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Recreational Stream Crossing Effects on Sediment Delivery and Macroinvertebrates in Southwestern Virginia, USA

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Abstract Trail-based recreation has increased over recent decades, raising the environmental management issue of soil erosion that originates from unsurfaced, recreational trail systems. Trail-based soil erosion that occurs near stream crossings represents a non-point source of pollution to streams. We modeled soil erosion rates along multiple-use (hiking, mountain biking, and horseback riding) recreational trails that approach culvert and ford stream crossings as potential sources of sediment input and evaluated whether recreational stream crossings were impacting water quality based on downstream changes in macroinvertebrate-based indices within the Poverty Creek Trail System of the George Washington and Jefferson National Forest in southwestern Virginia, USA. We found modeled soil erosion rates for non-motorized recreational approaches that were lower than published estimates for an off-road vehicle approach, bare horse trails, and bare forest operational skid trail and road approaches, but were 13 times greater than estimated rates for undisturbed forests and 2.4 times greater than a 2-year old clearcut in this region. Estimated soil erosion rates were similar to rates for skid trails and horse trails where best management practices (BMPs) had been implemented. Downstream changes in macroinvertebrate-based indices indicated water quality was lower downstream from crossings than in upstream reference reaches. Our modeled soil erosion rates illustrate recreational stream crossing approaches have the potential to deliver sediment into adjacent streams, particularly where BMPs are not being implemented or where approaches are not properly managed, and as a result can negatively impact water quality below stream crossings.

Keywords Recreation ecology · Trail erosion · Water quality · Universal Soil Loss Equation · Water Erosion Prediction Project · Soil erosion

Introduction

Trail-based recreation has experienced a recent upsurge in popularity, presenting an environmental disturbance threat to forested ecosystems (Bosworth 2007) and demanding increased management resources for trail maintenance and erosion prevention (Lynn and Brown 2003; Olive and Marion 2009). Trails in forested recreation areas provide passage to all-terrain vehicles, bike riders, cross-country skiers, hikers, horseback riders, pack animals, and snowmobilers (Deluca et al. 1998; Törn et al. 2009; Pickering et al. 2010). Recreational trails often intersect streams or rivers at water crossings which create pathways for eroded soil to enter adjacent streams and decrease water quality. Erosive disturbances that introduce sediment into streams such as traffic, crossing installation, and approach runoff that occur adjacent to and within stream crossings can alter total suspended solids, conductivity, temperature, and pH in streams, particularly if crossings are improperly designed or inadequately managed (Lane and Sheridan 2002; Aust et al. 2011; Wear et al. 2013). Increased fine sediment loads from recreational trails can also reduce fish and macroinvertebrate habitat and alter watershed

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processes downstream from trail crossings (Brown 1994; Chin et al. 2004; Arp and Simmons 2012). Increased fine sediment loads in waterways result in higher costs for downstream water treatment plants and increased frequency of dredging ditches, streams, lakes, and harbors (Clark 1985). User surveys indicate eroded trails decrease recreational opportunities and reduce the level of enjoyment experienced by recreationists, for whom these trails are ultimately managed (Manfredo et al. 1996; Lynn and Brown 2003).

Soil erosion along trails is a multiple-step process that begins with hikers and other recreational traffic trampling vegetative cover and scuffing away litter and duff layers (Manning 1979). The underlying soil is then compacted by trail traffic resulting in an impervious soil surface that is more prone to surface water runoff and creates conditions that increase the likelihood of soil erosion (Olive and Marion 2009). Although the amount and type of trail traffic influence the rate of erosion, even remote, infrequently traveled trails have measurable signs of soil erosion (Weaver and Dale 1978; Bratton et al. 1979). Type of trail usage influences soil erosion with horseback riding resulting in soil erosion rates that are two to eight times as high as rates caused solely by hikers (Wilson and Seney 1994; Deluca et al. 1998; Olive and Marion 2009). Trails with steeper grades, silty or sandy soils, and low levels of organic matter also result in erosion-prone conditions (Olive and Marion 2009). Another factor that influences erosion risk is the timing of trail use, with trail use during wet and muddy conditions and heavy rain events resulting in trail widening and higher suspended sediment loads downstream from trail stream crossings (Bayfield 1973; Ayala et al. 2005). An initial step for land managers is to identify existing stream morphologies, trail conditions, and trail use patterns with the highest soil erosion potentials and then to balance soil erosion reducing best management practices (BMPs) with maintaining access and enjoyment for recreational trail users.

Previous research on recreational trail erosion has primarily focused on identifying underlying factors such site condition, user type, traffic frequency, user behavior, and trail design with the majority of research conducted in areas other than areas adjacent to stream crossings (Bratton et al. 1979; Wilson and Seney 1994; Deluca et al. 1998; Olive and Marion 2009; Törn et al. 2009; Pickering et al. 2010). Sedimentation has been visually observed in streams adjacent to stream crossing approaches used by motorized and non-motorized traffic (Olive and Marion 2009; Wilkerson and Whitman 2009), and deposition rates have been quantified in streambeds below off-road vehicle ford crossings (Brown 1994). Few studies have evaluated soil erosion rates for approaches to recreational stream crossings (Ayala et al. 2005). Negative downstream impacts on water quality have been demonstrated below off-road vehicle trail and operational forest skid trail and road stream crossings (Chin et al. 2004; Ayala et al. 2005; Neal et al. 2007; Aust et al. 2011; Wear et al. 2013). Two commonly used stream crossing structures include culverts (pipe structures that allow for the diversion and passage of water underneath trails) and fords (low water crossings that allow for traffic to pass directly on top of streambed or on supportive material such as geotextiles or hardened bottoms that allow passage of fish and other biota to upstream) (Virginia Department of Forestry 2009). Research evaluating downstream changes in water quality below operational forest skid trail and road crossings has indicated that crossing type can influence the degree to which water quality is impacted (Aust et al. 2011).

Additional research to evaluate impacts of recreational approaches and associated sediment delivery potentials into adjacent stream crossings and to quantify consequential downstream impacts on water quality is needed to better understand the implications and magnitude of trail erosion at stream crossings. Therefore, the objectives of this study were to: (1) use soil erosion models to provide soil erosion rate estimates for recreational trail approaches to selected stream crossings as potential sources of sediment inputs to the Poverty Creek watershed, a 2,145 ha watershed located in the Southern Appalachian Mountains, (2) validate sediment yield estimates using macroinvertebrate sampling as biological indication of altered water quality below stream crossings, and (3) discern whether water quality impacts vary between culvert and ford stream crossing types.

Methods

Study Area

This study was conducted along the recreational trails in the Poverty Creek Trail System (Pandapas Pond Recreation Area) located within the George Washington and Jefferson National Forest in Montgomery County, Virginia (Fig. 1). The trail system is maintained by the Eastern Divide Ranger District of the US Forest Service and is contained within the 2,145 ha Poverty Creek watershed (USDA 1972). The Poverty Creek watershed, which is a tributary to the New River, contains approximately 7.6 km of perennial streams and 47.5 km of intermittent streams which typically carry flow from late fall through late spring/early summer. This recreational area contains 27.4 km of unsurfaced multiple-use, single-tracked trails that receive year-round, non-motorized traffic from hikers, mountain bikers, and horseback riders. Although the trail system is open year-round, user traffic is seasonal, with less

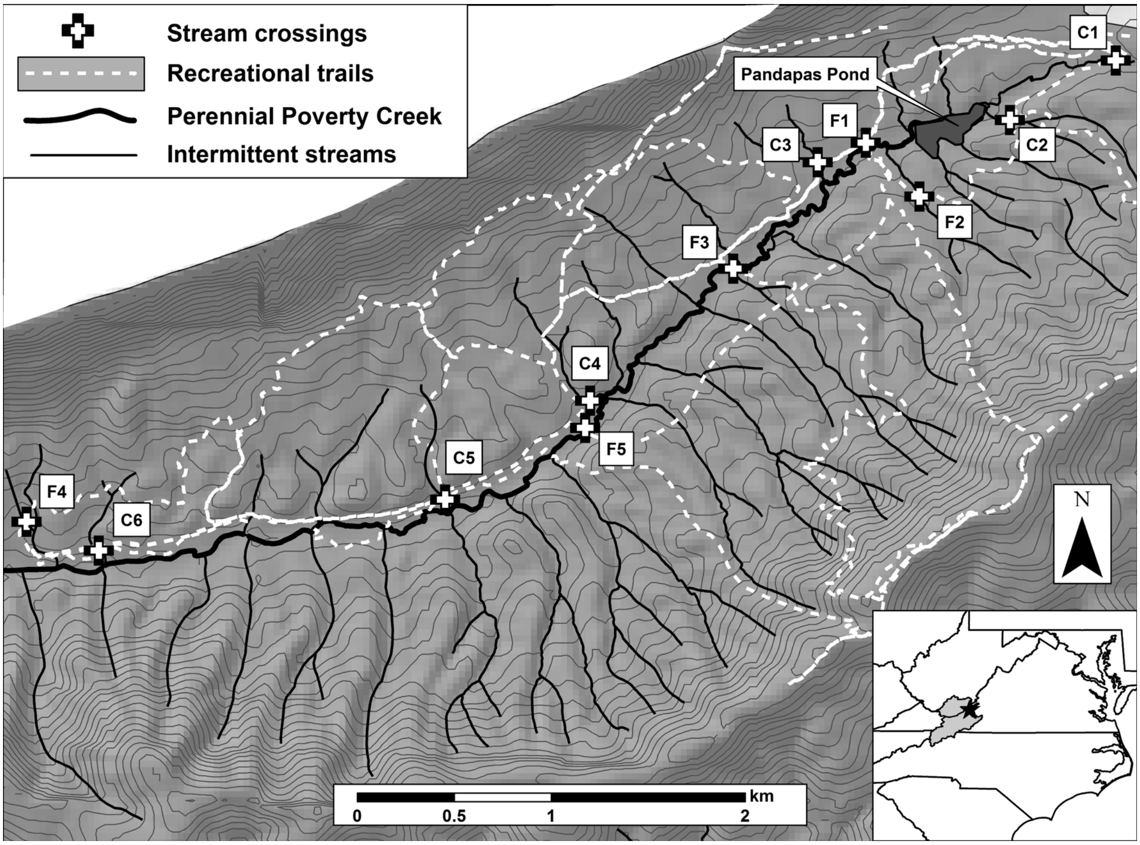

Fig. 1 Recreational stream crossings located throughout the Poverty Creek trail system in the Washington and Jefferson National Forest in Southwestern Virginia. Crossings are labeled as C (culvert) and F (ford) followed by the respective crossing number. Contour lines represent 10 m of change in elevation. *Shaded areas* in *inset* illustrate the upper and middle New River sub-basins

