WATER QUALITY SIGNALS FROM RURAL LAND USE AND EXURBANIZATION IN A MOUNTAIN LANDSCAPE: WHAT'S CLEAR AND WHAT'S CONFOUNDED?[1]

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ABSTRACT: In mountainous landscapes with high climatic and geomorphic variability, how do rural land uses and exurbanization alter hydrology and water quality? We evaluated effects of rural land use and exurbanization on streamflows, suspended sediment concentrations and loads, specific conductance, and summer water temperatures in 12 streams and rivers within the Upper Little Tennessee River basin in the southern Appalachian Mountains. Eleven streams featured low levels of development (>61% forest cover) but differed in land use patterning, basin size, annual precipitation, and watershed morphology. One urban stream, located within the largest town in the basin, provided the high development comparative endpoint. Even low levels of rural development and exurbanization were associated with substantial increases in suspended sediment concentrations, sediment loads, and summer stream temperature daily maxima and diurnal variation. Observed summer temperature increases were much larger than would be expected due to global climate change over the next century. Specific conductance was idiosyncratic among the smaller streams. These water quality changes were not accompanied by streamflow changes that were discernible amid the high natural variation in precipitation and geomorphology. The water quality findings suggest the need for applying the best management practices, including riparian buffers, to even low levels of rural development.

(KEY TERMS: urbanization; water quality; sediment transport; temperature; hydrology; best management practices.)

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INTRODUCTION

It is well known that the conversion of relatively undisturbed forests or grasslands to urban cover (lawns, roads, parking lots, and buildings), row crop agriculture, or pasture causes abrupt and dramatic changes to the hydrologic and water quality characteristics of streams (*e.g.*, Booth and Jackson, 1997; Walsh *et al.*, 2005). It is also understood that these responses are affected by spatial patterns of development (Carle *et al.*, 2005; Alberti *et al.*, 2007) and landscape context (Utz *et al.*, 2016). However, we know relatively little about the water quality effects of low levels of forest conversion particularly in mountainous environments (Wenger *et al.*, 2009), although as little as 1% impervious coverage has been associated with declines in some stream macroinvertebrate taxa (King *et al.*, 2011).

In this study we characterize and quantify differences in flow, sediment, specific conductance, and

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stream temperature in 12 stream and river sites within the rural but exurbanizing (Kirk *et al.*, 2012; Gustafson *et al.*, 2014) Upper Little Tennessee River (ULTR) basin in the southern Appalachian Mountains of North Carolina. Nine of these sites were nonnested, independent small watersheds (4-30 km^2). Three larger watersheds included all or some of the smaller sites. These 12 basins incorporated the variability in land cover and development patterns observed in this mostly rural region.

Without human management, the region would support forest cover everywhere except areas of recent disturbance (wind storms, fires, or landslides) and a few ridges of exposed rock. By altering vegetation, soils, runoff processes (Price *et al.*, 2010, 2011), riparian characteristics (Jensen *et al.*, 2014; Jackson *et al.*, 2015), and inputs of chemicals and light (Webster *et al.*, 2012), recent exurbanization within the southern Appalachians (Gragson and Bolstad, 2006; Kirk *et al.*, 2012; Gustafson *et al.*, 2014) is a key force driving spatial and temporal variation in stream water quality (Gardiner *et al.*, 2009; Webster *et al.*, 2012). Exurbanization, sometimes known as amenity-driven development, is the migration of urban residents to rural areas for reasons other than employment (see Vercoe *et al.*, 2014). The resulting conversion of forest lands into second and retirement homes can substantially alter hydrologic processes and water quality. For example, infiltration rates on the lawns and pastures that replace forests are an order of magnitude lower than in forest soils (Price *et al.*, 2010), and the associated loss of dynamic water storage in compacted soils appears to drive lower baseflows in streams at sites with lower levels of forest cover (Price *et al.*, 2011). Relative to forest lands, small row crop fields, unpaved roads and driveways, roadside ditches, and land disturbance mobilize and convey higher quantities of sediments and nutrients to stream systems (Price and Leigh, 2006a, b; Webster *et al.*, 2012). Chemicals introduced to the landscape by fertilization, septic tanks, small chemical spills (*e.g.,* automotive hydrocarbons), or road de-icing increase the ionic strength of groundwater, thus increasing the specific conductance in streamflow (*e.g.,* Conway, 2007).

This research was guided by several hypotheses about how rural land usage, exurbanization, and landscape characteristics would affect aspects of downstream water quality, specifically hydrographs, suspended sediment concentrations and loads, specific conductance, and summer stream temperatures. While we expect even low levels of nonforest land usage to alter hydrology through increased Horton overland flow (Price *et al.*, 2010) and altered storage and release from compacted soils, bare soils, and road surfaces (Price *et al.*, 2011), such changes in this landscape are likely to be small relative to the variation in precipitation and basin morphology among these watersheds. We expect that suspended sediment concentrations, loads, and specific conductivities will increase in streams in watersheds with higher levels of land converted from forest cover (Price and Leigh, 2006a, b) and will increase to a greater degree in basins where such conversion takes place on hillsides. Riparian forest removal is common in residential and agricultural areas in this landscape (Jones *et al.*, 1999; Jensen *et al.*, 2014; Long and Jackson, 2014; Jackson *et al.*, 2015), so we expect summer maximum stream temperatures and diurnal variation to increase with increasing development. Stream temperatures are affected by a variety of landscape characteristics (*e.g.,* Booth *et al.*, 2014), but radiation fluxes, as affected by riparian conditions and channel width, are the dominant drivers of maximum water temperatures (Brown and Krygier, 1970; LeBlanc *et al.*, 1997; Li *et al.*, 2012). We were unsure whether the larger river sites would integrate the water quality signals of the tributaries, and thus feature concentrations, loads, and temperatures intermediate among the tributaries, or whether they would have distinct water quality signatures due to nonlinear scaling of land use effects.

METHODS

To address these hypotheses and predictions, we evaluated water quality variation across the basin by monitoring flow, suspended sediments, specific conductance, and temperature for 18 months at 12 stream locations throughout the basin stratified by land use distributions characteristic of the region (Table 1; Figure 1). The nine smaller basins drained less than 30 km^2. Of these, two basins were nearly fully forested ("forest"); three basins featured forested hillslopes but rural valleys ("valley development"); three basins featured rural valleys but also some amount of residential development on the ridges ("mountainside development"); and one basin was urbanized (within the town of Franklin).

We evaluated scaling issues by monitoring three U.S. Geological Survey (USGS) gage sites in the basin to see if specific conductivities, sediment concentrations, and sediment loads in these receiving waters were reflective of large-scale basin land use or instead reflected legacies of past disturbance. Widespread erosion from land clearing, poor forestry, and hillslope agriculture in the 19th and early 20th Centuries deposited large amounts of sediment on the ULTR floodplains, and the ULTR and its major

TABLE 1. Physical Characteristics, Water and Sediment Yield Estimates, and Land Cover Types of the 12 Studied Watersheds, Grouped by Land Use Pattern.

