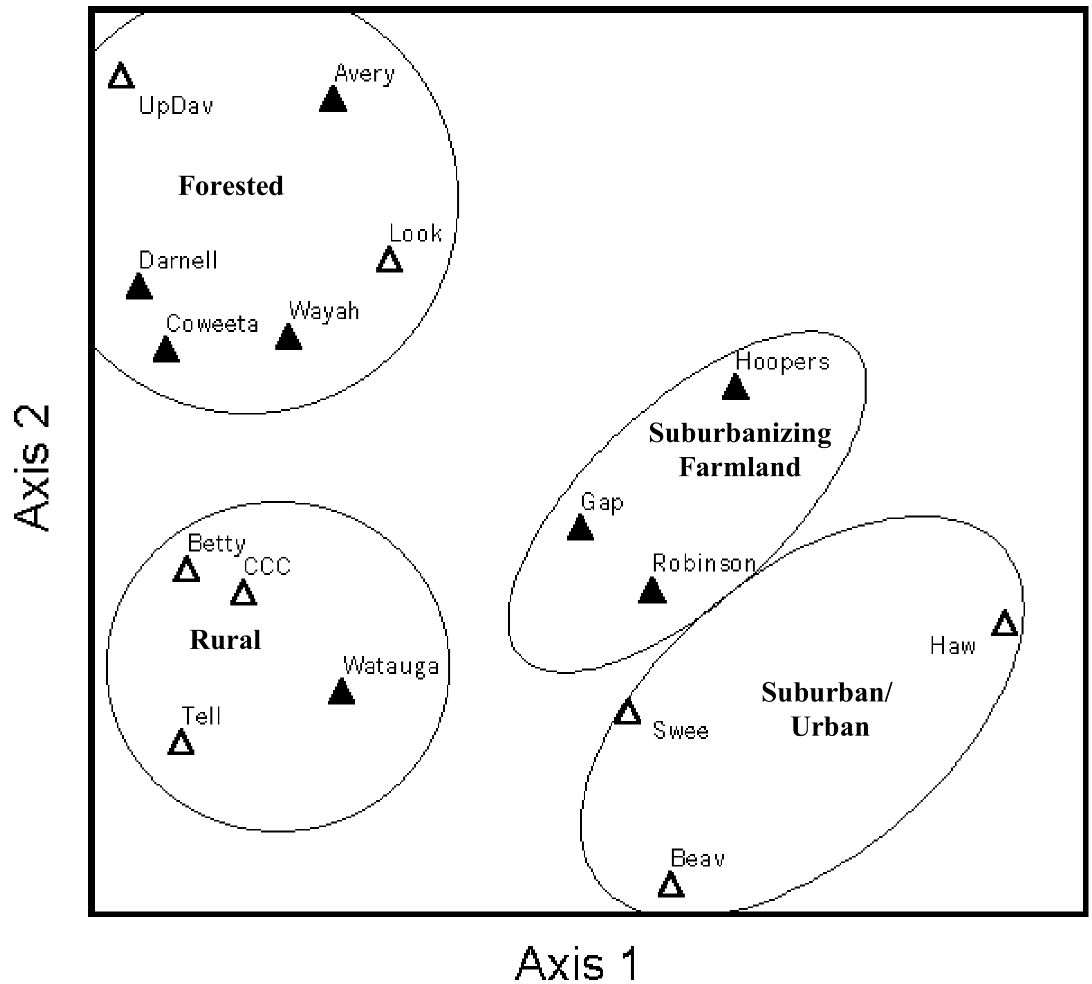

Figure 5.3. Ordination results demonstrate that fish assemblages separate four distinct groups from the 16 sites throughout the study area. These two axes explained 88% of the variance in the original dissimilarity matrix.

with a minimal stress of only 8.8 (McCune and Mefford 1999). The first axis separated fishes found in forested sites (low scores on axis 1) from fishes found in less forested watersheds. Transformed 1993 % forested cover in the watershed was a strong explanatory variable for axis 1 scores using linear regression ($R^2 = 0.69$; $p < 0.0001$). The second axis was interpreted as a gradient from more developed watersheds with low scores to less developed watersheds with higher scores. Mean log-transformed water temperature in late summer explained 60% of the variation in axis 2 ($p < 0.0001$), suggesting that land uses that increase water temperature separate sites along this axis. Examples of land uses that can increase water temperature include pavement or clearing of riparian vegetation.

The eight hazard sites were separated into the following four groups based on the ordination of fish assemblages from all 16 sites (Figure 5.3): (a) “forested” sites included Coweeta, Avery, Darnell, and Wayah; (b) “rural” included Watauga; (c) “suburbanizing farmland” included Hoopers, Gap, and Robinson; and (d) “suburban/urban” did not include any hazard sites. Forested watersheds had the lowest building and road densities. The forested sites included the US Forest Service sites in the LT and FB basins as well as Darnell and Wayah Creeks. Rural watersheds had higher building and road densities but were still predominantly forested. There were no FB sites in the rural category. Betty, Campground, Tellico, and Watauga were grouped with rural sites and supported mixed land uses, including agriculture and housing. Suburbanizing farmland sites had substantially less forested area than the forested and rural categories; this distinction had been true historically as well (Table 5.1). All three Cane Creek sites clustered together in the suburbanizing farmland category. Suburban/urban sites had the highest road and building densities, and the sites near Asheville were all grouped together in this category.

The four groups are distinct from one another, but they can also be grouped in two nested sets where development intensity is nested within proportion of forested area. Rural watersheds are more developed than forested ones, and suburban/urban watersheds are more developed than suburbanizing farmland ones. The former two are more forested than the latter two. Subsequent analyses used that scheme in a nested ANOVA to draw inferences about the impacts of development on streams in forested and less forested watersheds; degree of development was nested within degree of deforestation. These ANOVA also tested the validity of the classes established on the basis of biological data only.

Development intensity within forested and less forested watersheds was associated with statistically significant differences among chemical concentrations within these streams. Cation concentrations (K, Na, Ca, and Mg; Table 5.2) varied consistently among the nested categories of streams analyzed. Streams in deforested watersheds had higher concentrations than forested ones; increased development intensity was associated with higher concentrations within both groups. Concentration of SO_4 and NO_3 followed the same pattern. NH_4 appeared to be at higher concentrations in streams of developed watersheds, but there was only very weak evidence that development intensity was a statistically-significant nested factor. Stream temperature variation was statistically related to development intensity within forested and less forested watersheds.

Forested sites included more rainbow trout (*Onchorhynchus mykiss*), brown trout (*Salmo trutta*), longnose dace (*Rhinichthys cataractae*), and mottled sculpin (*Cottus*

Table 5.2. Mean values and nested ANOVA results for streams in forested and less forested watersheds; degree of development is nested within degree of forestation. Transformations were used to approximate normality. Chemistry and total catch data were log-transformed. Angular transformations (Sokal and Rohlf 1995) were used for remaining assemblage structure data.