trail use occurring during winter months and the heaviest traffic occurring in the months of May and October (Kratzer 1993). Construction of recreational trails first began in 1976. However, all trails used in this study were constructed after 1991, when the Pandapas Pond Recreation Area expanded recreational opportunities (Kratzer 1993). Many of the recreational trails originate from the Pandapas Pond Day-Use Area, which includes picnic areas, two parking lots, and a 3.3 ha pond that was constructed in 1951 and is stocked for fishing. Several first- and second-order intermittent streams supply water to the pond. Poverty Creek Trail is the longest trail within the Poverty Creek Trail System. This trail meanders along the floodplain of Poverty Creek, a first-order perennial stream originating at Pandapas Pond, and crosses the stream in multiple locations. Several additional trails merge with the Poverty Creek Trail and lead to the ridges of Gap and Brush Mountains after crossing several first- and second-order intermittent streams. In total, there are approximately 47 recreational trail stream crossings in the Poverty Creek watershed.

Mean annual precipitation in the study area is 1,039 mm (NOAA 2011). Average January temperature is 5.6 °C, and average July temperature is 24.3 °C (NOAA 2011). Dominant soils within the trail system include Craigsville series cobbly sandy loam (loamy-skeletal, mixed, superactive mesic Fluventic Dystrudepts), Jefferson series gravelly silt loam (find-loamy, siliceous, semiactive, mesic Typic Hapludults), and Berks-Weikert association composed of Berks channery loam (loamy-skeletal, mixed, active, mesic Typic Dystrudepts) and Weikert channery silt loam (loamy-skeletal, mixed, active, mesic Lithic Dystrudepts) (USDA and NRCS 2013).

Field Methods

We selected 11 recreational stream crossings from the 47 total crossings present within the Poverty Creek Trail System so that there was a representation of culvert ($N = 6$) and ford ($N = 5$) crossing structure types (Fig. 1). Three of the selected ford crossings (F1, F3, and F5) were located on the first-order perennial Poverty Creek with the

Table 1 Stream and trail width characteristics at selected culvert (C) and ford (F) stream crossings

Crossing	Stream class and order	Streambed substrates[a]		Channel slope (%)	Channel width (m)	Trail width (m)
		Upstream	Downstream			
C1	I (1)	Cobble	Cobble	1.0	3.7	3.6
C2	I (2)	Cobble, gravel, sand	Gravel, sand, silt	3.0	1.8	3.4
C3	I (2)	Cobble, gravel, sand	Gravel, sand, silt	4.0	1.7	3.4
C4	I (1)	Cobble, gravel	Sand, silt	4.0	4.8	1.7
C5	I (1)	Cobble, gravel	Cobble, gravel	4.0	2.6	2.7
C6	I (2)	Cobble, gravel	Cobble, gravel	8.0	3.7	0.9
F1	P (1)	Boulders, cobble, gravel	Boulders, cobble, gravel, sand	1.0	4.4	2.4
F2	I (2)	Cobble, gravel	Cobble, gravel	3.0	2.1	1.1
F3	P (1)	Boulders, cobble, gravel	Gravel, sand, silt	1.0	6.9	3.4
F4	I (1)	Cobble	Cobble	4.0	1.5	1.5
F5	P (1)	Boulders, cobble, gravel	Boulders, cobble, gravel, sand	0.0	8.2	1.4

Streams crossed by recreational trails are abbreviated as I (intermittent) and P (perennial)

[a] Based on visual observations of pebble size classes (Wolman 1954)

remaining crossings situated on first- and second-order intermittent streams (Table 1). A distance of 1.2 km occurred between the F1 and F3 crossings, while the F3 and F5 were 1.3 km apart (Fig. 1). All stream crossings occurred in areas characterized by forested overstory with rhododendron frequently present in the understory along stream banks, typical of mountainous streams in the Southern Appalachian Mountains. Trails crossing streams were all single-tracked and designated for multiple-use recreational activities including horseback riding, mountain biking, and hiking. Average trail widths ranged from 0.9 to 3.6 m for trail segments approaching stream crossings (Table 1).

At each stream crossing, slope-length was measured as the distance along each approach that runoff would directly flow into the adjacent stream crossing without being diverted or stored (Dissmeyer and Foster 1984). Slope percent was measured along the approach length using a clinometer, and slope shape was classified as linear, concave, convex, or s-shaped. Percent bare soil was observed along each trail approach at 100 points on random transects, and percent canopy cover was estimated with a spherical densiometer above points with bare soil. The percentage of slope-length in steps, covered with invading vegetation, and containing potential woody debris, gravel, or other on-site depression storage features was estimated for each stream approach.

Streams crossed by trails were evaluated for substrate composition in streambed, channel width, and channel slope. Substrate material in each downstream and upstream reach was visually observed and classified as silt (<0.06 mm), sand (<2 mm), gravel (2–64 mm), cobble (65–256 mm), or boulder (257 to >1,024 mm) (adapted from Wolman 1954). Channel width was measured at each stream crossing (Table 1). Channel slope was measured over the upstream and downstream reaches combined (Table 1).

A rapid bioassessment using benthic macroinvertebrate sampling was used to detect changes in water quality downstream from each crossing. Upstream reaches provided reference conditions or a control to which to compare below crossing, downstream conditions. Macroinvertebrates were collected along a 100 m reach upstream and downstream from each of the 11 recreational stream crossings according to standard rapid bioassessment protocols for benthic macroinvertebrates (Barbour et al. 1999). Samples were collected over a 3-day period in late April 2013, the month when Pandapas Pond Recreation Area receives the third highest number of visitors (Kratzer 1993) with upstream and downstream reaches for each crossing being sampled on the same day. We acknowledge there could potentially be additional sources of non-point sediment input within the watershed other than the trail segments adjacent to the stream crossings, and as a result, we intentionally sampled and analyzed upstream and downstream reaches separately and used the upstream reach as a reference in order to minimize the influence of any other potential sediment inputs on downstream reach observations.

Using a 10 m buffer between the stream crossing and the upstream and downstream reaches, we collected 20 kick samples with a 0.3 m^2 D-frame dipnet with a 0.5-mm-mesh filter (Barbour et al. 1999). Kick samples were taken along each reach such that habitat type was sampled proportionally to its presence in the stream thus allowing us to associate changes in channel morphology with potential shifts in taxonomic richness. For example, if a reach was characterized by 50 % pools, 25 % riffles, and 25 %

woody debris, and then 10 kicks were conducted in pools, 5 in riffles, and 5 along or under woody debris. Sampling was conducted at the furthest downstream location first and then progressed upstream to avoid cross-contamination of samples. Materials collected from the kick samples were preserved with 70 % ethyl alcohol, placed in a cooler, and transported to the laboratory.

Laboratory Work

A total of 22 macroinvertebrate samples (11 crossings, upstream and downstream reaches sampled separately) were filtered and rinsed to remove excess sediment using a No. 30 sieve (0.595 mm). Large organic matter was rinsed and removed from the composited samples following an inspection for presence of macroinvertebrates. All macroinvertebrates were identified to family-level, as possible, and enumerated (Voshell and Wright 2002). Identification to family-level has proven to provide valuable information in analyzing macroinvertebrate-indices of water quality (Vowell 2001; Roy et al. 2003). Reference specimens were preserved in 95 % ethyl alcohol.

Data Analysis

Soil erosion rates were estimated for trail stream crossing approaches using Universal Soil Loss Equation for forestry (USLE-Forest; Dissmeyer and Foster 1984) and Water Erosion Prediction Project (WEPP, Windows interface version 2012.8) prediction models. USLE and WEPP models were originally developed for agricultural applications but have since been modified for forestry applications. USLE was the first widely accepted erosion prediction model and as a result has been widely used and modified in the past. WEPP was originally developed to replace USLE and to address areas where USLE was not intended for use, such as steep forested slopes. WEPP is physically based on hydrological and soil processes and requires additional input parameters than the USLE-Forest model, which is based on previously established empirical relationships between independent variables and quantified soil loss (Dun et al. 2009). Although WEPP was designed to replace USLE, we used both models in order to evaluate how soil erosion estimates on recreational stream crossing approaches modeled using USLE compare to other erosional studies that used USLE in the past and to conduct similar comparisons using WEPP estimates. Previous comparisons of USLE and WEPP estimated erosion rates with actual sediment trap data have illustrated that these models can vary in accuracy but have the capacity to effectively rank erosion rates from different land uses or BMP treatments (Tiwari et al. 2000; Wade et al. 2012; Brown et al. 2013).