Stream Code	Stream Name	Site Elev. (m)	Max. Elev. (m)	Drain. Area (ha)	Stream Slope (%)	Chan. Width (m)	Water Yield (mm)	Sediment Yield (tons/ha/yr)	Flashiness Index	Baseflow Index (%)	Land Cover (%)			
											Forest	Ag.	Built	Other
Forest														
BCW9	Ball Creek	689	1,550	720	0.014	9.65	743 (5)	0.16 (11)	0.0091 (6)	82.5	97.4	0.0	2.22	0.42
RAYB	Ray Branch	694	1,626	1,470	0.023	10.15	801 (1)	0.24 (9)	0.0175 (1)	69.0	96.9	0.0	0.56	2.53
Valley Development														
JCKP	Jones Creek	694	1,533	1,560	0.021	11.00	640 (6)	0.16 (11)	0.0115 (4)	81.2	92.1	2.70	2.31	2.91
SSKP	S Fork Skeenah Creek	639	1,114	601	0.010	8.50	744 (4)	0.41 (6)	0.0047 (12)	91.6	84.9	3.24	4.83	7.00
COCR	Cowee Creek	618	1,510	3,012	0.019	11.35	339 (110)	0.32 (7)	0.0070 (8)	89.0	91.2	3.74	2.84	2.27
Mountainside Development														
CACR	Caler Fork	620	1,357	1,738	0.012	10.15	295 (12)	0.26 (8)	0.0069 (9)	91.3	85.1	3.81	3.47	7.66
BUPP	Bates Branch	623	996	382	0.009	8.10	795 (2)	0.74 (2)	0.0133 (2)	81.9	70.0	15.0	7.25	7.84
WAHZ	Watauga Creek	620	1,232	1,669	0.009	9.55	412 (10)	0.45 (5)	0.0125 (3)	85.7	73.3	6.04	14.4	6.35
Urban														
FROG	Crawford Branch	621	886	528	0.019	5.75	456 (9)	0.22 (10)	0.0050 (11)	94.8	29.3	13.9	48.4	8.43
River														
CARG	Cartoogechaye Creek	615	1,661	14,789	—	25[1]	618 (8)	0.59 (4)	0.0089 (7)	73.5	76.1	9.1	7.2	7.7
PRGA	LTenn.R. — Prentiss	612	1,591	36,260	—	35[1]	750 (3)	0.85 (1)	0.0062 (10)	74.0	78.7	7.8	8.0	5.4
NEMG	LTenn.R — Needmore	537	1,661	112,923	—	73[1]	625 (7)	0.60 (3)	0.0110 (5)	74.3	78.5	8.8	7.6	5.1

Notes: amsl, above mean sea level; "Other" Land Cover combines scrub (range 0.1-12.0%), barren (0.0-1.4%), and wetland (0.0-0.4%).
Parenthetic values for water yield, sediment yield, and the flashiness index are ranks among the 12 streams.
[1]Channel width estimated using Google Earth Pro (2015).

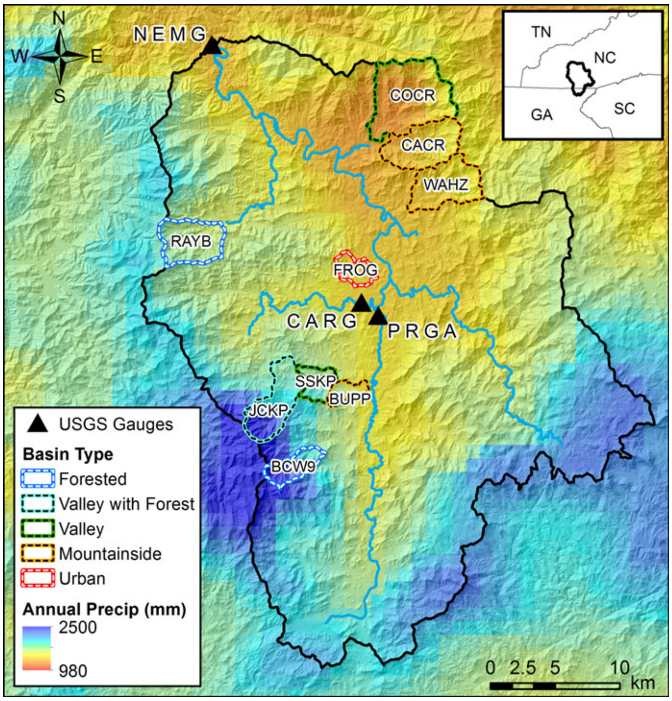

FIGURE 1. Upper Little Tennessee River Basin with Monitored Tributaries and U.S. Geological Survey (USGS) Gage Sites and Average Annual Precipitation (PRISM) Showing a Strong Precipitation Gradient from the Southern and Western Streams to the Northeastern Streams. Stream codes are given in Table 1. Stream basin polygons are color coded by development type. GA, Georgia; NC, North Carolina; TN, Tennessee; SC, South Carolina.

tributaries continue to erode this material from the riverbanks (Leigh, 2016). The three river sites all drain basins larger than 148 km^2 with greater than 76% forest cover.

Study Area

The ULTR basin above the USGS Needmore (NEMG) gage (#03503000) encompasses all study streams and drains 1,130 km^2 of the Blue Ridge ecoregion of the southern Appalachians in western North Carolina and northeastern Georgia (Figure 1). Elevations range from 537 m at NEMG to 1,661 m above mean sea level at the peak of the basin divide. The entire basin lies within the Blue Ridge physiographic province and is underlain by crystalline igneous and metasedimentary rocks, metamorphosed 350-450 mya (Robinson *et al.*, 1992; Wooten *et al.*, 2003). The rocks are well weathered, and a mantle of saprolite several meters to 30 m deep drapes the hillslopes (Hewlett, 1961; Hadley and Goldsmith, 1963; Southworth *et al.*, 2004). The rugged topography features steep, usually forested slopes and relatively flat colluvial and alluvial valleys. All of the smaller streams feature gravel/cobble substrates and slopes of 1-2%, and these streams are thus capable of transporting much of their fine sediment load to the bigger river segments below.

The climate is cool, wet, and highly spatially variable, with average annual rainfall ranging from 2,050 mm/yr in the southwest to 1,350 mm/yr in the northeast (PRISM Climate Group, Figure 1). Due to rugged topography and associated orographic effects on weather patterns, nearby basins in the southern Appalachians may receive substantially different precipitation and feature high within-basin microclimate variability. The North Carolina State Climatologist reports that 30-year precipitation averages in the valleys range from 1,382 mm/yr in Franklin to 1,824 mm/yr at the Coweeta Hydrologic Laboratory, with mean valley air temperatures around 12.7°C (1971-2000; NCSCO, 2014).

Rural residences and small farms are concentrated in the valleys. In the late 1800s and early 1900s, the region was logged and much of the land was converted to agriculture, but pastures and croplands on the steep slopes were subsequently abandoned and then reverted to forest (Gragson and Bolstad, 2006), but not before widespread erosion and valley sedimentation had occurred (Leigh, 2016). The region is now experiencing in-migration and vacation home development driven from the surrounding megapolitan region (Kirk *et al.*, 2012; Gustafson *et al.*, 2014). Newer developments are more likely to be built higher on the mountainsides or ridges where the views are better (Kirk *et al.*, 2012), so these "mountainside development" basins feature high elevation developments above the previously developed valleys. Several factors, including large amounts of federal land ownership, the difficulty of farming or building and maintaining roads on steep slopes, and also cultural norms, contribute to high forest cover (79%) over the entire basin (Table 1). In all but the urban basin, the percentage of built land is below the commonly identified 10% impervious surface threshold for water quality degradation (Schueler *et al.*, 2009).

Watershed land cover data were provided by the Coweeta Long Term Ecological Research (LTER) site. The land cover classification was derived from 2006 NASA Landsat Thematic Mapper imagery classified by Jeffrey Hepinstall-Cymerman and Hunter Allen (Hepinstall-Cymerman, 2011). BCW9 and RAYB were forested sites and represented minimally disturbed hydrologic and water quality conditions for the ULTR. JCKP, SSKP, and COCR watersheds represented classic "valley development" land use patterns. The CACR, BUPP, and WAHZ basins all featured "mountainside development." Overall land cover in the COCR and CACR basins was very similar, but CACR featured a large, high-elevation, ridge top and mountainside development with an extensive gravel road system. This development was typical of those that failed following the 2008 housing market collapse. The FROG basin represented the “Urban” endpoint within the ULTR basin. The three river sites were either located along the Little Tennessee River mainstem (Needmore gage, NEMG, gage #03503000 and Prentiss, PRGA, gage #03500000) or on a major tributary to the mainstem (Cartoogechaye Creek; CARG, gage #03500240), and their watershed areas ranged from 148 to 1,130 km^2.