	Forested		***Less Forested***		**Pr > F**
	Less Developed (n=6)	*More Developed (n=4)*	*Less Developed (n=3)*	*More Developed (n=3)*	
	Physical and Chemical				
NO_3 (mg l^{-1})	0.044	0.083	0.326	0.692	0.004
NH_4 (mg/l^{-1})	0.011	0.014	0.021	0.059	0.13
O-PO_4 (mg l^{-1})	0.018	0.024	0.021	0.025	0.95
K (mg l^{-1})	0.534	0.691	1.831	2.014	0.0008
Na (mg l^{-1})	1.523	1.438	4.847	4.839	0.0006
Ca (mg l^{-1})	0.980	1.286	5.360	6.589	0.0001
Mg (mg l^{-1})	0.407	0.482	1.728	2.838	0.0001
SO_4 (mg l^{-1})	0.968	1.145	2.754	4.922	0.0017
Mean Temperature (°C)	17.8	19.3	18.9	21.0	0.0001
	Assemblage Structure				
	Feeding Guilds				
% Invertivores	66.5	30.5	40.6	19.2	0.0021**,‡
% Specialized Insectivores	15.1	43.0	15.7	18.5	0.03‡ n.s.
% Herbivores	2.8	7.5	6.4	4.5	0.76
% Omnivores	4.0	13.7	32.5	47.5	0.0079
% Piscivores	4.1	3.4	1.2	3.9	0.8
	Reproductive Guilds				
% Pelagophils	12.4	5.6	19.7	22.5	0.093
% Lithophilic Broadcast Spawners	0.5	3.6	2.2	3.2	0.28
% Lithophilic Brood Hiders	28.9	58.7	63.4	51.5	0.01‡ n.s.
% Lithophilic Nest Spawners	54.9	28.5	10.6	2.8	0.0018‡**
% Polyphilic Nest Spawners	0.1	1.0	2.6	10.8	0.071‡ n.s.
	Range and Distribution				
% Cosmopolitan†	4.8	17.5	42.7	65.6	0.0001‡**
% Highland Endemics†	62.8	55.3	46.3	18.8	0.01‡*

‡Bonferonni adjustment used due to high Pearson's r correlation with another tested variate.

n.s. Not statistically significant after Bonferonni adjustment

* Equivalent to $p < 0.05$ after Bonferonni adjustment; ** equivalent to $p < 0.01$ after adjustment; **** equivalent to $p < 0.0001$.

†Scott and Helfman 2001

bairdi) than rural ones. The rural sites were dominated by mirror (*Notropis leuciodus*) and warpaint shiners (*Luxilus coccogenis*), among others. In suburbanizing farmland streams, fantail and swannanoa darters (*Etheostoma flabellare* and *E. swannanoa*) and blacknose dace (*Rhinichthys atratulus*) were prevalent. Cyprinids (*Lepomis* and *Micropterus* spp.) dominated suburban/urban streams.

Feeding and reproductive guilds supported the pattern followed by the chemical and temperature data (Table 5.2). Feeding guilds yielded two statistically significant relationships, according to the nested ANOVA statistical design. Invertivores were more dominant in less developed drainages (forested and suburbanizing farmland) than in their developed counterparts (rural and suburban/urban, respectively). Invertivore dominance was also more common in forested (forested and rural) streams than in less forested (suburbanizing farmland and suburban/urban) stream types. Invertivores were more dominant in streams of forested watersheds than in any other stream type. Omnivory followed the opposite pattern; it was generally more common in the suburbanizing farmland and suburban/urban watersheds than in the other two. Rural streams had more omnivores than forested ones, and suburban/urban watersheds were more dominated by omnivores than suburbanizing farmland streams. Also, omnivore dominance increased monotonically from forested to rural to suburbanizing farmland to suburban/urban watershed types. The mean percentage of omnivores in suburban and urban assemblages was around 40% compared to 9% among the forested and rural ones.

Reproductive guilds provided similarly mixed results. Percentages of lithophilic nest spawners varied in a statistically significant way according to the nested ANOVA. Lithophilic nest spawning taxa had higher proportions among the less developed sites within forested and less forested watersheds. They were also more prevalent in streams draining forested watersheds, relative to less forested ones (t-test; $p = 0.001$). Proportions of pelagophils, lithophilic broadcast spawners, and lithophilic brood hiders appeared to differ within forested sites but not within the less forested ones. Pelagophils were more dominant at the forested sites than the rural ones. Lithophilic broadcast spawners and lithophilic brood hiders each had higher proportions in rural than in forested watersheds. Polyphillic nest spanners had higher proportions among sites in less forested watersheds and smallest proportions in the more disturbed settings. They appeared to increase monotonically from forested to rural to suburban to urban watersheds (Table 5.2).

Distributional affinities appeared to be closely related to the watershed categories analyzed here. Cosmopolitan taxa were more common among more disturbed sites in both forested and in less forested watersheds (Table 5.2). These taxa were also more common in the less forested watersheds than in the forested ones, overall. The inverse trend for highland endemics was also supported. They monotonically decreased as a percentage of all fishes collected from forested to rural to suburban to urban watersheds. These two groups of fishes are not all-inclusive. On average, they comprise about 76% of the fishes at a given site, but that figure varies considerably. At Avery Creek, only 33% of individuals were highland endemics and 0% were cosmopolitan species. The largest percentage of highland endemics at a given site were present at Wayah Creek (82%); cosmopolitan individuals were about 5% of the total catch. The largest percentage of cosmopolitan fishes was found at Haw Creek (87%). Here, 13% of individuals were highland endemics, so these two groups did in fact comprise the entire assemblage of fishes collected there. Richness was lowest in the forested class of watersheds while proportions of cosmopolitans increased from forested to rural to suburbanizing farmland to suburban/urban (Figure 5.4).

Watershed Classes

On the basis of land use characteristics, k-means clustering separated all watersheds throughout the study region into eight statistically defined classes (Table 5.3). Those were reclassified into four distinct groups based on the biological and physical observations described above. Those four classes correspond to the ones analyzed above, namely (a) forested, (b) rural, (c) suburbanizing farmland, and (d) suburban/urban.

The forested watersheds comprised 27% of the total number of watersheds in the size and elevation range examined. Their watershed and stream buffer zones were almost completely forested (usually contained > 95% forested area). When they remain forested, streams of the crystalline southern Blue Ridge are typically cool, clear, low in nutrients and primary productivity, and have relatively low fish diversity (Wallace et al. 1992, Mayden 1987, Scott and Helfman 2001). The forested category should correspond to those general conditions. Roughly 1/3 of the watersheds analyzed fell into the rural category. These were predominantly forested in the 1970's and 1990's but had appreciable amounts of agricultural land use. Within 100 m of streams, agriculture comprised 20% to 40% of the total land use in watersheds classified as rural in 1993.

Suburbanizing suburban and suburban/urban categories each were characterized by more pavement and developed land area than the forested and rural categories (Table 5.3). They also had substantially more agricultural land than the latter two classes.