The USLE for calculating potential soil loss is as follows:

$$A = R \times K \times LS \times CP$$

where A represents the computed soil loss per unit area per unit time, $R =$ rainfall and associated runoff impact, $K =$ soil erodibility factor, $LS =$ combined slope-length and slope-steepness factor, and $CP =$ a cover-management factor (Dissmeyer and Foster 1984). A rainfall index value (R) of 150 erosion index units (EI) was obtained for the Poverty Creek Trail System area from Wischmeier and Smith's isoerodent lines published in Dissmeyer and Foster (1984). For whole-soil erodibility factors (K), we used 0.28 for stream approaches on soils from the Jefferson series, 0.24 on Berks and Weikert soils, and 0.17 on Craigsville soils (USDA and NRCS 2013). For each approach, the slope-length and slope-steepness factor (LS) were calculated as

$$LS = \left(\frac{\lambda}{72.6}\right)^m \times (65.41 \sin^2\theta + 4.65 \sin\theta + 0.065)$$

where $\lambda =$ slope-length in feet, $\theta =$ slope angle in degrees, and $m = 0.2$ for slopes < 1 %, 0.3 for slopes 1 to 3 %, 0.4 for slopes 3.5 to 4.5 %, and 0.5 for slopes ≥ 5 % (Dissmeyer and Foster 1984). Recreational trail approaches were treated as tilled soils due to the absence of topsoil and heavy trafficking. Therefore, the cover-management practice factor (CP) was a product of the relevant subfactors for tilled soils including (a) bare soil, residual binding, and soil reconsolidation, (b) canopy above bare soil, (c) percentage of slope-length in sediment trapping steps, (d) presence of fine roots associated with invading vegetation, and (e) percentage of slope with on-site-depression storage features (Dissmeyer and Foster 1984).

Soil erosion rates were also estimated for each trail crossing approach using WEPP software which was downloaded from the National Soil Erosion Research Lab (http://www.ars.usda.gov/Research/docs.htm?docid=10621). WEPP associated climate, soil, and land cover files were also obtained from the National Soil Erosion Research Lab. Simulations predicting soil loss were conducted following input of slope characteristics and selected climate, soil, and land cover or management files. Local climate data were obtained through the CLIGEN (version 4.30) parameter file for station Blacksburg 2 VA which was located approximately 6 km from the crossing locations. Appropriate soil files for each Virginia-based soil series were determined based on each approach location, and the associated parameters were used for the WEPP simulations (USDA and NRCS 2013). Approaches situated on the Craigsville series had an interill erodibility 4.8×10^6 kg s m^{-4}, rill erodibility of 0.006 s m^{-1}, critical shear of 2.9 Pa, and a hydraulic conductivity of 8.6 mm h^{-1}. Parameter values

for approaches on the Jefferson series were 5.4×10^6 kg s m^{-4} for interill erodibility, 0.020 s m^{-1} for rill erodibility, 3.5 Pa for critical shear, and 5.11 mm h^{-1} for hydraulic conductivity. Approaches located on the Berks-Weikert soils had an interill erodibility 5.1×10^6 kg s m^{-4}, rill erodibility of 0.015 s m^{-1}, critical shear of 3.5 Pa, and a hydraulic conductivity of 3.8 mm h^{-1}. Recreational trails are currently not an existing management type in the WEPP model; therefore, we used the "Forest Bladed Road" management type as it was most similar to the unsurfaced trail approaches and despite the name, represented initial road conditions and did not include a recent blading treatment for trail segments. Modeled soil erosion rates were multiplied by the area of each approach to determine the potential sediment yield at each crossing. Modeled soil erosion rates and sediment yields estimated by USLE and WEPP models for the two approaches at each crossing were averaged so that one value represented each selected crossing. USLE-Forest and WEPP estimated soil erosion rates and sediment yields along the approaches were characterized and compared between culvert and ford crossing types using the NPAR1WAY procedure, two-sample Wilcoxon rank sum exact tests, in SAS version 9.3 (SAS 2012).

Macroinvertebrate data from each reach were entered into the Ecological Data Application System (EDAS) obtained from the Virginia Department of Environmental Quality (http://www.deq.virginia.gov/Programs/Water/WaterQualityInformationTMDLs/WaterQualityMonitoring/BiologicalMonitoring.aspx) to calculate two macroinvertebrate-based water quality indices: Family-level Hilsenhoff Biotic Index (FHBI) and Family-level Virginia Stream Condition Index (FVSCI). The FHBI is calculated (adapted from Lenat 1993) as

$$\mathrm{FHBI} = \sum \left(\frac{TV_i N_i}{N} \right)$$

where FHBI is the water quality index value that ranges from 0 (excellent) to 10 (poor); TV_i = the family-level tolerance value (0–10; least to most tolerant of pollution), N_i abundance of ith family or taxa, and N total number of individuals in the sample (Hilsenhoff 1988). The FVSCI is a multi-metric index that incorporates 8 metrics: (a) number EPT (pollution-sensitive Ephemeroptera, Plecoptera, and Trichoptera) taxa, (b) number total taxa, (c) % Ephemeroptera, (d) % Plecoptera and Trichoptera minus Hydropsychidae, (e) % Chironomidae, (f) % top two dominant taxa, (g) FHBI, and (h) % scrapers (Burton and Gerritsen 2003). FVSCI values range from 0 to 100, with FVSCI < 61 classified as "impaired" according to Virginia's DEQ monitoring standards. Based on these indices, if water quality decreases below stream crossings, we would expect to observe increased FHBI levels and decreased FVSCI based on the respective scale of each index. In addition to the calculated indices, we also noted changes that occurred in the percent of each sample composed by macroinvertebrates from the family Chironomidae and order Oligochaeta as these have been found to be more tolerant of increased sediment levels in streams (Longing et al. 2010). Clinger organisms, species that "cling" to boulders, cobble, and woody debris, were also observed for change in downstream reaches as they are less tolerant of increased fine sediment and thus, typically decline in number and richness in streams with higher sediment loads and turbidity levels (Longing et al. 2010; Pollard and Yuan 2010).

Correlations were examined to evaluate the relationship between estimated potential sediment yield and downstream change in the FHBI and FVSCI macroinvertebrate-based index values as biological indication of altered water quality below stream crossings. Specifically, Spearman's rank sum correlation coefficients (ρ) were evaluated between the percent change in FHBI and FVSCI indices downstream, (downstream − upstream value/upstream value) × 100 %, and predicted sediment yield calculated by multiplying each respective approach area by the predicted soil erosion rate by USLE and WEPP models. To evaluate whether stream crossing structure type is influential to macroinvertebrate communities, downstream changes in the FHBI and FVSCI indices were compared between culvert and ford crossing types using the NPAR1WAY procedure in SAS (SAS 2012). All statistical analyses were performed at a significance-level $\alpha = 0.05$.

Results

Modeled Soil Erosion Rates and Sediment Yields for Approaches

Modeled soil erosion rates were higher for unsurfaced recreational trail approaches leading to fords than to culvert crossings (Fig. 2; Table 2). Predicted erosion rates ranged from 1.2 to 8.8 tonnes ha^{-1} year^{-1} for culvert and 2.0–9.7 tonnes ha^{-1} year^{-1} for ford approaches based on the USLE-Forest model estimates. Annual erosion rates predicted by USLE-Forest were not significantly different between approaches to culvert and ford crossing types ($P = 0.429$; Fig. 2). WEPP estimated erosion rates ranged from 2.5 to 10.0 tonnes ha^{-1} year^{-1} for culvert approaches and 10.7–20.5 tonnes ha^{-1} year^{-1} for ford approaches. Annual soil erosion rates modeled using WEPP were significantly higher ($P = 0.004$) for approaches to fords than to culvert crossings (Fig. 2).

Predicted sediment yields, determined by multiplying the approach area by the respective modeled erosion rate, ranged from 4.0 to 20.3 kg year^{-1} for culvert and 1.8–50.8 kg year^{-1} for ford approaches when calculated

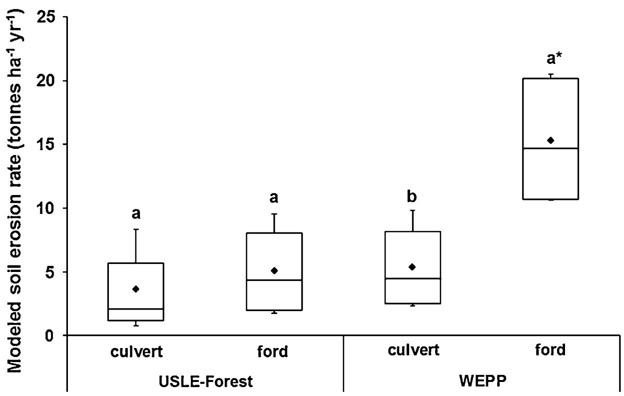

Fig. 2 Estimated soil erosion rates (tonnes ha^{-1} year^{-1}) for approaches to culvert and ford crossing types modeled using USLE-Forest and WEPP prediction models. Crossing types with *different letters* within each model were significantly different at $\alpha = 0.05$. Differences were determined using Wilcoxon rank sum exact tests. Diagonal data points represent means for culvert ($N = 6$) and ford ($N = 5$) crossings