Flow, Specific Conductance, Temperature, and Channel Width Measurements

The 12 sampling locations were instrumented with a multiprobe datasonde (Hach Hydromet, Loveland, Colorado, model MS5) used to record hourly specific conductance (SpC) and temperature measurements and also a 24-bottle automated water sampler with a submerged water collection intake (Teledyne ISCO, Lincoln, Nebraska, model 6712) (the mention of trade names or commercial products is solely for the benefit of the reader and does not imply recommendation or endorsement of the product). Automated samplers were equipped with a pressure transducer that measured water level every 15 min (Teledyne ISCO, model 720) and datasondes were also programmed to record water temperature hourly.

Site instrumentation with automated samplers began August 16, 2010 and ended October 1, 2011. Specific conductance and temperature recording by the datasondes began March 1, 2010 and ended October 1, 2011. For the nine tributary sites, SpC, temperature, and water level data were recorded simultaneously from September 1, 2010 to October 1, 2011. Stream discharge data for the common dates from the aforementioned USGS gages were used for the river sites. Specific conductance and temperature data were collected at the river sites from January 2011 to October 2011, with the common period of sampling at all river sites being February 22, 2011 to October 1, 2011.

Stage data were converted into flow by developing rating curves by fitting polynomials to periodic manual measurements of varying stages of flow recorded during the sampling period. At RAYB, SSKP, BUPP, and WAHZ, Manning’s equation was used to estimate discharge during full-channel events. For these four sites, estimated bankfull discharges were integrated with the manual measurement data and then the curve was refit. A detailed cross section and channel slope were surveyed at the ISCO site to inform Manning’s equation. At all sites, channel width was determined from 31 measurements of bankfull width measured upstream of the ISCO site at increments of one channel width as estimated by three preliminary measurements.

Total Suspended Solids Measurements

Automated samplers were programmed to collect twenty-four 1,000 mL water samples throughout the duration of five to six storms (Table 2). Because of the variability in storm duration, precipitation amounts, and stream responses, the units were programmed to collect samples at various time intervals during the rising and falling limbs of the storm hydrographs. Prior to collecting each sample, the unit purged and rinsed the intake tubing. Samples were stored in 1,000 mL polyethylene bottles, which were chilled by ice within the unit. When the program had completed, sample bottles were retrieved and transported to the Coweeta Hydrologic Laboratory near Otto, North Carolina for processing and analysis. Approximately 700 mL of each sample was filtered through prerinsed and weighed Whatman GF/C glass 1.5 microfiber filter paper (GE Healthcare Life Sciences, Pittsburgh, Pennsylvania). The filters were then dried at 105°C for 2.5 h and weighed again to determine the solids content (mg/L) of each sample. For quality control, a minimum of three blank filters was also processed with each sample by filtering 200 mL of deionized water through each one and drying as described above.

Data Analysis

Among-site variation was analyzed for flow, water yield, baseflow fraction, flashiness, total suspended solids (TSS), sediment yield, SpC, and temperature as detailed below. Linear regressions were performed using Microsoft Excel 2007 (Microsoft Corporation, Redmond, Washington). All other statistical analyses were conducted using SigmaPlot 11.0 with an *a priori* probability level of $\alpha = 0.05$.

Flow Variation and Water Yield. Flow analyses included: (1) plotting of flow duration curves (FDCs), (2) calculation of flashiness indices (Baker *et al.*, 2004), and (3) calculation of baseflow percentages. All streamflow data were normalized by watershed area to produce mm/unit time. Flow duration curves and flashiness indices were generated from 15 min flow data based on the 13 months of data common to all sites. Stormflow and baseflow were separated using the Lyne and Hollick recursive digital filter (Nathan and McMahon, 1990; Ladson *et al.*, 2013). For this and water yield calculations we used

TABLE 2. Number of Storms, Total Suspended Solids (TSS) Samples, Regression Variables, R^2 Values, and TSS Hysteresis Patterns of the 12 Studied Watersheds.

					Normalized TSS-Flow Power Function			TSS Hysteresis Pattern					
Site	**Storms**	**Total TSS Samples**	**Ave. (St. dev.) (mg/L)**	**Median (mg/L)**	**Constant**	**Exponent**	R^2	**Clockwise**	**Counterclockwise**	**None**	**Figure Eight**	**Complicated**	**Indeterminate**
Forest													
BCW9	6	134	30 (68)	10	1.001	0.701	0.27	3	0	0	1	2	0
RAYB	6	143	31 (71)	8	1.130	0.647	0.43	2	0	1	1	1	1
Valley Development													
JCKP	5	107	45 (76)	14	0.840	0.925	0.50	2	1	0	0	2	0
SSKP	6	119	74 (182)	24	1.121	1.354	0.24	5	0	0	0	1	0
COCR	6	127	54 (101)	24	0.879	1.431	0.43	3	1	0	1	1	0
Mountainside Development													
CACR	6	134	137 (304)	41	1.017	1.915	0.52	5	0	0	0	0	1
BUPP	6	134	147 (356)	40	0.850	1.852	0.57	3	0	0	0	2	1
WAHZ	5	108	154 (331)	43	1.246	1.498	0.55	3	1	0	0	1	0
Urban													
FROG	6	128	131 (255)	40	0.968	2.724	0.53	0	0	0	1	5	0
River													
CARG	5	137	102 (133)	58	0.809	1.042	0.57	0	0	0	3	2	0
PRGA	5	127	131 (153)	58	0.780	1.298	0.77	3	0	0	1	1	0
NEMG	6	144	107 (158)	26	0.796	1.481	0.82	1	0	0	0	4	1

Note: Site codes defined in Table 1.

mean daily flows for one year. We padded each end of the data with 30-day reflection and used three passes (forward, backward, and forward). For each stream, we ran a series of calculations with α values ranging from 0.8 to 0.98. For all streams, the baseflow index (BFI = annual baseflow as a fraction of total annual flow) tended to asymptote at values of α below 0.9, so we used an α value of 0.85 for all streams. We tested regressions of the baseflow index against basin-wide percentages of forest cover and the sum of built, agricultural, and barren lands.

Total Suspended Solids Variation and Sediment Yield. Box plots and one-way analysis of variance (ANOVA) and Holm-Sidak tests were used to compare differences in TSS distributions among the stream and river sampling locations. Normalized rating curves of TSS *vs.* flow were generated for each site by dividing individual TSS and corresponding flow measurements by the median TSS and median flow of the individual stream. Simple linear regressions were fit to the rating curves and visually assessed for trends across the sites. Inspection of the R^2 values indicated that TSS/flow relationships were lowest in the smaller and less-developed basins and tighter in the larger and more developed basins (Table 2). We tested this observation by regression of these R^2 values against log-transformed drainage area and the percentage of built, agricultural, and barren land in the basin.

Sediment yield (t/ha/yr) was estimated using the flow-interval method (Verhoff *et al.*, 1980). Nine flow intervals or "bins" were calculated for each of the sample sites using the minimum and maximum flows. Instantaneous loads were calculated from TSS measurements and their corresponding instantaneous flows. Instantaneous loads were then allocated to their respective bins, with a minimum of three instantaneous loads used in each bin. A small number of instantaneous loads had to be reused in adjacent bins so that at least three instantaneous loads would be used in each bin. The mean instantaneous load from each bin was then calculated. These mean instantaneous loads were allocated to their respective flows throughout the flow record for each stream, standardized to 15-min intervals (to match the discharge records), and then summed to produce a total sediment yield estimation for the sampling period. Forward stepwise regression was used to evaluate which combinations of watershed area, watershed land use variables, and riparian land cover variables best explained sediment yields.