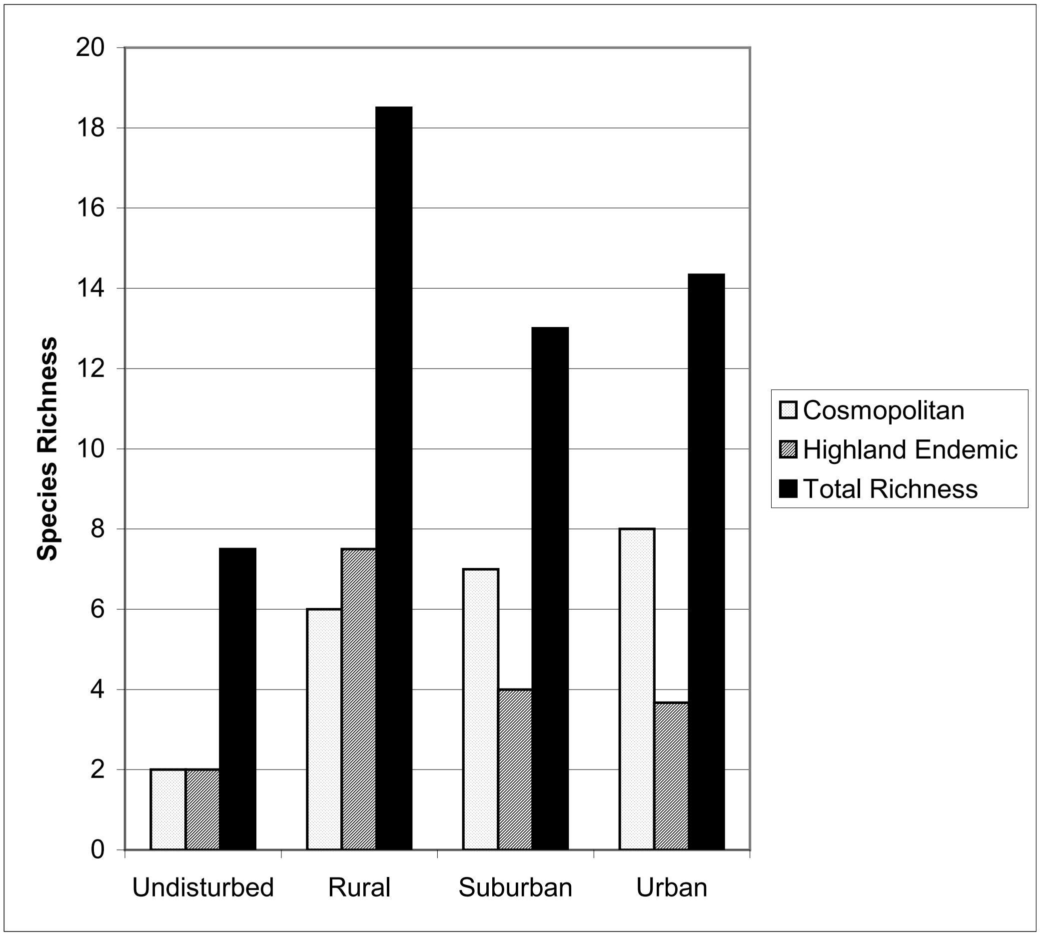

Figure 5.4. Species richness was maximal at rural sites, i.e. mostly forested watersheds with some human disturbance due to roads or housing. Cosmopolitan taxa were more common in more disturbed, urban watersheds than in undisturbed forested watersheds. Highland endemics were most diverse at rural sites and represented the smallest proportion of numbers of taxa at the suburbanizing farmland and suburan/urban sites.

Table 5.3. Mean values for land cover characteristics of eight watershed classes identified throughout the study area. The eight were combined into four categories previously identified using ordination and which were the basis of further statistical analyses. WS represents land use in the entire watershed contributing to stream channels. Buffer refers to the area within 100 m of stream channels.

			Area (%) With Given Land Cover						**Road Density ($km\ km^{-2}$)**		
Description	**Sub-heading**	**n**	**1970 Forest-WS**	**1970 Forest-Buffer**	**1993 Forest-WS**	**1993 Forest-Buffer**	**1993 Agric.-Buffer**	**1993 Agric.-WS**	**Im-proved**	**Unim-proved**	**Total**
Forested	Forested with Few Roads	8	99.4	97.6	98.8	97.3	1.2	0.8	0.10	0.065	0.10
	Forested	32	98.2	96.5	96.9	94.4	1.8	1.3	0.51	0.19	0.52
Rural	Lightly Agricultural	34	90.8	85.4	92.7	87.0	8.5	4.9	0.51	0.21	0.52
Suburban	Developed Agricultural 1	19	67.1	56.5	67.9	54.9	35.5	24.5	0.73	0.17	0.73
	Developed Agricultural 2	7	43.2	33.8	40.9	35.1	50.1	46.6	0.84	0.12	0.84
Urban	Suburban	31	80.5	72.0	80.8	68.8	22.3	13.7	0.67	0.14	0.68
	Suburban/Urban	12	70.1	60.8	72.2	63.3	12.4	8.9	1.20	0.070	1.20
	Urban	6	39.4	34.5	43.4	43.6	20.9	21.1	1.40	0.094	1.40

Suburbanizing farmland watersheds had the most agricultural land among all the groups identified. Developed land use accounted for 10-15% of suburbanizing farmland watershed areas, on average. Suburbanizing farmland watersheds comprised about 1/3 of all watersheds in the region. Suburban/urban watersheds had 10-40% of their area in developed land use and represented approximately 26 of the 149 watersheds examined (17%). Road densities were highest in urban watersheds. These watersheds clearly had the greatest impervious area among all watersheds in the study area.

In the region surrounding the focal basins (LT and Cane Creek), differences among watersheds are apparent by examining the land use map of the area with the watershed classes overlain (Figure 5.5). On average, watersheds in the LT that met the elevation and size criteria established above (Figure 5.2) were over 95% forested. In contrast, Cane Creek watersheds in the FB were only 70% forested, on average. Most of the Pigeon and Tuckaseegee River basins between 10 and 40 km^2 had drainage outlets above 720 m. The three basins in the Pigeon River that did meet size and elevation restrictions were each forested, according to the classification scheme presented. Watersheds in the Tuckaseegee basin consisted of rural and suburbanizing farmland types, with one suburban/urban watershed in the central portion of the basin.