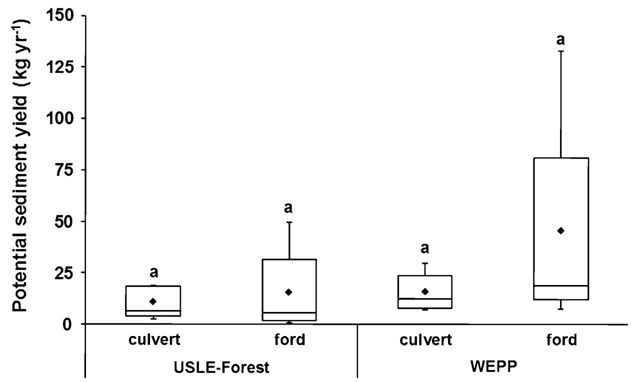

Fig. 3 Potential sediment yields (kg year^{-1}) for approaches to culvert and ford crossing types using USLE-Forest and WEPP prediction models. Crossing types with *different letters* within each model were significantly different at $\alpha = 0.05$. Differences were determined using Wilcoxon rank sum exact tests. Diagonal data points represent means for culvert ($N = 6$) and ford ($N = 5$) crossings

Table 2 Average slope, modeled soil erosion rate (tonnes ha^{-1} year^{-1}), and potential sediment yield (kg year^{-1}) for trail approaches to selected culvert (C) and ford (F) stream crossings in the Poverty Creek Trail System in Southwestern Virginia

Crossing	Slope (%)	Estimated soil erosion rate (tonnes ha^{-1} year^{-1})		Potential sediment yield (kg year^{-1})	
		USLE-Forest	WEPP	USLE-Forest	WEPP
C1	2.0 (1.4)	1.2 (0.9)	2.7 (0.4)	5.9 (5.3)	12.9 (5.1)
C2	1.0 (0.0)	2.3 (0.0)	2.5 (0.0)	7.1 (0.5)	7.7 (0.5)
C3	4.0 (2.8)	5.2 (4.8)	7.8 (7.1)	20.3 (19.3)	30.4 (28.2)
C4	7.5 (0.7)	8.8 (1.7)	10.0 (2.5)	19.6 (2.6)	22.2 (4.2)
C5	2.0 (1.4)	1.7 (0.6)	2.9 (0.6)	8.4 (9.3)	13.0 (13.5)
C6	4.8 (0.4)	2.7 (1.9)	6.4 (1.0)	4.0 (3.4)	8.5 (0.5)
F1	9.0 (7.1)	4.6 (3.4)	10.7 (10.6)	14.2 (10.7)	33.3 (33.3)
F2	12.0 (0.0)	2.4 (0.4)	20.5 (8.9)	1.8 (1.7)	20.9 (24.8)
F3	9.5 (3.5)	6.7 (2.6)	19.8 (8.7)	50.8 (28.5)	137.5 (34.9)
F4	4.5 (2.1)	2.0 (0.2)	10.7 (6.0)	4.1 (0.1)	23.1 (15.1)
F5	10.5 (2.1)	9.7 (7.8)	14.7 (4.5)	6.8 (2.0)	12.0 (3.4)

Values were estimated using Universal Soil Loss Equation for Forestry (USLE-Forest) and Water Erosion Prediction Project (WEPP) models. Presented crossing values are the average of the left and right approach values with the standard deviation in parentheses

using USLE-Forest (Fig. 3; Table 2). Predictions from the WEPP model for approaches to culvert crossing types ranged from 7.7 to 30.4 kg year^{-1}, whereas ford approaches ranged from 12.0 to 137.5 kg year^{-1} in estimated sediment yield. Sediment yield values along the approaches were not significantly different between culvert and ford crossing types based on modeled estimates from USLE-Forest ($P = 0.792$) or WEPP ($P = 0.177$).

Stream Characterization for Macroinvertebrate Habitat

Streams crossed by culverts were classified as either first- or second-order intermittent streams, while fords crossed a mixture of first- and second-order intermittent as well as the first-order perennial Poverty Creek (Fig. 1; Table 1). Streambeds were primarily composed of cobble and gravel which is typical of first-order mountainous streams in the southern Appalachian Mountains. The three fords (F1, F3, F5) crossing the first-order perennial Poverty Creek were also characterized by boulders in the streambed. Smaller particles, sand and silt, were observed more frequently in downstream reaches than in upstream reference reaches.

Stream Crossing Impacts on Macroinvertebrates and Water Quality

Downstream reaches had higher FHBI and lower FVSCI values which reflected a decrease in water quality below

Table 3 Downstream changes (downstream − upstream) in Family-level Hilsenhoff Biotic Index (FHBI) and Family-level Virginia Stream Condition Index (FVSCI) values and along with some of the metrics reflected in the FVSCI: number of total taxa; percent of sample composed by Ephemeroptera, Plecoptera, and Trichoptera minus Hydropsychidae (EPT), Chironomidae, Oligochaeta, and clingers

Crossing	Stream class and order	Downstream changes						
		Index		Metric				
		FHBI	FVSCI	Taxa (#)	EPT (%)	Chironomidae (%)	Oligochaeta (%)	Clingers (%)
C1	I (1)	0.3	−7.0	−4	−14	9	9	−13
C2	I (2)	0.4	−14.8	−1	−25	−4	33	−14
C3	I (2)	0.5	−8.8	0	−11	−2	−1	−3
C4	I (1)	1.2	−23.7	−4	−33	3	37	−29
C5	I (1)	1.8	−13.5	−4	−54	14	36	−37
C6	I (2)	0.0	−4.5	−1	1	2	−2	−7
Average		*0.7*	*−12.1*	*−2.3*	*−22*	*4*	*19*	*−17*
F1	P (1)	0.9	−26.8	−4	−42	1	50	−25
F2	I (2)	0.9	−1.7	0	−33	1	22	−28
F3	P (1)	1.1	−11.7	1	−40	5	31	−22
F4	I (1)	0.5	3.0	0	11	0	−2	2
F5	P (1)	1.1	−7.8	1	−23	10	13	−22
Average		*0.9*	*−9.0*	*−0.4*	*−25*	*3*	*23*	*−19*

Positive FHBI and negative FVSCI values indicate a decrease in associated water quality for downstream reaches when compared to upstream reference reaches at for the 11 crossings examined in the Poverty Creek Trail System. Change in metric values was negative if metric values decreased downstream. Values in italics represent average change for culvert and ford crossing types. Streams crossed by recreational trails are abbreviated as I (intermittent) and P (perennial)

stream crossings, downstream, than in upstream reaches (Table 3). As previously described, the FHBI values range from 0 (excellent) to 10 (very poor quality), while FVSCI values range from 0 to 100, with values <61 indicating impairment. A majority (75 %) of upstream reaches were classified as having "excellent" water quality according to FHBI and were "not impaired" (58 %) as determined by FVSCI. However, when downstream reaches were examined, a different trend surfaced. Fewer downstream reaches were considered to have "excellent" (17 %) water quality with more downstream reaches falling into the "very good" (42 %), "good" (33 %), and "fair" (8 %) FHBI categories. Based on the FVSCI, the majority of reaches downstream from crossings had 'impaired" (83 %) water quality.

Evaluation of Spearman's rank sum correlation coefficients indicated our USLE-Forest sediment yield estimates along the crossing approaches were significantly correlated ($\rho = -0.69$; $P = 0.019$) with percent downstream change in FVSCI values. However, the relationships between percent downstream change in FHBI and USLE-Forest sediment yield estimates ($\rho = 0.20$; $P = 0.555$) or between the percent changes in FHBI ($\rho = 0.25$; $P = 0.467$) and FVSCI ($\rho = -0.22$; $P = 0.518$) and WEPP sediment yield estimates were not significant. Crossing type did not appear to have a significant influence on the observed percent change in downstream values for FHBI ($P = 0.177$) and FVSCI ($P = 0.662$) between culvert and ford crossing types. Mean downstream changes in FHBI and FVSCI appeared to be similar between crossings located on perennial streams ($n = 3$; FHBI $= +1.0$ and FVSCI $= -15.5$) when compared to crossings located on intermittent streams ($n = 8$; FHBI $= +0.7$ and FVSCI $= -9.0$; Table 3).

A total of 29 families representing 9 orders were observed during macroinvertebrate sampling in upstream and downstream reaches (Table 4). Of particular importance were families of the more pollution sensitive order Ephemeroptera: Baetidae, Ephemerellidae, Ephemeridae, and Heptageniidae and order Plecoptera: Chloroperlidae, Nemouridae, Perlidae, and Perlodidae. Although a majority of these families were present in both upstream and downstream reaches, changes in the numbers proportion and dominance of each taxa found in reach samples occurred, thus creating change to the calculated macroinvertebrate-based water quality indices (Table 3). As an additional indication of change: total number of taxa; percent EPT (excluding the more pollution tolerant Hydropsychidae); and the percent of macroinvertebrates classified as clingers decreased below most recreational stream crossings (Table 3). The proportion of macroinvertebrates of the more pollution tolerant Chironomidae and Oligochaeta organisms was higher in nearly all downstream reaches (75 %).