Specific Conductance Variation. Hourly specific conductance measurements were collected over the same time period as the flow measurements associated with the water samples collected by the automated samplers. Distributions of SpC data were plotted to analyze watershed variation during the sampling period. One-way analysis of variance (ANOVA) and Holm-Sidak tests were used to compare differences in the mean SpC of the different sampling locations. Specific conductance *vs.* flow relationships and storm hysteresis loops were examined to evaluate differences among basins and development patterns.

Summer Stream Temperatures. Stream temperature time series were examined graphically to identify a period of relatively consistent elevated summer temperatures from June 10 through July 5, 2010. Maximum and minimum daily temperatures were extracted for each day, and diurnal variation was calculated as the difference between the two. Means, standard deviations, and medians were tabulated for maximum and minimum temperatures and diurnal variations. Differences in averages were assessed using one-way ANOVA and Holm-Sidak tests. We expected that summer maximum temperatures and diurnal variation would be positively associated with drainage area, channel width, % watershed developed, and % riparian zone developed and negatively associated with elevation, % forest cover in riparian zones in the entire watershed, and % forest cover in the 300 m of riparian zone above the gage. We generated a Pearson Product Moment Correlation matrix to examine correlations among response variables and watershed factors expected to affect temperatures. Because of high multicollinearity and low numbers of stream sites, no further statistical analyses of temperature were attempted.

RESULTS

Precipitation and Flow during Study Period

The Coweeta Hydrologic Laboratory recorded 1,875 mm of precipitation during the 13-month study period. Discharges at the Needmore gage averaged 27.3 m^3/s in 2010 and 24.0 m^3/s in 2011, which were 6.5 and 17.8%, respectively, below the long-term annual mean of 29.2 m^3/s (1945-2011) (USGS, 2014).

Flow Duration Curves, Flashiness, and Water Yield

Water yields over the study period varied across the 12 watersheds from as little as 295 mm to as much as 801 mm (Table 1). Watershed location appeared to be the best predictor of flow and water yield. Sites located in the eastern portion of the basin (BCW9, RAYB, JCKP, SSKP, and BUPP; Figure 1) had the highest median flows and greatest water yield during the study period (Table 1, Figure 2). Water yield was not predicted by land use type. Forested watersheds (BCW9 and RAYB) had high water yields compared to other sites, but these subbasins also occurred in the highest precipitation regions. Unit-area flow duration curves and median discharges for the nine tributary sites varied considerably across high, median, and low flows (Figure 2), and the curves did not cluster by land use. The flow duration curves of the river sites were very similar to one another, and their shapes were different from the tributary curves with very low flows in the 100-90% exceedance range and high flows in the 30-1% exceedance range, suggesting the river floodplains alter the storage and release of water relative to tributary inputs. With the exception of one mountainside development site (BUPP), the largest unit-area peak flows (flow-exceedance percentile of $Q_{0.01}$–Q_1) occurred in the forested watersheds. Baker flashiness indices varied by a factor of 3.7, ranging from 0.0047 at SSKP to 0.0175 at RAYB (Table 1), but displayed no systematic relationships with basin land use or basin size. The two forested sites had the first- and sixth-highest flashiness indices. Four of the top six flashiness index values occurred in the high-precipitation western basins. The exceptions were WAHZ, a mountainside development site in the low precipitation region and, ironically, NEMG at the bottom of the ULTR basin. Similarly, the low end of the flow duration curves and the baseflow percentages also showed no apparent patterns related to land use (Figure 2, Table 1). Baseflow percentages were not significantly related to basin-wide percentages of forest cover or developed cover (the sum of the built, agricultural, and barren land percentages). Ironically, the urban stream, FROG, featured the most stable hydrographs, with relatively low peak flows and the highest baseflow percentage. Conversely, one of the forested streams, RAYB, featured the lowest baseflow percentage, and this is a steep stream in a narrow valley draining the highest and steepest watershed. Baseflow percentages for the three river sites were nearly identical and lower than all streams except RAYB.

Total Suspended Solids Concentrations and Sediment Yields

At the 12 sampling locations, 68 automated sampler programs were executed during storms in the 13-month sampling period and 1,542 TSS samples were filtered (Table 2). Storm TSS hysteresis curves were not consistent within or across watersheds and were dominated by clockwise (44%), complicated (32%), and figure eight (12%) patterns (Table 2). Median TSS concentrations varied from as little as 8-10 mg/L in the forested sites to 50-60 mg/L in the river sites CARG and PRGA (Figure 3, Table 2). Total suspended solids concentrations varied

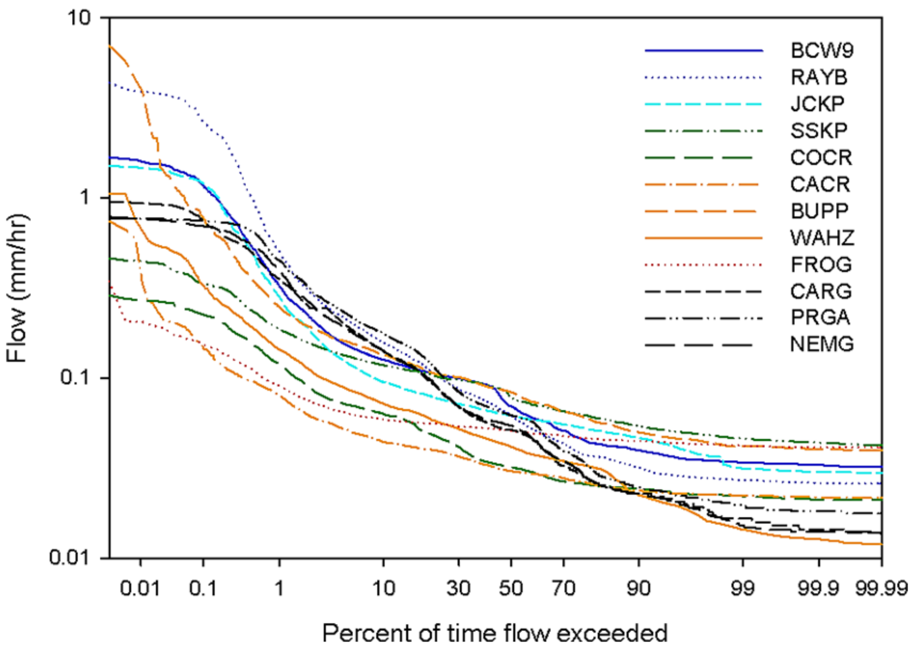

FIGURE 2. Flow Duration Curves for Study Sites during the 13-Month Common Sampling Period. Unexpectedly, the forested streams featured the very high peak flows and also low flows, probably due to the higher precipitation in the south and west sides of the study areas. High flow behavior among the streams seems to track with variability in average annual precipitation (Figure 1).

predictably with watershed land cover patterns (Figure 3). Total suspended solids concentrations in the valley development streams were substantially higher than in the forest streams (median values about twice as large), but these differences were not statistically significant due to high TSS variability in each stream. Total suspended solids values in the mountainside streams and the urban stream were about 4X greater than in the forest streams, and these differences were statistically significant. Total suspended solids concentrations in the river sites featured some of the highest variability, and CARG and PRGA had the highest and second-highest median TSS values across sites. Average TSS values in the river sites were on par with the mountainside and urban streams.

All TSS rating curves indicated strong flow-driven release of solids (Figure 4, largest p-value of 0.026), but the strength of these relationships varied considerably, with R^2 values ranging from a low of 0.27 in forested BCW9 to a high of 0.82 at NEMG. Multiple regression indicated that 72% of the variance in arcsine-transformed R^2 values could be explained by a linear function of log-transformed drainage area (p = 0.001) and the fraction of land that was built, agricultural, or barren (p = 0.053). The slopes of the normalized TSS/Q power relationships appeared strongly related to land use and basin size (Figure 4). For the nine tributary watersheds, the forested sites had the lowest slopes. Slopes were higher in the valley streams, higher still in the mountainside streams, and highest in the urban stream. Slopes of the TSS/Q power relationships at the river sites were intermediate, clustering with the valley sites.

