ABSTRACT

GALE, SUSAN MARIE. Linkages between Watershed Characteristics, Hydrologic Responses, and Instream Conditions. (Under the direction of Dr. Ryan Emanuel).

The effect of watershed morphometry and land cover on watershed hydrologic responses and local instream ecological conditions at the watershed outlet were examined for sixty-nine watersheds across North Carolina. Two watershed hydrologic responses—the runoff ratio and stream flashiness— and multiple instream characteristics were significantly correlated with two environmental drivers: mean watershed slope (%) and total impervious area in the watershed (%TIA). Watersheds with a mean slope >10% had low %TIA, slope and runoff ratio were positively correlated, and stream flashiness was consistently low or moderate. Increasing slope was also correlated with improved instream conditions, including lower specific conductance, better instream habitat, less stressed biological communities, and a regulatory use support status of fully supporting. For watersheds with mean slope <10%, %TIA was variable and positively correlated with both the runoff ratio and stream flashiness when TIA was above a threshold of 4%. Similar patterns of response to slope and TIA were identified for water chemistry, biological community health, regulatory stream use support status, and instream habitat when TIA >4%, demonstrating a coupling of reach-level instream conditions and watershed-scale hydrologic function to a single stressor. The 4% TIA threshold is well below the existing regulatory requirements in NC, which requires remediation of stormwater impacts when TIA exceeds 24% TIA. These results suggest that significant degradation of watershed hydrologic responses and instream conditions may occur well below that threshold, and that the focus of stormwater management—minimizing the effects of peak flows—may be insufficient to address other changes that results from

increases in watershed TIA, such as loss of infiltration capacity, groundwater recharge, and evapotranspiration.

Basin morphometry, described using a basin form factor, correlated with stream flashiness and with metrics derived from the watershed width functions, though the strength of the correlations (Spearman ρ) was somewhat low. The results do suggest an existing linkage between watershed morphometry, the structure of the flow path network, and hydrologic responses, as has been proposed in the geomorphic instantaneous unit hydrograph theory. Stream flashiness was more strongly associated with more rounded basins that had width functions that were negatively skewed and exhibited higher kurtosis. The higher flashiness response in these types of watersheds was attributed to the hydrologic routing of a precipitation inputs during storm events through a relatively small number of hydrologic pathways near the watershed outlet over a shorter time period.

These findings have direct impacts on watershed rehabilitation and management, and suggest that mitigating for deleterious impacts to instream conditions due to anthropogenic stressors, such as increased development, must necessarily address the concomitant modifications to the full range of basic hydrologic functions of watersheds.

Linkages between Watershed Characteristics, Hydrologic Responses, and Instream Conditions

by
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A thesis submitted to the Graduate Faculty of
North Carolina State University
in partial fulfillment of the
requirements for the degree of
Master of Science

Natural Resources

Raleigh, North Carolina

2014

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DEDICATION

This is for my Mom, who never understood why any daughter of hers would want to work outside with the bugs and the snakes and the heat, but she loved and supported me anyway. That’s what moms do.

This is for my sister, Janet, who is the one that instilled in me an awe for the natural world. She would let me tag along on her summertime field trips to the neighborhood fields and creeks when we were kids, taught me about the life histories of butterflies and cicadas, and somehow talked our Dad into letting her raise 30 chicks in our suburban basement. What a hoot.

Last but not least, thanks to Mike, who has put up with my stress, whining, lack of sleep and the ensuing bad moods, and all the other negative side effects that go along with being in grad school. He deserves a gold star.

BIOGRAPHY

Susan is originally from the metropolitan DC area. She received a B.S. in Biology from Virginia Tech in 1993, and her exposure to field ecology and aquatic entomology in her time there is what piqued her interest in freshwater systems. She moved to Raleigh, NC in 1998 and within the year began working for the NC Division of Water Quality. She held a series of jobs at DWQ over the next 13 years, with a primary focus on stream and watershed monitoring and assessment, and was fortunate to work with a wide variety of experts in water chemistry, aquatic biological communities, wetland monitoring, botany, fluvial geomorphology, and environmental policy. But one can only dip so many bottles, collect so many bugs, and map so many streams before one looks up to the ridgeline and wonder how the watershed drives the physical, biological, and chemical conditions in the stream where one is standing. This was what inspired Susan to return to school to get her M.S. degree, with a focus on hydrology as a driver for instream ecological conditions.

ACKNOWLEDGMENTS

I would like to acknowledge the technical guidance provided by my committee chair, Dr. Ryan Emanuel, and my committee members, Dr. Stacy Nelson and Dr. James Vose. The NC SAM validation portion of this project would not have been possible without the training, advice, and input provided by the members of the NC Stream Functional Assessment Team (SFAT) who developed the method. Larry Eaton (NC Div. of Water Resources), John Dorney (formerly of NC DWR), and Brad Allen (Atkins Environmental), in particular, provided significant guidance and assistance in applying the method and patiently addressed all of my questions.