Table 4 List of macroinvertebrates (order: family) found in reaches above (upstream) and below (downstream) the 11 selected stream crossings along the Poverty Creek Trail System

Order	Family	Upstream	Downstream
Amphipoda	Unknown	x	x
Coleoptera	Unknown	x	
	Dytiscidae		x
	Elmidae	x	x
	Psephenidae	x	x
Diptera	Athericidae	x	x
	Ceratopogonidae	x	x
	Chironomidae	x	x
	Unknown	x	x
	Empididae	x	x
	Simuliidae	x	x
	Stratiomyidae	x	
	Tabanidae	x	x
	Tipulidae	x	x
Ephemeroptera	Baetidae	x	x
	Ephemerellidae	x	
	Ephemeridae	x	x
	Heptageniidae	x	x
Megaloptera	Corydalidae	x	x
	Sialidae		x
Odonata	Gomphidae	x	x
	Libellulidae		x
Oligochaeta	Unknown	x	x
Plecoptera	Chloroperlidae	x	
	Nemouridae	x	x
	Perlidae	x	x
	Perlodidae	x	x
Trichoptera	Hydropsychidae	x	x
	Lepidostomatidae	x	x
	Limnephilidae	x	x
	Philopotamidae	x	x
	Polycentropodidae	x	x

Discussion

Modeled Soil Erosion Rates at Stream Crossings

Estimated soil erosion rates and corresponding sediment yield potentials predicted by WEPP were consistently higher than estimates derived from the USLE-Forest (Table 2). USLE-Forest and WEPP are models and were used in this to study as a tool to provide multiple soil erosion rate estimates and to allow for comparison of rates published for other forestland disturbances to characterize the magnitude of soil erosion along recreational trail approaches to stream crossings. Similar discrepancies between USLE-Forest and WEPP models have been reported for skid trails, forest roads, and forest road approaches to stream crossings (Aust et al. 2011; Wade et al. 2012; Brown et al. 2013). However, in each of these studies, despite accuracy issues, USLE-Forest and WEPP models proved capable of ranking conditions with greatest erosion potentials. Estimates associated with the WEPP model were based on the management file "Forest Bladed Road" which potentially contributed to higher erosion rate predictions in WEPP than in the USLE-Forest model. This management type was chosen as it was most similar to the conditions along the unsurfaced, high-traffic recreational trails. Differences in model parameters also contributed to estimate differences between the two models. USLE-Forest incorporates one overall erodibility factor for each soil series while WEPP includes different rill and interill erodibility factors, rate of effective hydraulic conductivity, and critical shear values for each soil type. Due to these differences between models, predicted soil erosion rates from this study were only compared with estimates from other studies that used the same erosion prediction model.

Soil erosion rates predicted by USLE-Forest were higher for recreational trail approaches (an average of 4.1 tonnes ha^{-1} year^{-1} for all stream crossings; Fig. 2) than published USLE-Forest estimates for adjacent forests. In the same forest type and physiographic region as this study, average estimated erosion rates were found to be 0.3 tonnes ha^{-1} year^{-1} in undisturbed forests and 1.7 tonnes ha^{-1} year^{-1} in 2-year old clearcuts (Hood et al. 2002). Erosion estimates from this study were found to be higher than published erosion estimates (0.0–0.7 tonnes ha^{-1} year^{-1}) for eastern forests (Patric 1976). If trail surfaces are highly eroded, as in the case of the Poverty Creek Trail System, recreationists' enjoyment of the trail-use experience may be significantly decreased (Lynn and Brown 2003).

Unsurfaced recreational trails examined in this study were used for multiple non-motorized (horseback riding, mountain biking, and hiking) activities. Average WEPP erosion rate estimates for non-motorized trail approaches to stream crossings in this study were lower (average of 9.7 tonnes ha^{-1} year^{-1} for all crossings; Fig. 2) than WEPP estimates for motorized off-road vehicle (ORV) recreational trail approaches to a single ford stream crossing which reached 126.8 tonnes ha^{-1} (Ayala et al. 2005). USLE-Forest erosion estimates in this study were also lower than USLE-Forest erosion estimates reported along bare horse trails (34.6–69.2 tonnes ha^{-1} year^{-1}) but, within the range of estimates where gravel had been applied (2.0–16.3 tonnes ha^{-1} year^{-1}) (Aust et al. 2005). Due to the limited number of recreation-based soil erosion studies, erosion estimates for the unsurfaced trail approaches were also compared with erosion rates along forest

operation skid trails. Soil erosion estimates from the USLE-Forest model along the multiple-use recreational trail stream approaches were lower than USLE-Forest predicted erosion rates along bladed forest skid trails where minimal, water bars only, BMPs were implemented (63.1 tonnes ha^{-1} year^{-1}) (Wade et al. 2012). Erosion estimates in this study were more similar to USLE predicted soil erosion rates for forest skid trails where hardwood slash (4.3 tonnes ha^{-1} year^{-1}), mulch (3.2 tonnes ha^{-1} year^{-1}), and pine slash (1.6 tonnes ha^{-1} year^{-1}) had been applied, according to BMPs to reduce soil erosion rates, than to erosion rates on bare trails (Wade et al. 2012).

Approaches leading to ford crossings had greater predicted USLE-Forest and WEPP soil erosion rate estimates than approaches to culverts (Fig. 2). Differences in approach erosion estimates between the two crossing types were significant when the WEPP erosion estimates were evaluated, but not when based on USLE-Forest estimates. Observed differences were most likely attributed to site conditions on which the approaches occurred. For instance, slopes along approaches to fords (9.1 %) were greater on average than those to culverts (3.5 %) (Table 2). Steeper slopes contributed to greater combined slope-length and slope-steepness factor (*LS*) values in the USLE-Forest model for ford approaches (0.73) when compared to culvert approaches (0.27) and had a similar impact on predicted estimates in the WEPP model. Trails and roads characterized by steeper grades or approach lengths have been shown to have higher actual and modeled soil erosion rates (Dissmeyer and Foster 1984; Olive and Marion 2009; Brown et al. 2013).

Stream Crossing Impacts on Macroinvertebrates and Water Quality

Downstream changes in macroinvertebrate-based index and associated metric values observed in this study illustrated a detectable decrease in water quality below trail stream crossings (Table 3). Previous research conducted below road and trail stream crossings has illustrated the negative impacts of soil erosion along crossing approaches and associated crossing traffic disturbances can have on water quality (Lane and Sheridan 2002; Ayala et al. 2005; Aust et al. 2011; Wear et al. 2013). Unsurfaced approaches have been cited as the major source of sediment deposition downstream from low-water ORV ford crossings (Brown 1994). Similar to water quality impacts at forest operational crossings, water samples collected downstream from an ORV ford crossing were found to contain higher suspended sediment loads than samples from upstream controls (Ayala et al. 2005). In this study, a negative relationship was found between increased potential sediment yields and downstream change in the FVSCI, illustrating that as estimated sediment yield increased, the FVSCI value decreased indicating decreased water quality. Similarly, a positive relationship was identified between sediment yields and FHBI indices illustrating that as sediment yield increased the FHBI value increased, also indicating a trend of decreased water quality with increased estimated sediment yield levels.

Stream crossing type, culvert or ford, did not appear to impact changes in the macroinvertebrate-based indices illustrating that water quality was impacted by stream crossings in general rather than by specific crossing type in this study. Further, although downstream changes in macroinvertebrates were detected indicating water or habitat quality changes were occurring, many of the changes were not significant from a statistical standpoint. Other studies have reported that there may be a time lag between decreased water quality and detection of these differences in macroinvertebrate populations (Kaller and Hartman 2004). Given the evidence we found of changes in streambed substratum composition below crossings (silt and sand were more frequent; Table 1) the full implications of stream crossing erosion may have yet to become detectable in the macroinvertebrate populations. Changes in streambed conditions may translate into changes in macroinvertebrate behaviors rather than detectable changes in macroinvertebrate populations (Rosenberg and Wiens 1978). Further, changes in water quality in this study were observed primarily based on changes in calculated index values; these values serve to indicate change and are not an absolute measure of abiotic factors associated with water quality. In this study, spatial location of the crossings within the watershed may have also influenced water quality and macroinvertebrate assemblages as topography, hydrology, and in-stream habitat can influence presence of macroinvertebrate assemblages (Longing et al. 2010). For example, Culvert 6, located lower in the watershed than the other crossings, had an 'impaired' upstream reach according to FVSCI criteria (Fig. 1). This upstream impairment may have masked downstream changes in macroinvertebrate-based index and metric values (Table 3).

Conclusions

We modeled soil erosion rates that were 13 times higher along trail approaches than in previously evaluated adjacent control forestlands (Hood et al. 2002). Estimated soil erosion rates were within the same range as skid trails and horse trails where BMPs had been implemented. Despite crossing type, downstream changes in FHBI and FVSCI values indicated reduced water quality with increased estimated sediment yield potentials despite whether

approaches were to a culvert or ford crossing type. Downstream changes in macroinvertebrate-based water quality index values illustrated recreational stream crossings are having an impact on conditions in streams crossed by recreational trails in the Poverty Creek Trail System. Results from this study further support the need for implementing BMPs to reduce soil erosion and protect water quality.

Increased usage rates and expansion of recreational trails justify the timeliness of evaluating the impacts of recreational trail approaches and stream crossings on water quality in other recreational areas and should be considered in designing or re-locating trails. Future research should be aimed at quantifying trail-eroded sediment that is actually delivered into streams at stream crossings, measurement of below crossing changes in stress-related water quality variables such as turbidity, TSS, and nutrient-levels, and evaluation of the effectiveness of recreational BMPs at reducing soil erosion and ameliorating sediment entry into waterways.