Sediment yields in the nine smaller watersheds ranged from 0.16 (BCW9 and JCKP) to 0.74 tons/ha/yr (BUPP), but eight of the nine tributary watersheds yielded ≤0.45 tons/ha/yr (Table 1). Yields were highest at the river sites (ranging from 0.59 at CARG to 0.85 at PRGA) and also at BUPP (0.74 tons/ha/yr). BUPP was a mountainside development watershed with new housing on one ridge, but it also experienced a large summer thunderstorm during the study that was not significant in the other watersheds. Sediment yields increased as the fraction of built, agricultural, and barren land increased (Figure 5), with the exception of the urban stream, FROG, where the sediment yield was quite low for a watershed with low levels of forest cover. FROG features extensive piping of the stream and runoff network, and thus its hydraulics are much different from the other watersheds. Excluding FROG, 68% of the variance in sediment yields was explained by the fraction of built, agricultural, and barren land in the watershed (p = 0.002).

Specific Conductance

Specific conductance distributions indicated substantial watershed individuality (Figure 6 and Table 3), with the mean value for every site significantly different from every other (p < 0.001). The forested watersheds (BCW9 and RAYB) had the lowest median SpC values (10 and 13 μS/cm), and one

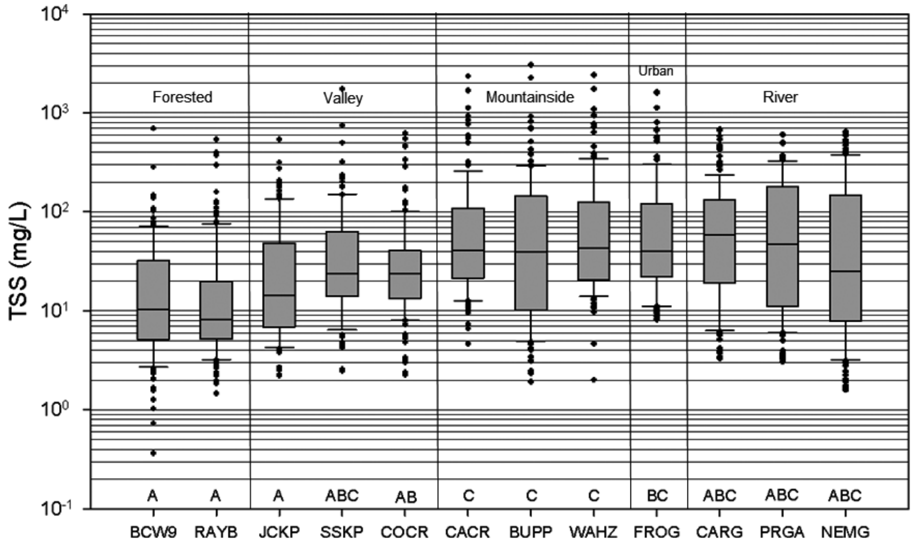

FIGURE 3. Box Plot of Total Suspended Solids Concentrations and Holm-Sidak Test Results from Storm Samples Collected over 18 Months at the Study Streams. Box is defined by 25th and 75th percentiles, solid line in box is median, and whiskers are 10th and 90th percentiles. Storm total suspended solids (TSS) concentrations in the mountainside development streams were very similar to the urban stream and were higher than the valley development streams. Storm TSS values in the valley development streams were generally higher than in forested streams, but the average values were not statistically different. Storm TSS values in the river sites were highly variable, but average values were similar to the mountainside and valley development sites.

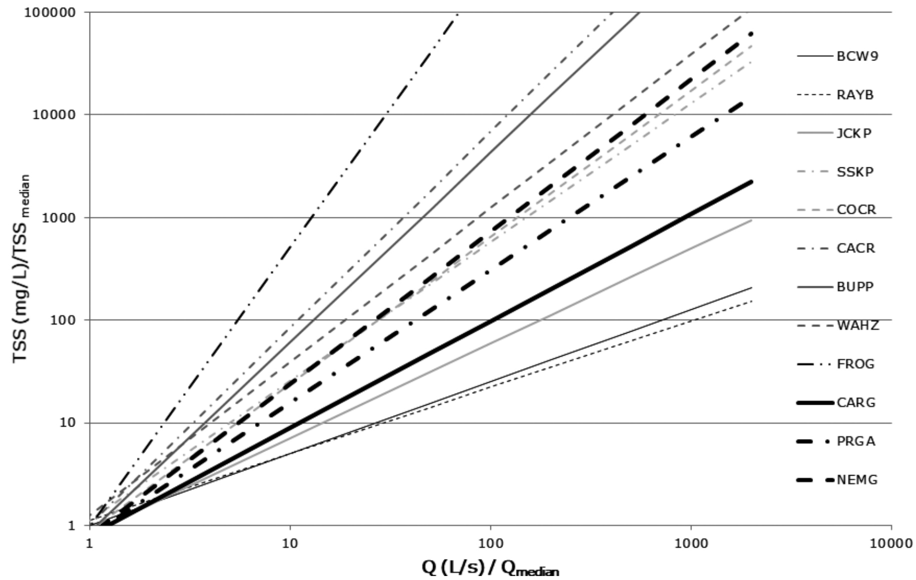

FIGURE 4. Normalized total suspended solids (TSS) *vs.* Discharge Rating Curves for Each Study Site Showing that the Slope of the Curve Increases with Increasing Watershed Development for the Nine Smaller Streams. The larger river systems appear to integrate the basin response, and these curves lie toward the middle of the set of curves from the smaller streams.

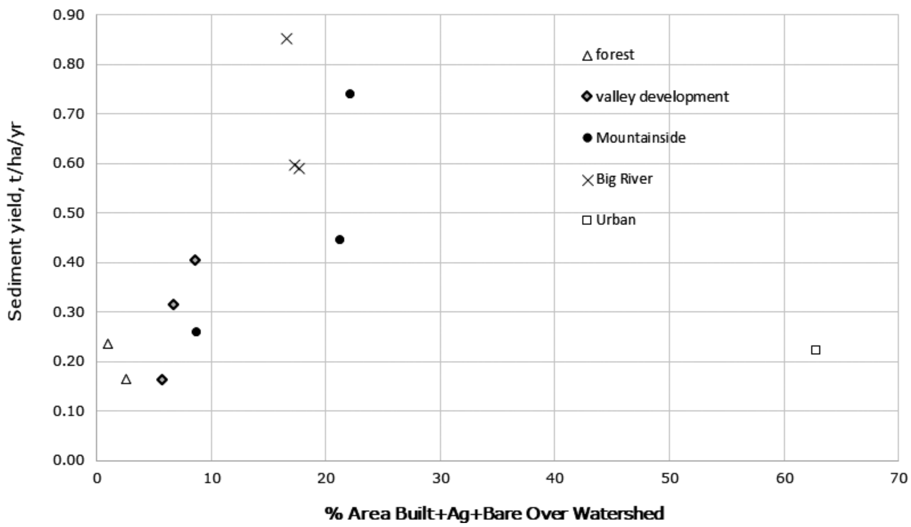

FIGURE 5. Sediment Yield as a Function of the Percentage of Watershed Area Characterized by the Sum of Built Lands, Agriculture, and Bare Soils. Sediment yield from the one urban stream, FROG, is clearly governed by different processes than the rural streams. The river sites feature particularly high sediment yields, indicating sediment sourcing from riverbanks in the main valley (Leigh, 2015).

mountainside development site (WAHZ), the urban site (FROG), and one river site (CARG) had the highest median values of 52, 44, and 42 μS/cm, respectively. All of the less forested sites except COCR featured substantially higher specific conductance values than the forested sites, indicating that human activities are contributing to substantial increases in dissolved ions. The two sites on the Little Tennessee River (PRGA and NEMG) had SpC measurements in between the forested and developed watersheds. Relationships between SpC and flow indicated dilution during stormflows in watersheds with little to no development (valley and forested sites). Watersheds with mountainside development had no relationship between SpC and flow. Conversely, the urban site, FROG, exhibited a dilution relationship, probably resulting from high flow dilution of more constant sewer and septic sources. The river sites also exhibited SpC dilution relationships.