Finally, I'd like to acknowledge the professional inspiration provided by H.B.N. Hynes in his seminal 1975 lecture, "The stream and its valley", in which, among other things, he espoused the benefits of aquatic ecologists befriending hydrologists.

We may conclude then that in every respect the valley rules the stream… We must, in fact, not divorce the stream from its valley in our thoughts at any time. If we do, we lose touch with reality.

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CHAPTER 1. INTRODUCTION

1.1 Background

The ecological communities found in lotic systems reflect the long- and short-term interactions of biotic and abiotic factors not only within the local reach, but within the watershed as a whole (Hynes 1975). However, the flow regime for a given system—defined as the magnitude, frequency, duration, timing, and rate of change of stream flow—has been identified as one of the primary drivers of aquatic ecological condition (Karr 1991). What is considered a “natural” flow regime varies from system to system and is scale-dependent (Allan, Erickson, Fay 1997; Vannote et al. 1980), and similar anthropogenic impacts in seemingly similar watersheds will not necessarily produce identical responses (Allan, Erickson, Fay 1997; Poff et al. 1997), may vary regionally (Poff, Bledsoe, Cuhaciyan 2006), or may require multiple indicators of instream ecological condition in order to capture the effects (Clapcott et al. 2012). Even small differences in land use, such as changes in vegetation type from deciduous to evergreen trees, can result in measurable instream changes to flow regimes over time (Swank and Douglass 1974). Certain land uses, particularly agriculture and development, are often associated with a range of deleterious effects on instream conditions and communities, which occur via multiple, response-dependent pathways or cascades involving hydrologic, geomorphic, erosional, and depositional elements (Burcher, Valett, Benfield 2007). Legacy land uses, particularly agriculture, have been shown to have persistent and detectable effects on the current form of the stream network (Jefferson and McGee 2012) and may be a stronger environmental forcing stimulus for instream communities than more recent land uses (Harding et al. 1998).

I have observed a number of cases of aberrant streams in my personal experiences working in NC streams: some streams simply seem to show more resilience to disturbance. This concept of resilience has been offered as one definition of stream function (Norris and Thoms 1999), in that a highly functioning stream will effectively withstand moderate perturbations before reaching the "tipping point" where measurable instream degradation begins to occur. Alternatively, some streams, once disturbed, seem to have a more difficult time recovering from disturbances within their watersheds. But what provides this resilience? The key may be in the structure of the stream network and its relationship to its hillslope, geology, soils, current and past land use, and climate—the unique "heritage" that helped to create its particular form and now dictates how water is transported through the system (Leopold 1964). Addressing this question necessitates inclusion of more than just raw measures of watershed disturbance (e.g., the fraction of developed land cover in a watershed), but also an understanding of the spatial structure of patterns of land use, particularly in relation to stream networks and the watershed outlet (Caylor, Scanlon, Rodriguez-Iturbe 2004; Mejia and Moglen 2010; Turner 1989) and their combined effects over time.

1.2 Watershed structure and the hydrograph response

The natural landscape and stream network of a watershed are inextricably linked. Precipitation produces overland and subsurface runoff of water that modifies the landscape; geology, soils, and vegetation are drivers for the spatial patterns of those modifications, and ultimately they work in concert to self-organize into flow path patterns (Rodriguez-Iturbe

1993). The distance traveled along these flow paths, assuming uniform flow conditions, is proportional to the time it takes for precipitation inputs within the watershed to travel to the outlet as quickflow. Therefore, the time of transfer of water inputs to its instream response, as documented by the stream hydrograph, during a theoretical, homogenous storm event should be predicted by the distribution of flow paths. This distribution provides a description of the watershed topology and is the cornerstone of the geomorphic instantaneous unit hydrograph (GIUH) theory (Rodriguez-Iturbe and Valdes 1979), which has been proposed as an underlying process for unification of various empirically-derived hydrologic principles, such as the mechanisms for Hack's law (Rigon et al. 1996) and Horton's laws (Rinaldo and Rodriguez-Iturbe 1996) and description of the shape of a basin (Rinaldo et al. 1995). It is being investigated as a method for classifying catchments (Moussa 2008) and has been expanded to heterogeneous systems through the examination of the importance of hillslope vs. instream flow (D'Odorico and Rigon 2003); three-dimensional considerations, such as changes in elevation within watersheds (Gupta and Mesa 1988); and impacts of climate on the runoff response (Nippgen et al. 2011).