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CHATTOOGA RIVER WATERSHED ECOLOGICAL/SEDIMENTATION PROJECT

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Abstract As an integral part of the comprehensive water quality investigation of the Chattooga River watershed, an ecological and sedimentological study was conducted on selected stream reaches within the study area. The objective of this study was to conduct a sediment yield study and determine if sediment was a primary cause of physical and biological impairment to streams within the watershed. As result of this study, accelerated sedimentation has been identified to be the leading determinant in loss of habitat and reduction in bedform diversity within the study area. Good correlation was observed between aquatic ecology and normalized total suspended solids (TSS) data. Based on overlaying the biological index on TSS normalized to discharge/mean discharge, TSS concentrations greater than 284 mg/l adversely affected aquatic macroinvertebrate community structure. However, based on historic regional suspended-sediment concentrations, a normalized TSS concentration of 58 mg/l or less during storm flow provides an adequate margin of safety and is protective of aquatic macroinvertebrates in the Blue Ridge physiography. Corresponding turbidity limits of 69 and 22 NTU established the threshold of biological impairment and margin of safety, respectively. Previously, a similar turbidity of 25 NTU has been recommended for stream restoration management plans. Relative to reference streams, impaired streams yielded higher bedload and suspended load. The results of this study showed that road density and associated sediment sources accounted for 51% of the total sediment loading.

INTRODUCTION

In response to issues included in the settlement of the Georgia Total Maximum Daily Load (TMDL) lawsuit, EPA was required to conduct an evaluation of the Chattooga River watershed to determine if waters within the watershed were not meeting designated uses (Sierra Club, Georgia Environmental Organizations, Inc., Coosa River Basin Initiative, Inc., Trout Unlimited, and the Ogeechee River Valley Association, Inc., Versus: U.S. Environment Protection Agency (EPA); Carol Browner, Administrator, EPA and John Hankinson, Regional Administrator, EPA Region 4). For those waters not meeting designated uses, EPA was required to determine the cause of non-support and develop the appropriate TMDL.

Sedimentation has been reported to be the leading determinant in loss of habitat and reduction in bedform diversity within the study area. The State of Georgia is initiating a statewide effort and geographic calibration of reference conditions for assessing the ecological status of its water resources using biological assessment. However, the effort has not been completed. As an interim solution, it was necessary to develop reference conditions at the scale of the Chattooga Basin. The objective of this study was to conduct a sediment yield study and determine if sediment was a primary cause of physical and biological impairment to streams within the watershed. The results were correlated with aquatic ecological data to develop an overall condition of the watershed.

Setting The Chattooga River watershed, located in northeast Georgia, northwest South Carolina, and southwest North Carolina, has a total drainage area of approximately 180,000 acres, and is entirely within the Blue Ridge Ecoregion. Land cover within the watershed is primarily forested, with some areas of commercial development, urban and residential use, and agriculture. Although the average "forested" land cover within the watershed is greater than 96%, there has been concern that gradual increases in sediment inputs to streams may be causing ecological impairment. Consequently, EPA Region 4 began an evaluation of water quality conditions within the Chattooga River watershed, and how they may have changed due to forestry or forestry-related practices. To accomplish this, sampling and analysis was undertaken in 1997-2000 by U.S. EPA Region 4 for biological and habitat quality, channel morphology, selected water chemistry, and sediment yield.

METHODS

Aquatic Ecology A total of 3 reference sites and 56 other sites were sampled from six subwatersheds: Headwaters (n = 14), Lower Chattooga (n = 3), Middle Chattooga (n = 10), Stekoa Creek (n = 7), West Fork (n = 11), and Warwoman Creek (n = 11). Biological sampling methods were focused on benthic macroinvertebrates and used modified rapid bioassessment protocols (RBP) (Plafkin et al. 1989, Barbour et al.1999, and U.S. EPA's Region 4, Ecological Assessment Branch-*Draft* Standard Operating Procedures 1999). Reference sites were selected prior to initiation of sampling based on habitat condition, *in situ* water chemistry and surrounding land use. Reference sites R1 and R2 were located in the Chattooga River watershed and reference site R3 was on the upper Chattahoochee River outside of the Chattooga watershed. It was determined that the reference sites were representative of least-impaired conditions of the Blue Ridge Ecoregion. Data for all 59 stations were analyzed using a multimetric approach, in agreement with the recommendations of U. S. EPA (Gibson et al. 1996). From the raw data, 17 metrics were calculated including: total taxa, number of Ephemeroptera , Plecoptera, and Trichoptera (EPT) taxa , number of clinger taxa (clingers),percent clingers, percent most dominant taxon, percent 2^{nd} dominant taxa, percent tolerant organisms, number of intolerant taxa, percent diptera, percent Chironomidae, percent EPT, North Carolina Biotic Index (NCBI), percent collectors, percent filterers, percent scrapers, percent shredders, and percent predators.

From the original list of 17 metrics, five were selected that had the greatest ability to detect impairment, determined by examining the position of the *a priori* reference sites to the overall distribution of metric values. For the most appropriate metrics, scoring criteria were determined based on the 95^{th} percentile of all metric values for those metrics that decrease with impairment (Barbour et al.1999). For those that increase with impairment, the 5^{th} percentile was used. This approach was used since there were no *a priori* impaired sites against which to calibrate. Each metric was scored according to its relation to the 95^{th} (or 5^{th}) percentile standard (Table 1). Eighty-five percent (85%) of the area below the 95^{th} percentile standard (or 15% above the 5^{th} percentile) was equally divided into four ranges and each range is given a numeric value of 0, 2, 4, or 6. A score of zero was the farthest away from the percentile standard (i.e., zero was most unlike the best attainable conditions and 6 was the score closest to the percentile standard). One exception was the "North Carolina Biotic Index" (NCBI), for which the scoring criteria developed by Lenat (1993) were used.

Table 1. Table of metrics and percentile distribution for each.

Metric	Min	05^{th}	Median	95th	Max	Percentile Standard	Expected Response to Stressors
EPT taxa	3	10	15	21	25	95	Decrease
% EPT	27.9	36.7	66.7	85.0	95.4	95	Decrease
% 2 dominant taxon	19.2	22.0	30.0	52.8	65.4	5	Increase
NCBI	2.6	2.7	4.1	5.6	6.2	5	Increase
Clinger taxa	7	7	17	23	24	95	Decrease

A final biological index was assigned to each site based on a simple sum of the scores for the five metrics. An assessment rating was then assigned by dividing the range of the overall index scores into 5 categories. Narrative descriptions of the assessments correspond to:

< **Very Good** - best attainable conditions indicating no impairment to the aquatic community;
< **Good** - close to best attainable conditions but at risk and possibly influenced by limited stressors;
< **Fair** - some biological impairment observed, due to minor stressor input;
< **Poor** - substantial impairment of stream biota observed, due to moderate stressor input; including habitat degradation;
< **Very Poor** - severe impairment of stream biota observed, due to major stressor input, including habitat degradation.

Sediment Sampling Seventeen stream reaches were selected for storm flow investigations based on the following criteria: (1) relative degree of biological impairment as measured using RBP; (2) position within the watershed; (3) relative geomorphic condition; and (3) access logistics. The storm flow investigations were conducted during three storm events (March 28-30, 1998, June 15-17, 1999 and March 16-17, 2000). Prior to storm flow sampling, tape

downs were established and appropriate cross-sections for gaging and sediment collection were identified. Base flow discharge and sediment samples were collected prior to the storm initiation. Precipitation was measured at Clayton, Georgia for response planning and rapid deployment of sample teams during the storm flow study. In addition, several rain gages were strategically deployed within the watershed to address rainfall distribution. Also, stream stage was monitored in Stekoa Creek at Clayton for response planning.

A total of 58 observations were made across the 17 stations. *In-situ* measurements at each station included tape downs (start and finish), stream discharge, turbidity, and collection of suspended and bedload sediment. Stream discharge was gaged simultaneously with sediment collection. Water column samples were collected using a depth integrating suspended hand-line sampler (US DH-59). Field turbidity was determined *in-situ* at ambient air conditions using a HACH™ Model 2100P Turbidity Meter. Turbidity was field determined for future use by EPA Region IV and state water quality personnel as a rapid means of identifying potential sediment impaired streams ("red flags"). Consequently, sample temperature was not adjusted prior to measuring turbidity. Laboratory determination of total suspended solids (TSS) and total dissolved solids (TDS) followed USEPA Methods 160.2 and 160.1, respectively. Whole samples were filtered for TSS analysis. Because the TSS data were produced without subsampling, they should be directly comparable to suspended-sediment concentration data (SSC) (Gray et al. 2000 and personal communication with John Gray, USGS). Bedload sediment samples were collected utilizing a 6-inch cable suspended bedload sampler or a 6-inch wading type bedload sampler, transported to the laboratory in 1-liter containers, and processed for particle size determination (PSD) in the laboratory using the EPA-SESD wet sieve method (SESD-EAB Draft SOP, Jan. 99). The procedure was followed with the exception of the silt/clay separation step that was not required since the samples were collected in coarse Nitex™ mesh bags (250 :m).

Laboratory results of dry-weight, bedload samples (M_b, grams) were converted to bedload transport rate (Q_b, tons/day) by the following equation (Edwards and Glysson 1988):

$$Q_B = K(W_T/T)\, M_T \qquad (1)$$

where Q_B = bedload discharge (tons/day);
K = converts grams/second/foot to tons/day/foot
W_T = wetted surface (ft);
T = total time sampler on bottom (seconds);
M_T = total mass of samples (grams)

Regression relationships were tested against ANOVA at a 95% confidence level. Consequently, unless otherwise noted hereafter, significance was determined at $\alpha = 0.05$, based on a t-test using advanced regression.