Stream Temperature

Temperature time series in valley and mountain development watersheds and the urban watershed were shifted substantially upward from the forested

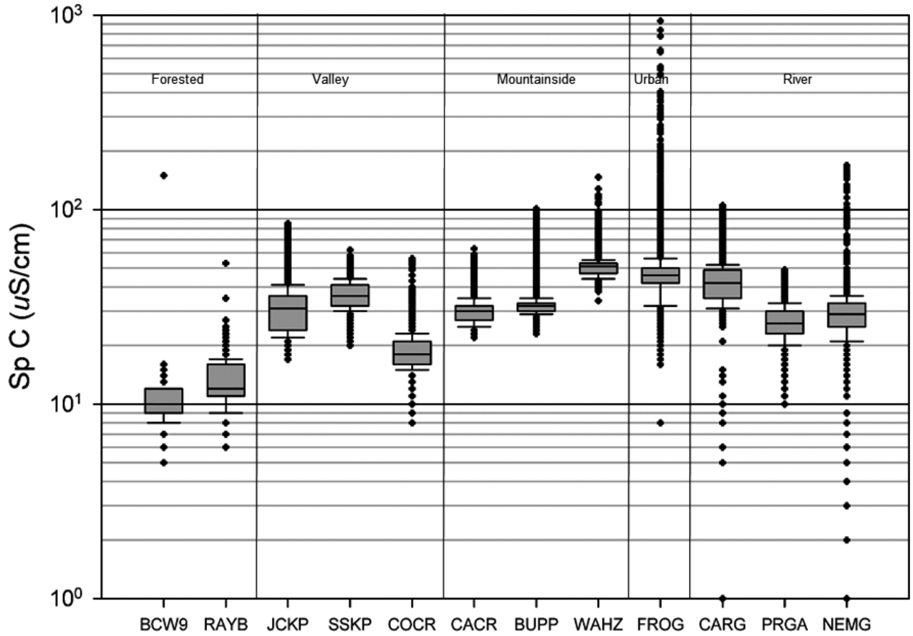

FIGURE 6. Specific Conductance (SpC) Values from All Measurements Collected at Study Sites. Specific conductance was substantially lower in the forested streams, but was highly variable among the streams within each development type, as well as among the larger river sites. Median SpC values are not highly correlated with any land cover statistic.

TABLE 3. Specific Conductance Statistics for Each Site. Mean specific conductances were significantly different among all sites.

Basin Type	Site	Ave. (St. dev.) (μS/cm)	Median (μS/cm)	Relationship w/Flow
Forest				
	BCW9	10.0 (1.6)	10	Dilution
	RAYB	13.6 (3.4)	13	Dilution
Valley Development				
	JCKP	30.7 (8.7)	29	Dilution
	SSKP	36.8 (5.9)	37	Indeterminant
	COCR	18.8 (3.7)	17	Dilution
Mountainside Development				
	CACR	30.2 (3.8)	31	Stasis
	BUPP	33.3 (7.1)	32	Stasis
	WAHZ	52.0 (5.0)	52	Stasis
Urban				
	FROG	44.2 (25)	44	Dilution
River				
	CARG	42.7 (10)	42	Dilution
	PRGA	27.2 (5.2)	27	Dilution
	NEMG	29.2 (5.0)	29	Dilution

watersheds (Figure 7). Maximum temperatures were increased more than minimum temperatures, so diurnal variation was also higher in the developed watersheds. Even in the forested streams, summer daily maximum stream temperatures often exceeded optimal temperatures for coldwater organisms like trout and approached stressful temperatures for such animals, indicating little capacity to absorb additional energy inputs and maintain cold-water habitat.

Average maximum summer stream temperatures in the 12 sampling locations fell into five significantly different groups across a broad range of temperatures (18.7-28.3°C), and these groups closely matched the four development types (Table 4). The forested sites featured the lowest daily maximum summer temperatures of 18.7-18.8°C. The next highest maximum temperatures, 21.5-21.7°C, occurred in the three valley development sites. BUPP, WAHZ, and FROG constituted the next group, 22.7-22.8°C, with two mountain development sites and the urban site. The other mountain development site, CACR, grouped with the river sites CARG and PRGA at 23.7-24.2°C. Maximum temperatures in the largest river site, NEMG, were significantly and substantially higher than all other sites.

The range of differences in average minimum daily temperatures between lowest and highest site was smaller than for maximum temperatures, 6.1 *vs.* 9.6°C, and the minimum temperatures fell on a gradient and did not group as clearly as maximum temperatures (Table 4). The forested sites had the lowest minimum temperatures, the river sites had the highest, and the other watersheds were intermediate and did not group clearly by development pattern.

Diurnal variation in the forested sites averaged 1.07-1.08°C, substantially and significantly less than in all other watersheds (Table 4 and Figure 7). Among the other watersheds, diurnal variation did not group clearly by development pattern. The largest diurnal variations were observed in CACR, a

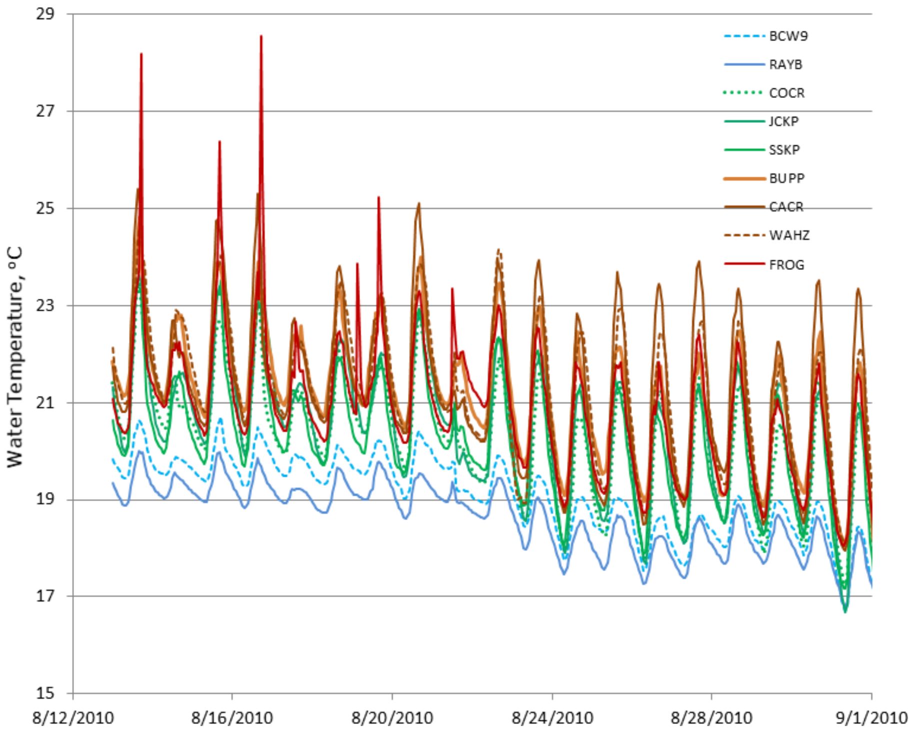

FIGURE 7. Comparison of Subhourly Temperature Time Series for All Nine Smaller Streams from August 13-31, 2010. Highest water temperatures occur in July and August in the study region. The forested streams are generally two degrees cooler than the valley development streams and feature much lower diurnal variation than all other sites. The urban and mountainside development streams feature the highest water temperatures.