Discussion

Rural sites had greater biological diversity than suburbanizing farmland ones. Suburbanizing farmland watersheds had greater biological integrity than suburban/urban ones. These results agree with observations from Wisconsin, where forested watersheds had higher biological integrity than agricultural watersheds (Wang et al. 1997) and where agricultural watersheds had higher biological integrity than urban watersheds (Wang et

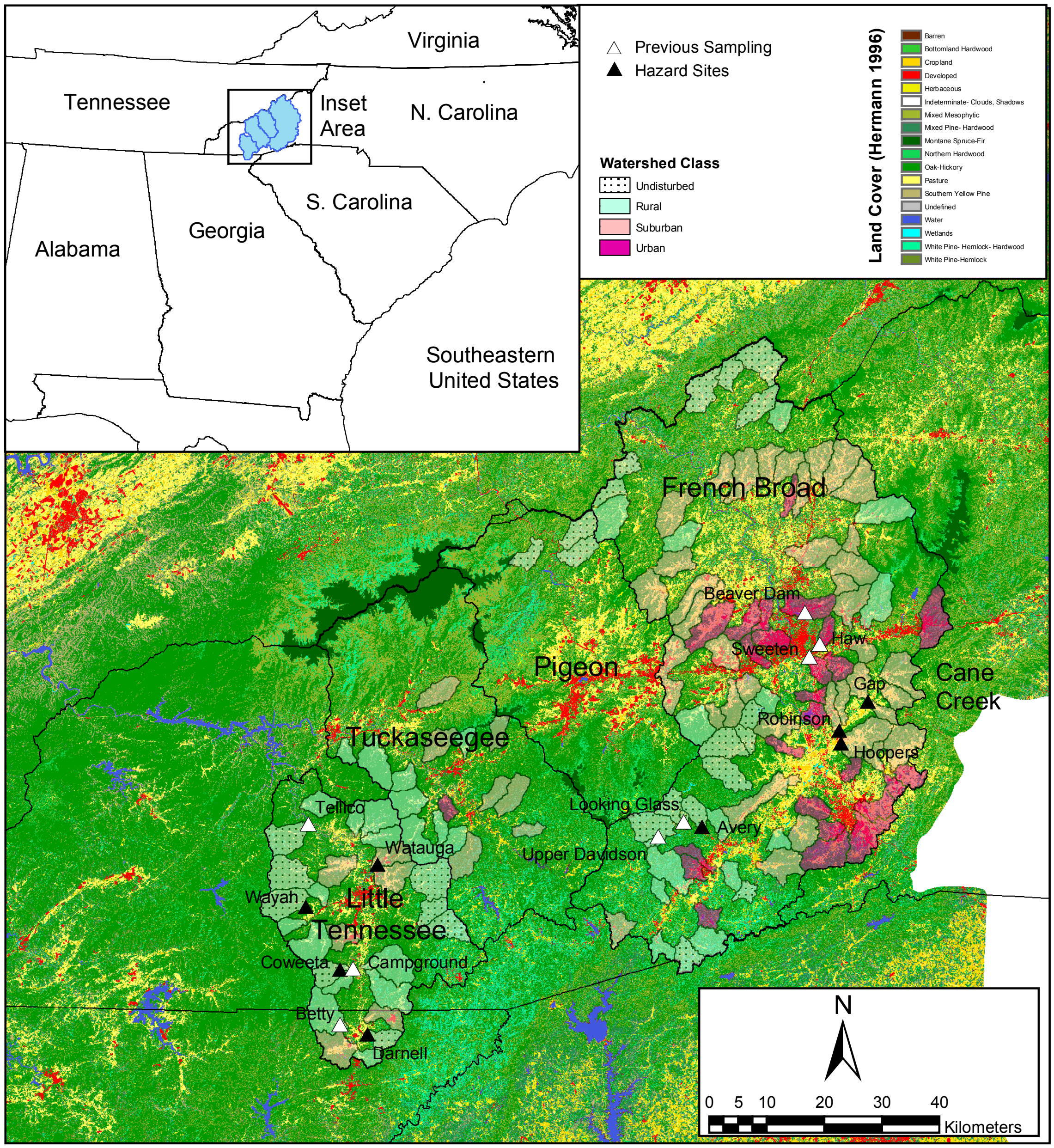

Figure 5.5. Land use data were used to classify watersheds into four land use categories corresponding to the groups identified with biological and physical ecosystem assessments.

al. 2000). Moderate development in forested watersheds will likely enhance productivity and diversity. Increased impervious coverage, however, may result in hydrologic and geomorphic changes in both rural and suburbanizing farmland streams (Wang et al. 2000, Doyle et al. 2000). Such a shift would be accompanied by increased storm flow, lower baseflow, channel widening, and perhaps smaller particle sizes in the stream bed as base flow stream power decreases and sediment supply increases. Resultant habitat homogenization (Berkman and Rabeni 1987) would decrease the availability of suitable reproductive or feeding areas for endemic taxa. Cosmopolitan taxa would then have a competitive advantage, allowing them to displace highland endemics (Scott and Helfman 2001).

In the next two decades, the forested hazard sites (Darnell and Wayah) are expected to become more like the rural sites. In 2020, they are expected to have higher diversity due to addition of highland endemic, invasive native (Scott and Helfman 2001) and invasive exogenous taxa (Rahel 2000) despite the decrease in proportions of highland endemic taxa. If development proceeds and more housing and roads are built and land use shifts toward the patterns for suburbanizing farmland watersheds, there will be a net species loss as highland endemics are lost more rapidly than invasive taxa are added (Scott and Helfman 2001). The suburbanizing farmland streams in the Cane Creek watershed are expected to move toward the faunal, physical, and chemical conditions found in the suburban/urban sites near Asheville. If this change occurs, highland endemics would comprise a minority of individuals collected, polyphilic nest spawners such as sunfish would increase, and proportions of invertivores would decrease.

Watauga, the only rural hazard site, is expected to become more like the current suburbanizing farmland sites. This watershed was classified as a suburbanizing farmland watershed in the land use classification procedure (Figure 5.5), but fish ordination (Figure 5.3) grouped it with rural watersheds. This contrast between land use and biological classifications raises important questions about the legacy of past land use (Harding et al. 1998) and the potential for ecosystem recovery (Carpenter et al. 1999). Watauga was the least forested of the rural hazard watersheds selected, and it had the second highest percentage of developed land among the eight hazard sites. It also had more agricultural land use than any of the LT, rural hazard sites. Watauga's watershed was only 76% forested in 1950. Since agricultural land use is often correlated with higher nutrient concentrations and sediment supply (Lowrance et al. 1995, Lowrance et al. 1986, Peterjohn and Correll 1984), biological integrity was expected to be lower at Watauga. Watauga has become more forested over the past 50 years, so perhaps its diverse fish assemblage reflects improvements in habitat conditions. This may be an example of reversible ecosystem change (Carpenter et al. 1999) as a formerly agricultural catchment has reverted to forested land cover. Composite hazard index predictions suggested, however, that Watauga may support new building construction more rapidly than other forested catchments.

Current land use data were extracted for 149 watersheds in the region with elevation and watershed sizes comparable to the 16 sites where detailed data were available, and those data were classified into the four groups associated with biologically distinct characteristics. Water quality observations suggest that the four categories represent a gradient of land uses with roughly monotonic physical responses from forested to rural to suburbanizing farmland to suburban/urban watersheds. The chemical differences among the four watershed categories were fairly consistent.

Although cation concentrations varied consistently among the watershed categories, fish measurements did not covary in a consistent way. Rural sites had fewer invertivores than forested ones, and urban watersheds had fewer than suburban ones. Proportions of omnivores followed the opposite trend, but not in a statistically significant way. Lithophilic nest spawners decreased, and cosmopolitan taxa increased in frequency and proportion along this putative gradient. Remaining assemblage metrics were more complex. Proportions of herbivores were least in forested and rural watersheds. Proportions of lithophilic brood hiders were also least in forested and rural watersheds. Specialized insectivores were most numerous at rural sites. Their proportions decreased with conditions typical of suburban and urban watersheds. Increased nutrient supply can increase primary production (Rosemond et al. 1993) thus supporting algivorous and omnivorous fish. Clearing riparian forests can further increase primary production in streams of this region (Webster et al. 1991). Therefore, higher proportions of specialized insectivores may reflect enhanced primary and secondary production at sites with rural watersheds.