RESULTS

Aquatic Ecology Biological conditions in most streams sampled in this study show little or no impairment. Seventy-eight percent (78%) of the sites were rated as "very good" (22 sites) or "good" (24 sites). Since greater than 96% of the watershed land cover is classified as forested, this result was expected. Streams rated as "good" (41% of all stream sites sampled) are defined as possibly being influenced by some stressors. Eleven sites (19%) were rated as "fair", and two sites (3%) were rated as "poor". No sites were rated as "very poor". Although some sedimentation, or the habitat effects of sedimentation, may have been evident at many sites, a negative biological response was not always evident. The sedimentation also may not have reached a level that would cause a biological response. Due to the fact that this project used multihabitat sampling of benthic invertebrates, samples were taken from some stream subhabitats that were not adversely affected by sediment deposition resulting in habitat loss. The three reference sites had high biological scores: 24, 22, and 28, respectively, out of a maximum possible score of 30. The most degraded biological community was observed in the Stekoa Creek subwatershed. This subwatershed has a higher percentage of bare land and less forest cover than other subwatersheds in the Chattooga River basin. Consequently, none of the sample stations were rated as "very good" (i.e., zero out of seven stations). Two stations were rated "good", four stations were rated as "fair", and one station was rated as "poor".

Bedload Sediment Bedload over the three storm events averaged 13.32 tons/day (range 0.02-176.96 tons/day, standard deviation = 41.28). Median bedload particle sizes (D_{50}) ranged from fine sand to very coarse sand. Bedload accounted for only 14 percent of the total sediment load (on average). By plotting bedload against discharge, bedload sediment rating curves for each of the three storm events were created (Figure 1). Relatively

good regression coefficients were observed within each storm event. However, regressed slopes varied between storm events.

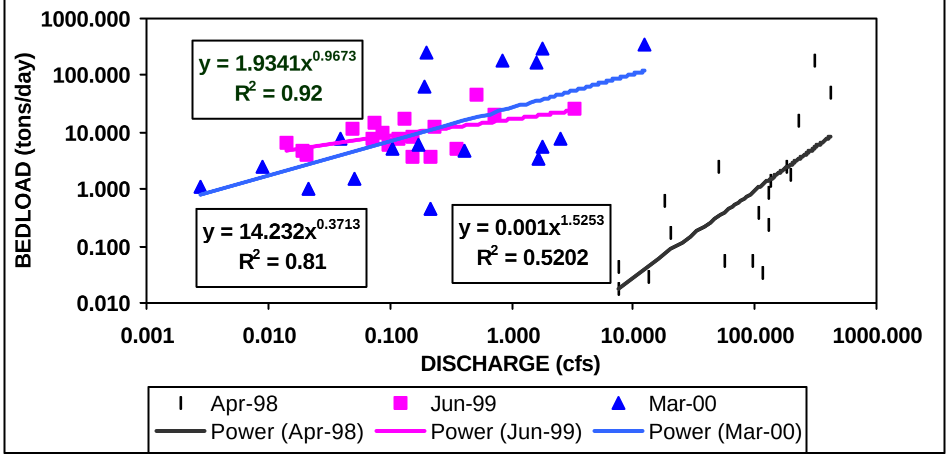


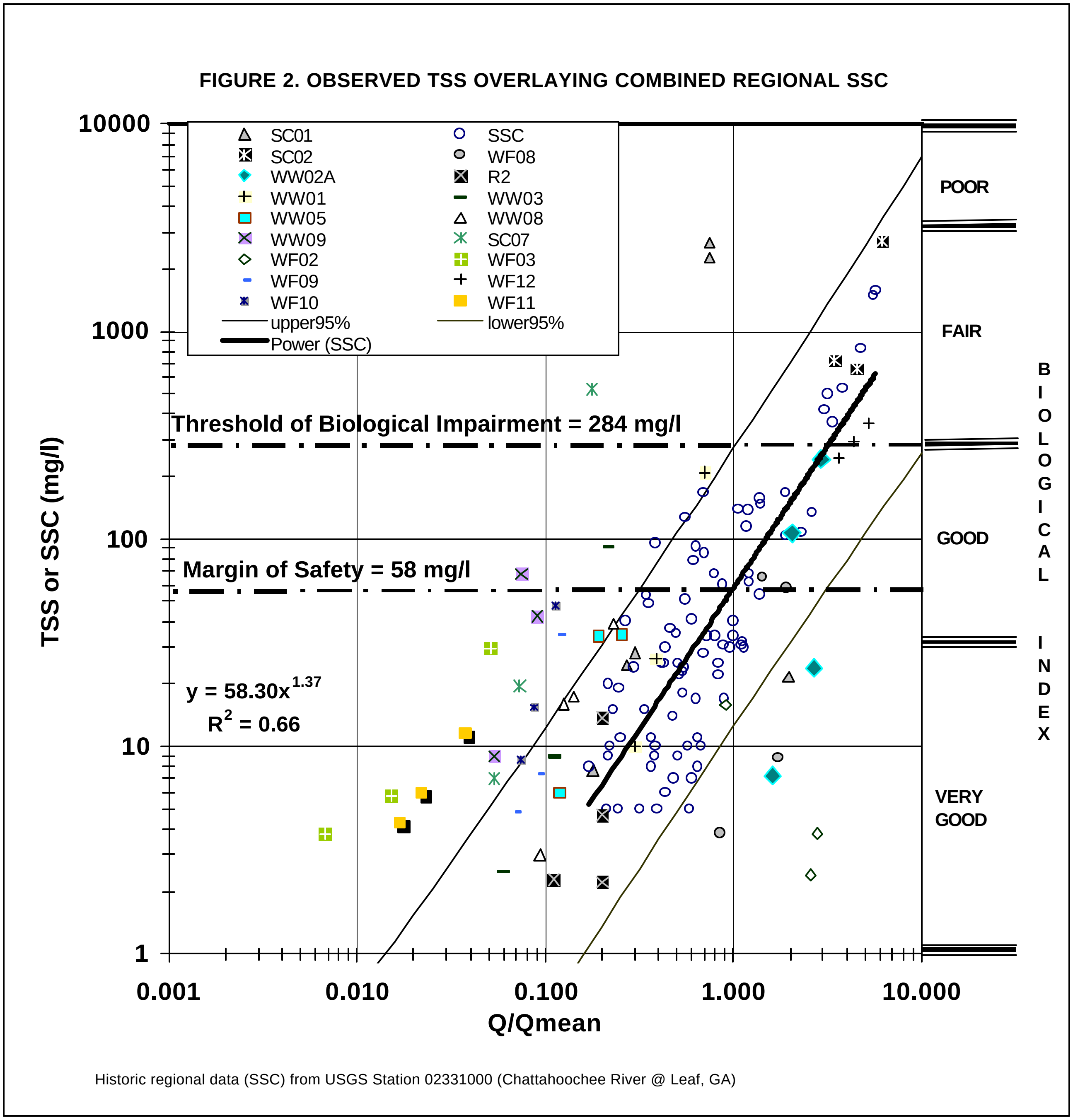
FIGURE 1. BEDLOAD SEDIMENT RATING CURVES

Suspended Sediment (Regional) Regional SSC data, compiled from the United States Geologic Survey records (Perlman 1984), were regressed against discharge normalized to mean discharge (Q/meanQ) (Holmbeck-Pelham and Rasmussen 1997). The USGS stream station utilized in development of the regional sediment curve was the Chattahoochee River near Leaf (Station no. 02331000) for the period of record, 1958 - 1984. TSS data from the Soque River station near Cornelia (02331250) and the Chestatee River near Dahlonega (02333500) were not used due to the difference in slope of the regression as compared to the Chattahoochee River station in the former and shift upward in the regression of the latter. An improvement was observed in the regression coefficient from 0.54 to 0.66 and, consequently, confidence in using the regional data set improved as a reference. In addition, SSC data from the Chattahoochee River was the most protective as compared to the other two datasets. Regional SSC (from the Chattahoochee River) regressed against Q/Qmean was observed to be significant (R^2=0.66, log transformed), given by (Figure 2):

$$\text{TSS or SSC} = 58.3(Q/Qmean)^{1.37} \qquad (2)$$

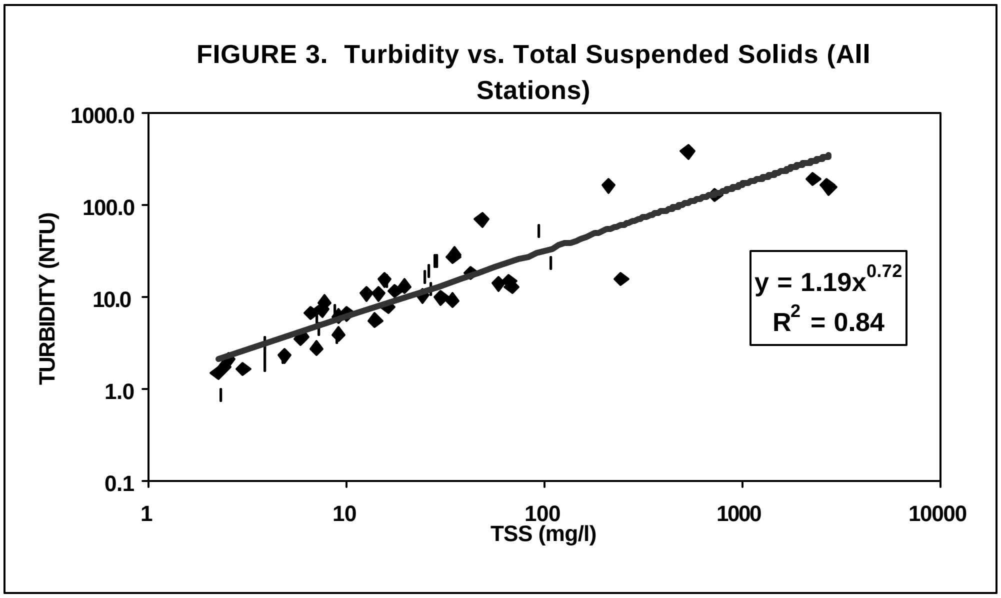
Suspended Sediment (this study) TSS over the three storm events averaged 85.3 tons/day (range 0.0002-3136.2 tons/day, standard deviation = 418.0). TSS accounted for the majority (86 %) of the total sediment load over the three storm events (on average). TSS, collected by vertical integration of the water column, was regressed against discharge (Q) and was observed to be highly variable between stations during the same storm event and between different storm events. In contrast, the log transformed relationship between TSS and NTU was significant (Figure 3). TSS data were compared against regional SSC by overlaying the two and constructing 95% confidence bands (Figure 2). Six stations, SC01, SC07, WW09, WF03, WF10 and WF11, were observed above the upper 95% confidence band (i.e., 6 out of the 17 stations during the three stormflow investigations). In general, data points that plot above the upper 95% confidence band are indicative of higher than "normal" concentrations of TSS for a given discharge to mean discharge. Other stations were observed to be below or within the normal range of the regional SSC data set. In addition, three stations, WW02A, WF02, and WF08, were below the lower 95% confidence band.