Watershed modifications, both natural and anthropogenic, can result in measurable changes to the flow network and consequently to the hydrograph response. For example, urbanization can drastically affect the structure of the drainage network, increasing characteristic flow velocities (celerity), decreasing response time, and increasing the magnitude of hydrograph flood peaks, a combination of responses that are generally referred to as stream flashiness (Graf 1977), and create a state of geomorphic disequilibrium (Hayden

1981). Differences in watershed conditions have long been implicated as one reason for the unpredictability and variability of observed flow regimes in streams with similar drainage areas (Sherman 1932). Variation in watershed shapes and hydrologic responses led to the empirical examination of the structure of basins and corresponding hydrographs, in turn leading to development of basic empirical hydrologic "laws" regarding basin morphometry (Hack 1957; Horton 1932) and stream network structure (Horton 1945). Heterogeneity of basins and differences in travel times due to differences in storage within networks was addressed by (Surkan 1969), resulting in an early description of the concept of the digital elevation model (DEM) and its use in modeling flow path network structure within watersheds. The advantage of this method was that it took into account the spatial variability within these systems. One of the most intuitive impacts on travel functions may be differences in land cover, which is just beginning to be addressed in the context of the width function; for example, within urbanized catchments (Ogden et al. 2011).

Many of these efforts involving prediction of the hydrograph focus on single storm events. The instream effects of long-term temporal variation in the hydrograph—the range of all baseflow and stormflow events over time—have not yet been considered in the context of the width function. An additional reach-level indicator that reflects the highly variable and stochastic characteristics of the hydrograph is needed to incorporate this temporal heterogeneity. As described previously, instream ecological condition is a result of the combination of abiotic and biotic factors over time, with the flow regime being a primary driver; therefore, instream ecological conditions may be an excellent response variable that would reflect the unique qualities of an individual watershed and how it transfers its precipitation inputs to its stream discharge outputs.

1.3 Current stream assessment methods

There are numerous methods for stream assessment, each with its own focus. The analysis and interpretation of stream stage and discharge monitoring data are often given primary importance, given its utility in flood prediction. Assessments based on water chemistry gained importance after the passage of the Clean Water Act (33 U.S.C. §1251 et seq., 1972), and acceptable concentrations of contaminants of concern were tied to numerical water quality standards that were developed based on acute and chronic toxicity to aquatic organisms or to human health. Applicable standards are currently determined based on a waterbody's stream classification, which determines its protected uses (NC DENR Division of Water Quality 2012a). For example, all surface waters of the state are protected for the support of aquatic life survival and reproduction but can also be protected for use as a public water supply or for recreation (e.g., swimming). State and federal regulatory agencies periodically review the frequency of exceedences of water quality standards at monitored locations, and make a determination of "supporting" or "impaired" for the waterbody. Impaired streams are often referred to as "303(d)-listed", referring to the section of the Clean Water Act (CWA) that mandates the use assessment process. Since streams can be protected for multiple uses, they can be impaired for one or more uses. Each assessment can be made based on one or more appropriate indicators, such as the ambient concentration of multiple pollutants or the health of aquatic communities. Regulatory agencies base these assessments on monitoring data collected within the previous five years that meet stated data quality criteria, such as the number of samples and sampling/analytical methods.

Geomorphologists use methods such as pebble counts (Wolman 1954) and channel measurements to characterize sediment transport processes. These physical measurements have been extended to instream habitat assessments, which are used by aquatic ecologists to determine the likelihood of successful colonization and continued success of aquatic species, such as fish, insects, or algae. Riparian condition has also been embraced by geomorphologists and aquatic biologists, as adjacent vegetation can promote stream channel stability and benefit instream communities through temperature regulation, carbon inputs, and woody debris. Each assessment method has been developed by different disciplines and necessarily reflects the processes or responses of interest to that field. However, integration of multiple assessments may be required to gain a holistic picture of local condition, although integration of results from multiple assessments is challenging.

Metrics derived from aquatic biological community sampling are commonly used as indicators for chronic and acute stressors in streams. Benthic macroinvertebrates in particular are commonly used due to their relative ease of sampling, their relatively long life spans, and low level of mobility (Merritt, Cummins, Berg 2008). While benthic community assessments are generally used for addressing water quality issues, Merritt, et al (2008) notes that a perceived shortcoming of this use is that the species diversity and sensitivity can be affected by other abiotic factors, such as diversity of habitat (e.g., substrate composition) and food sources, flow regime, and water temperature, and so they are often performed in conjunction with separate habitat assessments or water chemistry measurements to aid in interpretation of the bioassessments (NC DENR Division of Water Quality 2011). In combination, these types of co-assessments can provide a more holistic estimation of instream conditions and has led to wide adoption of rapid bioassessment methods using aquatic invertebrates by federal and state regulatory agencies as part of their aquatic resource assessment and management programs (Barbour et al. 1999). The NC Division of Water Resources (NC DWR, formerly Division of Water Quality) uses a semi-quantitative, multi-habitat sampling method, taxa tolerance values (i.e., the taxon's relative sensitivity to water quality stressors) and relative taxa abundance to calculate community diversity metrics, biotic indices that reflect the level of stream disturbance, and a categorical bioclassification (e.g., Poor, Fair, Good-Fair, Good, Excellent) derived from the other metrics (NC DENR Division of Water Quality 2011). The bioclassification is then used by NC DWR to determine if the assessed stream is supporting or impaired for aquatic life uses as part of the biennial 303(d)/305(b) assessment of the state's waters, as required under the Clean Water Act (NC DENR Division of Water Quality 2012a), with Poor and Fair rated streams considered as impaired.