The higher proportion of non-photosynthetic organic material at Wayah Creek could be due to deposition of leaf material or larger amounts of heterotrophic microbial biomass (bacteria and fungi). Either explanation suggests relatively less scouring in this stream because saltating sand grains should remove loose organic debris (Allan 1995). Gap Creek had much more chl-*a* than any of the other streams. This is a very open stream channel with relatively low discharge, so any nutrient subsidies would likely result in enhanced primary production. Reference streams had among the lowest AFDM values. Chlorophyll concentrations spanned the low to moderate range at Coweeta and Avery Creeks, relative to the other sites sampled. Chlorophyll-*a* concentrations at the Cane Creek sites spanned an order of magnitude. Robinson and Hoopers Creeks had very low values whereas Gap had high algal standing stocks.

Except among the suburban/urban watersheds, land use impacts were below several thresholds established in the literature in association with biological degradation of streams. Agricultural land use in the study area is usually well below the 50% threshold associated with loss of biological integrity for streams in southern Wisconsin (Wang et al. 1997). In all but the most urbanized watersheds, road density was below the 1.1 km km^{-2} density above which irreversible hydrologic changes are likely to ensue (Forman and Alexander 1998). Impervious surface area was generally below 10%, a threshold above which permanent hydrologic changes are likely to result from rapid stormflow runoff and lowered average stream depths (Booth and Jackson 1997). In contrast to most watersheds, one or more of these thresholds is exceeded in most of the suburban/urban watersheds identified using cluster analysis.

Land use change projections were partially validated since 15% of sites meeting composite hazard index criteria witnessed increased building density between 1990 and 1993. Forested and rural watersheds tend to be far from urban centers and relatively isolated from large transportation routes; these factors insulate such watersheds from rapid development (Wear and Bolstad 1998, Wear et al. 1998). Conversely, suburban and urban watersheds are subject to rapid change. The proportion of pixels where development was predicted within 100 m of streams was much greater for each of the

Cane Creek sites than for any of the LT sites (Table 5.1). Rates of land use and stream ecosystem change will likely cause watershed conditions in these sub-watersheds to continue to diverge. Rates of change will be slowest for the forested sites and most rapid for the suburbanizing farmland ones. Reference sites are expected to remain forested since public lands in this study area have had stable land use conditions through time (Turner et al. 1996). Shifts in ecosystem attributes are conceivable at reference sites if regional climate, vegetation, fish metapopulation structure, or major shifts in land use management schemes change dramatically.

Conclusion

Broad geographic observations, socioeconomic predictions, fish collections, and physical descriptions of streams were effectively synthesized through a simple classification scheme separating forested, rural, suburbanizing agricultural, and suburban/urban watersheds. Streams in this study region generally support diverse fauna. Among the watersheds where agriculture, building developments, or urban land uses occupy a substantial portion of their area, declines in biological integrity and increases in nutrient concentrations in water samples were noted. Levels of disturbance and human impacts were below several thresholds of land use associated with loss of biological integrity in other study areas, including % agricultural land (Wang et al. 1997), % impervious surface cover (Booth and Jackson 1997), and road density (Forman and Alexander 1998). The classification framework was a useful one, for it facilitates examination of key processes that are influential within one class of watersheds but that may be of less significance in another.

This research demonstrates how to use information about the trajectory of land use change to forecast likely future conditions in medium sized watersheds of the southern Blue Ridge physiographic province. Predictions of likely future land use were used to select sites for ongoing research where distinct shifts are expected in the physical structure and fish assemblages of streams. Those predictions were supported with biological and physical data. Historical land use and land cover data were used to classify all watersheds of comparable size and elevation in the region based on the knowledge gained from stream assessments. These classes provided a glimpse of overall conditions in the study area and helped explore possible future conditions in streams as they are influenced by human land use decisions. The consequences of land use for stream ecosystem change depend on the historical context of streams and their watersheds.

Acknowledgments

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Appendices

Appendix 5A. List of species collected in this study, their feeding preferences, preferred spawning substrate, and reproductive strategies. Omnivore = om; specialized insectivore = sp; invertivore = in; piscivore = ps. Remaining terms are explained in the text and in Simon 1999.

Species	Food	Substrate	Strategy
Ambloplites rupestris	ps	Polyphil[2]	Nest Spawner
Ameiurus brunneus	om	Speleophil[3]	Nest Spawner
Ameiurus nebulosus	om	Speleophil[2]	Nest Spawner
Ameiurus platycephalus	om	Speleophil[2]	Nest Spawner
Campostoma anomalum	hb	Lithophil[2]	Brood Hider
Catostomus commersoni	om	Lithopelagophil[2]	Broadcast
Clinostomus funduloides	sp	Lithophil[4]	Brood Hider
Cottus bairdi	in	Speleophil[2]	Nest Spawner
Cyprinella galactura	sp	Crevice[1]	Nest Spawner
Cyprinella monacha	sp	Crevice[5]	Nest Spawner
Cyprinus carpio	om	Phytophil[2]	Broadcast
Etheostoma blennioides gutselli	sp	Phytophil[2]	Substrate Choice
Etheostoma chlorobranchium	sp	Lithophil[1]	Nest Spawner
Etheostoma vulneratum	sp	Lithophil[3]	Nest Spawner
Etheostoma zonale	sp	Phytophil[2]	Substrate Choice
Hybrid Cyprinid	sp	Lithophil[3]	Brood Hider
Hypentelium nigricans	in	Lithophil[2]	Broadcast
Ichthyomyzon greeleyi	om	Lithophil[3]	Brood Hider
Ictalurus punctatus	om	Speleophl[2]	Nest Spawner
Lepomis auritus	in	Polyphil[1]	Nest Spawner
Lepomis cyanellus	in	Polyphil[2]	Nest Spawner
Lepomis gulosus	in	Lithophil[2]	Nest Spawner
Lepomis macrochirus	in	Polyphil[2]	Nest Spawner
Luxilus chrysocephalus	om	Lithophil[2]	Brood Hider
Luxilus coccogenis	sp	Lithophil[2]	Brood Hider
Micropterus dolomieu	ps	Polyphil[2]	Nest Spawner
Micropterus salmoides	ps	Polyphil[2]	Nest Spawner
Moxostoma (undescribed)	in	Lithophil[2]	Broadcast
Moxostoma anisurum	in	Lithophil[2]	Broadcast
Moxostoma carinatum	in	Lithophil[2]	Broadcast
Moxostoma duquesnei	in	Lithophil[2]	Broadcast
Moxostoma erythrurum	in	Lithophil[2]	Broadcast
Moxostoma macrolepidotum	in	Lithophil[2]	Broadcast
Nocomis micropogon	om	Lithophil[2]	Brood Hider
Notemigonus crysoleucas	om	Phytophil[2]	Broadcast
Notropis leuciodus	sp	Lithophil[1]	Brood Hider
Notropis lutipinnis	sp	Lithophil[3]	Brood Hider
Notropis photogenis	sp	Lithophil[2]	Brood Hider
Notropis rubellus	sp	Lithophil[2]	Brood Hider
Notropis spectrunculus	sp	Lithophil[3]	Brood Hider
Notropis telescopus	om	Pelagophil[1]	Broadcast