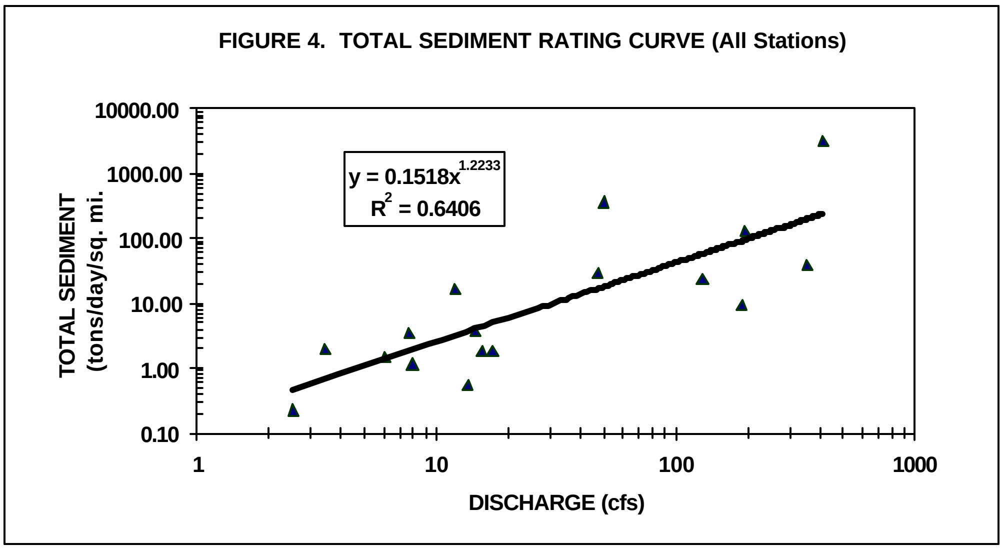
Total Sediment Bedload and TSS loadings were combined into total sediment load and plotted against discharge (Figure 4). Total loads were also plotted against road density (road length / corresponding drainage area) (Figure 5). Road density ranged from zero (R2 - Addie Branch, reference) to 6.60 (SC01 - Stekoa Creek. Road density represents the net impacts of road construction and maintenance, interception of subsurface interflow, routing of other non-point sources to the stream, and entrainment, mobilization, and transport of sediment to the stream. In contrast to drainage density, a significant increase in peak total loads in response to road density was observed at the two Stekoa Creek stations (SC01 and SC02).



CONCLUSIONS

Good correlation was observed between the biological index and normalized total suspended solids (TSS) data (Figure 2). TSS concentration normalized to discharge/mean discharge greater than 284 mg/l adversely affected biological community structure. However, based on regional suspended-sediment concentrations, a normalized TSS concentration of 58 mg/l or less during storm flow provides an adequate margin of safety and is protective of aquatic macroinvertebrates in the Blue Ridge physiography. Furthermore, corresponding turbidity limits from the above TSS estimates can be calculated from the NTU versus TSS relationship (Figure 3) as 69 and 22 NTU for the threshold of biological impairment and margin of safety, respectively.





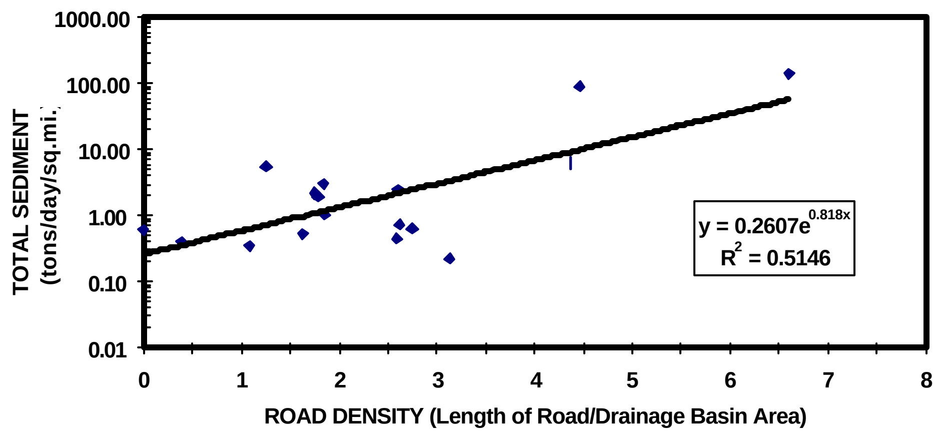
Relative to the reference stream (R2), impaired streams yielded higher bedload and suspended load. Based on the results of this study and comparison against regional sediment data, Stekoa Creek (SC01 and SC07) exhibits greater than “normal” suspended sediment loads. TSS concentrations from Addie Branch (R2) were within or below “normal” regional TSS concentrations. Total storm flow sediment load and peak total sediment loads did not increase significantly with drainage density. Increased sediment loads were correlated with an increase in road density. Road density and associated sediment sources accounted for 51% of the total sediment loading. Assuming that every road has at least one road ditch, road density nearly doubled the effective drainage density at the Stekoa Creek stations. The condition of the macroinvertebrate community of Stekoa Creek is rated as “fair” and is evidence of the impact of the accelerated sediment loads in the stream at stations SC01, SC02, and SC07.

DISCUSSION

Presently, several states are evaluating their water quality standards to include narrative or numeric turbidity and/or TSS standards. For example, Georgia has recently enacted a narrative standard for turbidity that is based on “visual contrast in a water body due to man-made activity” (DNR 2000). In addition, Alabama and Florida use 50 and 29 NTU above background, respectively; South Carolina allows a increase of ten percent above background; North Carolina uses 10 NTU for trout streams, 50 NTU for non-trout streams, and 25 NTU for non-trout lakes; Tennessee

uses a standard that does not allow any material effect on fish or aquatic life (Kundell and Rasmussen 1995). Holmbeck-Pelham and Rasmussen (1997) recommended a reduction in average turbidities to below 25 NTU for stream restoration plans in Georgia. In addition, a turbidity of 25 NTU was recommended by the Georgia Board of Regents' Scientific Panel as an instream turbidity standard (Kundell and Rasmussen 1995). Also, the report cited a TSS concentration of 80 mg/l as a threshold between moderate and low levels of protection for fish and aquatic invertebrates (NAS 1972).

FIGURE 5. PEAK TOTAL SEDIMENT LOAD DURING STORM EVENT



Similar findings were observed in this study. TSS concentrations greater than 284 mg/l resulted in biological impairment of macroinvertebrate communities. Also, TSS concentrations of 58 mg/l or less during storm flow provided an adequate margin of safety and were protective of aquatic macroinvertebrates in the Blue Ridge physiography. Furthermore, corresponding turbidity limits of 69 and 22 NTU established the threshold of biological impairment and margin of safety, respectively.

A relationship between TSS and turbidity (NTU) can be developed within a specific hydro-physiography. Turbidity can be used as a surrogate to TSS with the following assumptions and cautions: 1) the relationship between TSS vs. NTU is hydro-physiography specific; 2) turbidity includes inorganic and organic constituents including phyto- and zooplankton which can be extreme during the growing season; and 3) stream discharge and/or stage should be measured at the time of turbidity measurements and compared against a regional regression curve.
A biological endpoint is critical to addressing stream condition and beneficial uses. An index of biological integrity overlaying a sliding, sediment scale (concentration or load) is recommended. Additional surrogates need to be developed and tested between bedload versus embeddedness (MacDonald et al. 1991), bedload versus one-third lower bar (Rosgen 1996), and sediment load versus Pfankuch (1975) or RBP habitat assessments (Plafkin et al. 1989).

The relationship between suspended-sediment concentration and total suspended solids needs to be established for specific physiographies. In addition, in physiographies with high concentrations of clay particle sizes, filtration of the whole sample needs to be explored *in lieu* of withdrawing the supernatant using a J-tube.

The findings of this study emphasize the importance of incorporating aquatic ecological assessments into addressing the effects of accelerated sedimentation and deposition within a watershed. Biological endpoints (e.g., clinger-burrower ratio) can be directly applied to designate beneficial uses such as fishing and recreation. Consequently, comprehensive aquatic ecological studies are a critical component of identifying reference stream reaches and determining whether designated or beneficial uses are being met. Additional research should focus on developing fisheries and aquatic macroinvertebrate indices that are sensitive to impacts caused by accelerated sedimentation.

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