TABLE 4. Average Daily Maximum, Minimum, and Diurnal Variation of Stream Temperatures Measured between June 10 and September 5, 2010, the Warmest Part of the Year for Stream Temperatures. Here, they are grouped by temperature similarity, rather than by development type. Note that urban Crawford Branch now groups with mountainside sites Bates and Watauga, while Caler groups with two of the river sites. The largest river site at Needmore gage, sits by itself, the warmest of the 12 streams in the summertime.

Basin Type	BCW9 Forest	RAYB Forest	JCKP Valley	SSKP Valley	COCR Valley	BUPP Mountain	WAHZ Mountain	FROG Urban	CACR Mountain	CARG River	PRGA River	NEMG River
24 h Maximum temperatures												
Average	18.79	18.65	21.68	21.62	21.52	22.71	22.84	22.78	23.83	23.70	24.23	28.32
Std. deviation	0.89	0.89	1.09	1.03	1.07	1.03	1.01	1.73	1.16	1.46	1.47	1.52
(minus forest)			2.96	2.90	2.81	3.99	4.12	4.06	5.11	4.98	5.51	9.60
ANOVA result	A	A	B	B	B	C	C	C	D	D	D	E
24 h Minimum temperatures												
Average	17.71	17.52	18.34	18.35	18.57	19.33	19.21	18.99	19.41	20.58	21.56	23.76
Std. deviation	1.02	1.05	1.37	1.19	1.26	1.27	1.26	0.99	1.22	1.25	1.32	1.27
(minus forest)			0.72	0.74	0.95	1.72	1.59	1.37	1.79	2.97	3.94	6.14
ANOVA result	A	AB	BC	BC	C	C	C	C	C	D	D	E
Diurnal variation												
Average	1.08	1.07	3.38	4.44	2.97	3.26	3.38	1.79	3.79	3.11	2.67	4.56
Std. deviation	0.38	0.43	0.77	1.04	0.71	0.74	0.78	0.82	1.41	1.15	0.56	0.85
(minus forest)			2.24	2.17	1.85	2.27	2.53	2.69	3.32	2.01	1.57	3.46
ANOVA result	A	A	D	D	CD	D	DB	B	E	CD	C	E

mountainside development stream, and NEMG at the bottom of the basin. No pattern relative to land use was suggested for the diurnal variation in the intermediate sites (Table 4).

Quantitative attribution of summer water temperature characteristics to landscape variables was not possible due to the high correlation among many independent variables that affect stream temperature (Table 5). All watershed variables expected to affect stream temperatures except for percent watershed development exhibited high correlations with maximum stream temperature. However, the geomorphic variables of channel width, drainage area (log-transformed), and elevation were all highly correlated with one another (Pearson correlation coefficients ranging from 0.63 to 0.91, and all p-values less than 0.03). Riparian land cover over the whole basin was strongly correlated with maximum stream temperatures, but local riparian forest cover showed the strongest correlations with both maximum temperatures and diurnal variation. The data indicate that both riparian vegetation and network position strongly affect stream temperatures, but the number of sites was too small with respect to the number of highly correlated independent variables to model these relationships.

DISCUSSION

High variability in precipitation and topographic characteristics across the study watersheds drives hydrologic variability that outweighed or masked the effects of hydrologic alteration by human land uses in the study watersheds. Furthermore, storms were not uniform over the basin, and some streams experienced high rainfall events that may have skewed flow and sediment statistics. Therefore, we were not able to discern systematic differences in flow duration curves, streamflow flashiness, or baseflow percentages among the nine smaller watersheds. Neither land use, basin area, nor geographic location alone were predictors of flashiness. In fact, the two forested watersheds had higher unit-area peak flows and flashiness indices than most other watersheds, probably due to higher precipitation, shallower soils, and steeper valleys. One of these forested streams also had the lowest baseflow percentage. Hydrologically, the urban stream was an outlier not because of high peak flows, but due to its stable hydrograph and high baseflow percentage. This is the only basin with a municipal water supply, and we infer that leakage from the distribution system and the sewers is artificially subsidizing low flows (Rosburg *et al.*, 2017).

In the nine smaller watersheds, TSS concentrations varied predictably, with average and median values increasing with watershed disturbance (Figure 3). The slopes of the TSS/flow rating curves increased with watershed disturbance (Figure 4). There were two unexpected findings about sediment concentrations and yields. We did not expect sediment concentrations at the river sites to be as high as those in the mountainside development watersheds, but they were. Nor did we expect the river sites to produce three of the four highest sediment yields. Leigh (2016) showed that historical extractive logging and hillslope agriculture of a century ago led

TABLE 5. Pearson Product Moment Correlation Coefficients for Temperature Response Variables (average maximum daily temperature, Max. T, and diurnal variation, Di. Var.) and the Main Hypothesized Watershed (Wat.) Variables Affecting Temperature.

	Di. Var. °C	Width (m)	Log_{10} D.A.	Elev. (m)	% Wat. Dev.	% Rip. Forest	% Rip. Dev.	% Local Rip. Forest
Max. T. °C	**0.736** (0.0063)	**0.774** (0.0031)	**0.685** (0.0140)	−**0.913** (<0.0000)	0.449 (0.143)	−**0.909** (0.0007)	**0.865** (0.0026)	−**0.972** (<0.0000)
Di. Var. °C		0.326 (0.301)	0.204 (0.525)	−**0.679** (0.0153)	0.359 (0.252)	−0.630 (0.0691)	0.578 (0.103)	−**0.856** (0.0033)
Width (m)			**0.904** (<0.0000)	−**0.712** (0.0094)	−0.0273 (0.933)	0.633 (0.0673)	−**0.740** (0.0228)	0.541 (0.133)
Log_{10} D.A.				−0.632 (0.0275)	−0.111 (0.731)	0.195 (0.615)	−0.331 (0.385)	0.268 (0.486)
Elev. (m)					−0.373 (0.233)	**0.756** (0.0185)	−**0.675** (0.0461)	**0.876** (0.0020)
% Wat. Dev.						−**0.881** (0.0017)	**0.944** (0.0001)	−**0.688** (0.0404)
% Rip. forest							−**0.983** (<0.0000)	**0.882** (0.0016)
% Rip. Dev.								−**0.848** (0.0039)

Notes: D.A., drainage area.
Percent developed (Dev.) in this table is the sum of the percentages of built land, agricultural land, and barren land. Riparian (Rip.) cover was calculated for a corridor within 100 m of the main channel network, and local riparian looked at only the 200 m upstream of the gage. Highlighted correlation coefficients have p-values less than 0.05 and absolute values greater than 0.65. The only hypothesized watershed variable with no high correlations with the response variables is the percent of developed land over the whole watershed.

to deep sediment deposits on the ULTR floodplain and that the current river is actively cutting through those deposits. The high sediment concentrations and yields at the river sites appear to result less from current land use and more from the legacy effects of early forest conversion. Relative rates of floodplain accretion and bank erosion vary along the channel network here (Leigh, 2016), so our data cannot resolve the origin of bank-derived sediments. The second surprise was that sediment loads from the urban stream, Crawford Branch, were well below expectations based on the relationship with land cover and water yield observed in the other watersheds. Crawford Branch has been urbanized for a long time, and much of the stream system is piped or hardened. Finkenbine *et al.* (2000) proposed that urban streams reach a new stable quasi-equilibrium stability with coarser substrates and lower sediment yields 20 years after development. The lower-than-expected sediment yield from Crawford Branch may reflect some combination of post-development equilibrium and channel hardening.