Oncorhynchus mykiss	in	Lithophil[2]	Brood Hider
Perca flavescens	ps	Phytophil[2]	Broadcast
Percina aurantiaca	sp	Lithophil[3]	Brood Hider
Percina evides	sp	Lithophil[2]	Brood Hider
Percina squamata	sp	Lithophil[3]	Brood Hider
Phenacobius crassilabrum	sp	Lithophil[4]	Brood Hider
Pomoxis nigromaculatus	ps	Phytophil[2]	Nest Spawner
Pylodictis olivaris	ps	Speleophil[2]	Nest Spawner
Rhinichthys atratulus	om	Lithopelagophil[2]	Broadcast
Rhinichthys cataractae	sp	Lithopelagophil[2]	Broadcast
Salmo trutta	ps	Lithophil[2]	Brood Hider
Salvelinus fontinalis	in	Lithophil[2]	Brood Hider
Semotilus atromaculatus	om	Lithophil[2]	Brood Hider

Data sources: [1]Jenkins and Burkhead (1994); [2]Simon (1999); [3]congener with presumably similar reproductive habits; [4]professional judgement was used based on field observations of where spawning is thought to occur; [5]McLarney (1989, 1990)

Appendix 5B. Hazard site descriptions based on digital line graphs from 1:24,000 scale topographic maps, watershed boundaries (Chapter 2), 30-m USGS Level 1 digital elevation models, and land cover from 1950, 1970 (Wear and Bolstad 1998) and 1993 (Hermann 1996).

Site	Name	X-coord	Y-coord	Elev	% Forest 1993	% Developed 1993	% Ag 1993	% Forest 1970	% Forest 1950	sq. km
1	Coweeta	278818	3882457	677	98.5	0.0	1.2	99.5	98.4	16.6
2	Darnell	283588	3871167	660	98.4	0.0	1.5	98.9	97.4	14.7
3	Wayah	272822	3893401	664	97.5	0.1	2.2	98.2	98.0	34.9
4	Watauga	285390	3900773	618	84.9	5.3	8.8	84.0	76.4	16.8
5	Avery	341921	3907410	700	99.5	0.0	0.4	99.1	100.0	15.8
6	Gap	370762	3929143	661	71.5	0.9	27.5	70.3	64.3	20.3
7	Robinson	365782	3924142	648	65.5	6.2	27.9	64.4	63.9	14.8
8	Hoopers	366047	3921937	645	71.6	0.6	27.6	72.7	67.5	39.4

Appendix 5C. Hazard sites were visited on two different dates. This table provides sample dates and unique visit identifiers used in subsequent tables to record the date and location of a sample collected in the field.

Visit	Site	Date
16	8	07/19/2000
15	7	07/19/2000
14	6	07/19/2000
13	5	07/19/2000
12	4	07/19/2000
11	3	07/19/2000
10	2	07/19/2000
9	1	07/19/2000
8	8	07/12/2000
7	7	07/11/2000
6	6	07/11/2000
5	5	07/10/2000
4	4	07/06/2000
3	3	07/07/2000
2	2	07/05/2000
1	1	06/30/2000

Appendix 5D. Sediment cores from each site were separated into percentage of fine (< 2 mm) and coarse (> 2 mm) material by weight (refer to Appendix 5C).

Visit	% Fines	% Coarse
1	3.3	96.7
2	9.6	90.4
3	10	90
4	15.5	84.5
5	13.7	86.3
6	13.1	86.9
7	18.4	81.6
8	36.1	63.9

Appendix 5E. Sediment medial axes were measured on 100 bed particles selected by traversing each stream in a zig-zag pattern (refer to Appendix 5C).

Visit	Mu (mm)	Mu (phi)
1	64.40217	-4.6126
2	74.2449	-4.3602
3	81.6413	-5.29565
4	42.37895	-3.80421
5	83.3125	-5.48333
6	114.6392	-5.14639
7	21.17273	-2.92727
8	10.51456	-2.5699

Appendix 5F. Water chemistry data for each site (use Appendix 5C).

Visit	NO3	NH4	PO4	K	Na	Ca	Mg	SO4
9	0.043	0.0133	0.023	0.5711	1.7974	0.9881	0.373	0.558
9	0.042	0.0122	0.023	0.6932	1.8924	1.0438	0.4072	0.548
9	0.044	0.0191	0.047	0.6729	1.8824	1.0326	0.3939	0.624
10	0.03	0.0103	0.008	0.5738	1.7979	1.0178	0.3347	0.817
10	0.034	0.0138	0.024	0.5612	1.8369	1.0153	0.3433	0.795
10	0.039	0.0126	0.021	0.5344	1.8062	1.005	0.3332	0.803
11	0.052	0.0137	0.03	0.6471	2.2751	1.2943	0.7224	1.075
11	0.057	0.0164	0.02	0.6545	2.3032	1.2855	0.7235	1.101
11	0.051	0.0141	0.033	0.6275	2.2502	1.2475	0.7053	1.098
12	0.199	0.0508	0.006	1.8056	3.153	2.846	1.3348	3.82
12	0.199	0.0565	0.035	2.0523	3.3077	3.1885	1.402	3.751
12	0.189	0.0537	0.045	2.1045	3.7526	3.0606	1.3731	3.693
13	0.054	0.0175	0.012	0.6734	1.6635	1.1388	0.4421	1.513
13	0.067	0.0192	0.01	0.7049	1.7789	1.1804	0.4512	1.599
13	0.05	0.0136	0.026	0.6565	1.6455	1.1021	0.4267	1.516
14	0.233	0.0133	0.003	1.9294	5.6041	5.2306	1.7018	2.915
14	0.235	0.0061	0.011	1.9132	5.6413	5.1647	1.69	2.905
14	0.227	0.0145	0.015	1.9963	5.6909	5.2379	1.6934	2.9
15	0.45	0.0153	0.032	1.5493	4.062	6.0341	2.2568	2.207
15	0.446	0.0329	0.06	1.6916	4.2991	6.1002	2.2673	2.234
15	0.444	0.0244	0.023	1.6929	4.2996	6.0155	2.2612	2.247
16	0.336	0.0314	0.004	1.8897	4.672	4.9193	1.3715	3.145
16	0.334	0.0336	0.033	1.9141	4.6938	4.8859	1.3386	3.182
16	0.336	0.0317	0.036	1.9483	4.9613	4.8532	1.3286	3.335

Appendix 5G. Total suspended solids samples (s1, s2, s3), averages (TSSmu), and standard deviations (TSSstd) at each site (refer to Appendix 5C).

Visit	TSS	mg/l
1	s1	8
1	s2	9.6
1	s3	0.8
1	TSSmu	6.1
1	TSSstd	4.7
2	s1	0
2	s2	3.2
2	s3	40.8
2	TSSmu	14.7
2	TSSstd	22.7
3	s1	16
3	s2	6.4
3	s3	17.6
3	TSSmu	13.3
3	TSSstd	6.1
4	s1	132
4	s2	146.4
4	s3	118.4
4	TSSmu	132.3
4	TSSstd	14
5	s1	4
5	s2	2.4
5	s3	3.2
5	TSSmu	3.2
5	TSSstd	0.8
6	s1	12
6	s2	17.6
6	s3	14.4
6	TSSmu	14.7
6	TSSstd	2.8
7	s1	6.4
7	s2	6.4
7	s3	11.2
7	TSSmu	8
7	TSSstd	2.8
8	s1	10.4
8	s2	24
8	s3	23.2
8	TSSmu	19.2
8	TSSstd	7.6

Appendix 5H. Mean ash-free dry mass from three samples at each site (refer to Appendix 5C).

Visit	mg/m2	s.d.
1	0.847	1.112
2	1.067	1.34
3	4.766	10.989
4	1.733	0.839
5	0.877	0.289
6	2.189	0.895
7	0.796	0.839
8	1.176	0.983

Appendix 5I. Chlorophyll-*a* from three bulked samples from dominant substrate at three transects at each stream (refer to Appendix 5C).

Visit	Chl-a (μg/m2)	s.d.
1	35.05	21.2
3	115.69	112.35
4	89.07	42.8
2	109.05	119.69
5	60.39	47.35
6	299.64	226.93
7	42.42	29.56
8	48.72	18.13

Appendix 5J. Mean temperature at each site for August 3- September 15, 2000.

Site	Temperature (C)
1	17.2
2	17.3
3	18.5
4	19.4
5	18.0
6	18.6
7	18.8
8	19.2

Appendix 5K. Fish species codes and names.

sp	Common	Latin
3	Mountain Brook Lamprey	Ichthyomyzon greeleyi
22	Gizzard Shad	Dorosoma cepadianum
24	Central Stoneroller	Campostoma anomalum
26	Rosyside Dace	Clinostomus funduloides
29	Whitetail Shiner	Cyprinella galactura
31	Spotfin Chub	Erimonax monachus
35	Common Carp	Cyprinus carpio
44	Warpaint Shiner	Luxilus coccogenis
53	River Chub	Nocomis micropogon
54	Golden Shiner	Notemigonus crysoleucas
60	Emerald Shiner	Notropis atherinoides
43	Striped Shiner	Luxilus chrysocephalus
65	Tennessee Shiner	Notropis leuciodus
66	Yellowfin Shiner	Notropis lutipinnis
67	Silver Shiner	Notropis photogenis
68	Rosyface Shiner	Notropis rubellus
69	Saffron darter	Notropis rubicroceus
71	Mirror Shiner	Notropis spectrunculus
73	Telescope Shiner	Notropis telescopus
77	Hybrid Shiner	Hybrid Notropis
79	Fatlips Minnow	Phenacobius crassilabrum
87	Blacknose Dace	Rhinichthys atratulus
88	Longnose Dace	Rhinichthys cataractae
89	Creek Chub	Semotilus atromaculatus
90	River Carpsucker	Carpiodes carpio
93	White Sucker	Catostomus commersoni
98	Northern Hog Sucker	Hypentelium nigricans
103	Silver Redhorse	Moxostoma anisurum
104	River Redhorse	Moxostoma carinatum
105	Black Redhorse	Moxostoma duquesnei
106	Golden Redhorse	Moxostoma erythrurum
107	Shorthead Redhorse	Moxostoma macrolepidotum
108	Redhorse (undescribed)	Moxostoma (undescribed)
134	Rainbow Trout	Oncorhynchus mykiss
135	Brown Trout	Salmo trutta
136	Brook Trout	Salvelinus fontinalis
131	Muskellunge	Esox masquinongy
500	Eastern Mosquitofish	Gambusia holbrooki
112	Brown Bullhead	Ameiurus nebulosus
113	Flat Bullhead	Ameiurus platycephalus
116	Channel Catfish	Ictalurus punctatus
122	Stonecat	Noturus flavus
128	Flathead Catfish	Pylodictis olivaris

152 Mottled Sculpin	Cottus bairdi
156 White Bass	Morone chrysops
161 Rock Bass	Ambloplites rupestris
165 Redbreast Sunfish	Lepomis auritus
166 Green Sunfish	Lepomis cyanellus
168 Warmouth	Lepomis gulosus
170 Bluegill	Lepomis macrochirus
178 Smallmouth Bass	Micropterus dolomieu
180 Largemouth Bass	Micropterus salmoides
181 White Crappie	Pomoxis annularis
182 Black Crappie	Pomoxis nigromaculatus
300 Tuckaseigee Darter	Etheostoma blennioides gutselli
191 Greenside Darter	Etheostoma blennioides
195 Greenfin Darter	Etheostoma chlorobranchium
200 Fantail darter	Etheostoma flabellare
214 Redline Darter	Etheostoma rufilineatum
221 Swannanoa darter	Etheostoma swannanoa
224 Wounded Darter	Etheostoma vulneratum
226 Banded Darter	Etheostoma zonale
231 Yellow Perch	Percaflavescens
232 Tangerine Darter	Percina aurantiaca
236 Gilt Darter	Percina evides
242 Olive Darter	Percina squamata
247 Walleye	Stizostedion vitreum
400 Hybrid Cyprinid	Hybrid Cyprinid
301 Sicklefin Redhorse	Moxostoma sp. cf. macrolepidotum
175 Hybrid sunfish	Hybrid Lepomis sp.
19 Alabama Shad	Alosa alabamae
114 Snail Bullhead	Ameiurus brunneus
153 Banded Sculpin	Cottus carlinae
230 Hybrid darter (Etheostoma)	Hybrid Etheostoma

Appendix 5L. Fish collections data. Visit corresponds to Appendix 5C; sp corresponds to Appendix 5K. Freq refers to the number of individuals collected.