In 1977, the Coweeta Hydrologic Laboratory and LTER experimentally clear cut harvested a mixed hardwood watershed without riparian buffers and with new unpaved roads (built in 1975) (Swank *et al.*, 2001). Sediment yields, mostly contributed from the new roads, averaged about 0.85 t/ha/yr in the first 4 years after road construction, higher than the highest yield from a mountainside watershed evaluated here (BUPP). Most of this sediment was produced in two large storms that occurred shortly after construction (Swank and Webster, 2014). Over the next 10 years, sediment yields dropped to about 0.24 t/ha/yr, equivalent to that observed for the forested stream RAYB (Table 1). This comparison reveals that watersheds with mountainside developments produce sediment yields comparable to those produced by new unpaved logging roads cut into a forested watershed, but with mountainside developments, these high yields are long lasting.

Cross-landscape studies of water quality often use two simplifying assumptions; (1) that land cover (as determined by remote sensing) can be used to characterize land use, and (2) that landowner behavior is similar within a given land use (Arnold and Gibbons, 1996; see Discussion by Cadenasso *et al.*, 2007). However, in many of the study streams, there are only a few dozen streamside landowners, and we have observed that individual landowners engage in specific behaviors that likely have specific water quality effects. Observed examples of such behavior in the study streams include: stream diverted to clean a large dog kennel; streams diverted to ornamental ponds and other landscaping features; trash disposed on steep hillslopes on public lands; disposal of kitchen and yard waste in the stream; mysterious discharges from pipes on the streambanks; landscaping to the waterline; disposal of cow patties on streambanks; backhoeing the channel to "clean it up"; feeding of fish; disposal of animal carcasses; sluicing the flow for gem mining; and various direct physical alterations including wood removal, ornament rock harvest, beach creation, and hydraulic hardening. Some houses were located very close to streams, leading us to wonder if there were sufficient drain field lengths for the septic systems. Such landowner-specific behavior may partially account for the individuality of each watershed with respect to average specific conductance values. Other variables that might explain specific conductivities are the age of development and number of septic systems in each watershed.

In this region, riparian zones within residential or agricultural land uses have been frequently converted into grass, pasture, or single-tree buffers with substantial effects on stream temperature and channel form (Jensen *et al.*, 2014; Long and Jackson, 2014; Jackson *et al.*, 2015). Many landscape factors affect stream temperature (*e.g.*, Booth *et al.*, 2014) including network position (*e.g.,* McDonnell *et al.*, 2015), but solar insolation as affected by riparian vegetation shading is the largest driver of maximum summer temperatures (*e.g.,* LeBlanc *et al.*, 1997; Li *et al.*, 2012). Temperature data from these 12 streams indicated that summer temperature characteristics were affected by both riparian condition and network position, and that the relevant landscape characteristics were highly correlated with one another. Maximum temperatures in the valley sites and mountainside development sites were, respectively, 4 and 5°C warmer than the forested sites. Diurnal temperature variations in the nonforest sites were triple the variation in forested sites. These summer stream temperature conditions render the valleys in the valley and mountainside development watersheds unsuitable as summer habitat for cold-water species, even in watersheds with very low levels of development. Previous researchers have shown that fish assemblages in these valley streams are dominated by cosmopolitan invaders associated with piedmont streams (Jones *et al.*, 1999; Scott and Helfman, 2001), possibly reflecting temperature effects of riparian conversion. Here, the effects of riparian conversion on stream temperature are twice as large as average air temperature increases expected over the next century due to climate change, suggesting that riparian forest restoration could expand summer habitat for cold-water species and ameliorate climate change effects.

Impervious surface coverage has often been used as a surrogate metric for quantifying the degree to which a watershed has been affected by residential and commercial land use (Klein, 1979; Paul and Meyer, 2001; Booth *et al.*, 2002; Miltner *et al.*, 2004), but these results demonstrate significant variability among responses of individual water quality parameters to rural land use actions not captured by developed area alone (Utz *et al.*, 2016). In this exurbanizing landscape, even low levels of development have caused large increases in suspended sediment concentrations and loads and raised summer stream temperatures in valley streams to levels that exclude cold-water species native to the area (Scott *et al.*, 2002). Water quality conditions in the southern Appalachian landscape demonstrate the need for best management practices including riparian buffers (Burcher *et al.*, 2008). Maintaining high levels of forest cover and low levels of impervious surface is not sufficient for protecting stream water quality, as the data indicate that the actions of individual landowners are important at the scale of small streams. Furthermore, the data indicate that water quality is sensitive to the location of development within a watershed (Booth *et al.*, 2002; Carle *et al.*, 2005; Alberti *et al.*, 2007), and that higher elevation development more strongly influences water quality (Webster *et al.*, 2012).

This research, coupled with other water quality investigations in the region (Jones *et al.*, 1999; Scott *et al.*, 2002; Kirsch and Peterson, 2014; Frisch *et al.*, 2016), demonstrates that some water quality issues vary at the reach scale and are highly dependent on landowner-specific land use activities and riparian conditions. For example, behavioral surveys of landowners in this watershed found that 57% of landowners removed riparian trees and 60% actively removed wood from streams to reduce flooding risk and clean the stream. Older landowners were more likely to remove wood from streams and to believe that doing so improved stream health (Evans, 2013). Our observations suggest that individual actors and specific actions matter to stream water quality. Development patterns, riparian management, and implementation of best management practices strongly affect water quality in rural and exurbanizing basins.

CONCLUSIONS

Among these 12 watersheds, neither land cover nor land cover patterns explained the variation in peak flow behavior and flow duration curves. In a region with high variability in precipitation and geomorphology among watersheds, land use effects on hydrology are small with respect to natural controls on hydrologic behavior. The effect of development patterns on hydrology contrasted with the effects on sediment concentrations, sediment loads, and stream temperature, all of which varied systematically with watershed development patterns and/or watershed size.

Even low levels of valley development were associated with significant increases in suspended sediment concentrations, steeper sediment rating curves, and substantial increases in summer maximum stream temperatures and diurnal variation. Mountainside development exacerbated all of these changes. Suspended sediment yield was a function of both water yield (which varied greatly among watersheds due to steep precipitation gradients) and the fraction of basin converted from forest. The river sites exhibited the highest suspended sediment concentrations and three of the four highest sediment yield values, suggesting that bank erosion of previously documented legacy sediment deposits in the river valleys (Leigh, 2016) is contributing to river sediment yields.

Summer stream temperatures in the valley, mountainside, and river sites were approximately 3, 4, and 5°C warmer, respectively, than the forested sites, and their diurnal variations were 2-3X the diurnal variations in the forested sites. These summer temperature increases effectively eliminate the valley stream segments from the summer habitat for cold-water species, and these increases are much larger than the air temperature increases expected over the next century due to global climate change. Riparian forest restoration is key to increasing habitat for cold-water species in the southern Appalachians and to mitigating the effects of climate change on stream ecosystems. The warmest site was the ULTR at the Needmore gage, where the channel is over 70 m wide, and riparian shade is inconsequential. Larger rivers in the southern Appalachians will always be too warm for cold-water species in the summer when tributary habitat must support such species.

Specific conductances were lowest in the forested sites and generally increased with increasing levels of development, but each site featured a statistically distinct specific conductance signature. Each watershed seems to have its own specific conductance story possibly related to specific landowner activities, age of development, proximity and effectiveness of septic systems, and winter road salting. During our stream surveys, we observed numerous idiosyncratic landowner behaviors with water quality ramifications. These water quality observations from the southern Appalachians demonstrate that how land is used can be more important than how much land is used.

Without the application of best management practices, including forested riparian buffers, even low levels of rural and exurbanizing land uses were associated with substantial deterioration of water quality with respect to suspended sediment, stream temperature, and large increases in specific conductance. Development on mountainsides exacerbated these effects out of proportion to the amount of development. These water quality changes were mediated by landscape context and apparently by legacy effects. The high variability in precipitation and geomorphology among these watersheds masked any systematic hydrologic effects due to land use activities.

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