Visit	sp	freq
1	24	1
1	26	14
1	53	2
1	66	1
1	88	17
1	98	1
1	134	3
1	152	77
1	161	4
2	24	62
2	53	2
2	66	1
2	88	4
2	134	15
2	152	223
3	24	11
3	26	14
3	53	3
3	71	1
3	87	8
3	88	21
3	98	2
3	134	2
3	135	1
3	152	205
4	24	15
4	29	9
	31	1
4		
4	44	12
4	53	4
4	65	19
4	66	4
4	79	1
4	87	1
4	98	1
4	152	95
4	161	1
4	165	1
4	236	14
5	87	8
5	88	5
5	134	15

Visit	sp	freq
5	135	5
5	152	51
5	200	3
6	24	116
6	44	6
6	53	21
6	69	77
6	87	61
6	89	16
6	98	7
6	152	80
6	161	2
6	166	1
6	200	36
6	221	38
6	236	1
7	3	5
7	24	3
7	44	13
7	53	10
7	69	50
7	73	2
7	87	23
7	89	28
7	93	3
7	98	6
7	152	13
7	161	3
7	165	4
7	170	5
7	200	7
8	24	2
8	69	49
8	73	1
8	87	46
8	89	11
8	93	2
8	98	3
8	152	13
8	161	3
8	200	13
8	221	13

CHAPTER 6

CONCLUSION

This research demonstrates that GIS and related technologies, such as spatial databases, remote sensing, and simulation modeling, complement stream research based on sampling habitat characteristics, water quality, and biota. Digital spatial data and analysis techniques provide access to information about land use and terrain at many scales, and *in situ* sampling provides detailed information about patterns and processes in aquatic ecosystems. Spatial data and analysis methods used here were tailored to watershed-based research in the Blue Ridge physiographic province of western North Carolina, U.S.A.; watershed analyses contrast with analyzing patterns and processes at specific sites. Three themes integrate the research presented: (a) the importance of understanding accuracy limitations of spatial data; (b) simulation modeling is a powerful means of analyzing stream ecosystem properties because it integrates spatial patterns and landscape processes; and (c) watershed classification is a useful tool for geographical and ecological analyses of watersheds.

Accuracy limitations of digital watershed and simulated sediment yield data were quantified in Chapters 2 and 3. In Chapter 2, watershed boundaries extracted via automated delineation techniques were compared to boundaries that were hand delineated. Systematic errors were observed for all sizes of watersheds, and a minimum size of 250 ha was deemed necessary to minimize errors in spatial analyses that use automatically extracted watershed boundaries. The average distance between automatically extracted and hand-digitized watershed boundaries was 17 m, and 95% of points on automatically derived watershed boundaries were within 50 m of the hand-delineated ones. These errors are non-negligible for studies of small catchments, for example for first-order streams. In Chapter 3, varying data resolution was shown to strongly influence a simulation model designed to examine watershed sediment yield using the Revised Universal Soil Loss Equation (RUSLE). When the resolution of input data was coarsened beyond 90 m, model results were not strongly related to model results from 30-m inputs. The map scale of data used also influenced model results; low resolution soil data mapped at 1:250,000 scale led to higher sediment yield model predictions than high resolution soil data mapped at scales from1:12,000 to 1:63,000. When data resolution was coarsened for land cover data, model results were not strongly related to model results from the original 30 m data.

Chapter 3 demonstrated that simulation can be effective in estimating baseflow annual sediment yield. Sediment is a prevalent impairment to water bodies throughout the United States (US EPA 1997) that interferes with metabolism, feeding, and reproduction among many taxa, including invertebrates and fishes (Waters 1995). Because sediment is so prevalent and deleterious, simulation modeling can serve an important function in identifying watersheds with greater and lessor degrees of impairment. In the Upper Little Tennessee River basin, unimproved road density proved to be strongly related to model estimates of sediment yield, and thus it may be a useful indicator of hillslope-derived sediment in largely forested watersheds. In Chapter 4, proportions of smoky sculpin (*Cottus bairdi* ssp.) decreased with sediment yield rate (tonne * km^{-2} * yr^{-1}), as did the proportion of highland endemics, a group of species that have been suggested are especially vulnerable to anthropogenic stress due to their evolutionary adaptation to streams with cool temperatures, low dissolved solids concentrations, and coarse, rocky substrates (Mayden 1987, Scott and Helfman 2001). Another group of native taxa, called cosmopolitans, are widely distributed and can tolerate higher temperatures, nutrient loads, and sediment loads (Scott and Helfman 2001); among large watersheds, cosmopolitan proportions were highest where model predictions of sediment loading rate were also highest.

Classification proved to be useful in several contexts. In Chapter 2, attributes of watersheds extracted from digital elevation model data and digitized streams were condensed to four orthogonal principal components that suggested four distinct classes of watersheds and streams. Those classes included: (a) large streams and rivers with well-developed alluvial floodplains; (b) high gradient streams with relatively high relief and steep hill slopes; (c) moderate-sized, upper elevation watersheds with low gradients; and finally (d) large, sinuous tributaries to the main stem of rivers.

In Chapter 4, a more detailed analysis within the Upper Little Tennessee River basin revealed that physiographic and land use properties covary in predictable ways since agricultural, urban, and commercial activities are concentrated in the valley containing the main stem of the river. Statistical analyses of fish assemblages suggested distinct biological patterns between watershed classes. Fish assemblages in large watersheds (> 39 km^2) and small watersheds (< 39 km^2) were distinct. Among small watersheds, ordination revealed two distinct classes of watersheds based on data from fish collections. One group included sites with greater numbers of highland endemics. A second group of sites was typified by lower elevations, somewhat larger watershed areas, and greater proportions of cosmopolitan taxa. This separation of highland and lowland groups of fishes was not apparent in the streams with larger catchments. That lack of separation supported a current hypothesis that anthropogenic disturbance facilitates dispersal of widespread taxa that can tolerate warmer temperatures, higher nutrient loads, and more sediment (Scott and Helfman 2001).

In Chapter 5, classification provided insight into current land use conditions across a large area and enabled forecasts of future stream ecosystem conditions. This work demonstrated the importance of long term monitoring to analyze the legacy of past land use (Harding et al. 1998) as well as to predict geomorphic, chemical, and biological changes to streams in response to anthropogenic land use change. Predicted increases in building density were mapped using forecast projections (Wear and Bolstad 1998), and eight sites were selected where geomorphic, water quality, algae, invertebrate, and fish sampling are to continue over the next 20 years. Data from eight additional sites were combined with data from the eight new sites. Fish assemblages were used to identify four watershed categories: “forested”, “rural”, “suburbanizing farmland”, and “suburban/urban”. Stream chemistry, temperature, and fish assemblages were statistically different among watershed categories. If watersheds undergo new building and road construction, streams are likely to become warmer, higher in nutrients, and to support more omnivores and cosmopolitan taxa. Biological integrity appeared to be highest in rural catchments and lowest in urban ones. The 149 watersheds throughout the study area of comparable size and elevation were classified into four classes corresponding to the forested, rural, suburbanizing farmland, and suburban/urban categories identified previously. The database of watersheds was therefore used to extrapolate insights from detailed analyses to watersheds throughout the larger study region.

Technological solutions were tailored to stream research in a large area of western North Carolina, thus demonstrating that explicit needs for biological monitoring and watershed research can provide the impetus for innovative applications of GIS. Watershed assessments of ecological integrity underlie each of the database management, accuracy assessments, and modeling techniques advanced here. Similarly, new questions in stream ecology can be addressed by adopting GIS technologies. The availability of spatial data allows comparison of watersheds across large areas, analysis of possible long-term effects of past land use on present-day stream ecosystems, and projections of possible future conditions in streams and watersheds based on spatially-explicit modeling. Managers, scientists, and citizens will benefit if the two disciplines continue to advance in tandem to provide insight into the future condition of freshwater resources (Naiman et al. 1995).

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