Most of these previously described methods are time- and cost-intensive and some require extensive training for proper implementation, which limits the extent to which the state's waters can be assessed, and integration of results from multiple methods is difficult. However, a rapid, observational, multi-metric assessment method has been recently developed by state (North Carolina) and federal regulatory agencies, known as the NC Stream Assessment Method (NC SAM). It was developed to provide a single time- and cost-effective tool that is applicable to all streams in NC and capable of measuring multiple functions, including water quality, hydrology, and habitat, and integrates the results into an overall functional rating (NC Stream Functional Assessment Team 2013). It is intended to be integrated into the administration of the state 401 (Water Quality Certification) and federal 404 (Dredge and Fill) regulatory programs in NC (US Army Corps of Engineers 2013), and so in the future it may well become one of the most widely used stream assessment methods in the state. However, a validation of results from the NC SAM method against more commonly accepted methods has not been completed, with the exception of a recent study that focused on stream restoration/mitigation sites (Fernandez 2013) and did not include any non-restored streams. Additional calibration with existing monitoring and assessment data using streams with a range of reach-level and watershed conditions would provide additional assurance to the regulated community that the method is accurate, and would provide additional information on the strengths and sensitivities of this new method to its developers, which may support or suggest future refinements or enhancements.

1.4 Research questions

I believe that examining the watershed land cover, characteristics of the watershed width function, and ecologically-relevant instream indicators will provide additional evidence that basin morphology, topology, and condition are key influences on the hydrologic response of a watershed and consequently on instream ecological health. The relationship between network structure and local instream response will be strengthened through consideration of spatial patterns of land cover.

Individual assessments, such as physical channel condition, habitat, flow regime, water chemistry, or bioassessments, cannot, on their own, provide a full understanding of the biotic and abiotic factors that contribute to instream ecological condition. Multiple assessment methods increase the needed time, expense, and discipline-specific knowledge required for a full assessment, and integration of results is challenging. These factors have increased the interest in development of multi-metric assessment methods that would provide information equivalent to the discipline-specific assessments, and would also synthesize the information into an integrated score or rating that represents the overall ecological condition of the stream. The NC SAM method was developed using this conceptual framework. A validation against currently accepted assessment methods has not yet been completed on a wide range of unrestored streams in NC, but this type of comparison would provide information on NC SAM bias or "trueness". Method accuracy requires both low bias and high precision (American Public Health Association 1998), so a relative level of precision could be determined through repeated assessments at individual sites, preferably under varying field conditions. If NC SAM were shown to provide results comparable to those provided by currently accepted methods, the overall functional rating provided by NC SAM would provide a simpler and more integrated response variable for comparison to watershed conditions and hydrologic responses.

My specific objectives, research questions, and hypotheses were:

Objective 1: Validate NC SAM as an accurate and precise set of variables to measure instream ecological response. Does NC SAM provide results that are comparable to existing methods of stream assessment? Are NC SAM results repeatable under varying field conditions?

Hypothesis 1: If NC SAM is an accurate measure of instream ecological function, it should correlate significantly with results from currently accepted methods of local instream condition, and analogous measures (e.g., NC SAM habitat rating and a separate field habitat assessment) should respond similarly to watershed conditions and hydrologic responses. Results from repeated NC SAM assessments under different seasons and flow regimes will be identical or very similar for the majority of cases.

Objective 2: Evaluate relationships between watershed network structure, watershed condition, watershed hydrologic responses, and local instream responses at the watershed outlet. Do detectable linkages exist between the flow network structure, land cover/land use, hydrologic response, and local instream responses at the watershed outlet? Does the relative location of certain land classifications within a watershed affect the strength of these relationships? Is the relationship more detectable in watersheds with relatively homogenous land use?

Hypothesis 2: If the stream network structure of a watershed is a valid predictor for a stream's hydrologic response, and the flow regime is a significant driver for instream ecological condition, then the stream network structure will have a detectable relationship to one or more reach-level indicators of instream ecological conditions. Land use/land cover affects watershed hydrologic fluxes and responses, so consideration of the spatial relationship between specific land classifications and distance from the watershed outlet will strengthen correlations between watershed condition and instream reach-level indicators.

CHAPTER 2. METHODS

2.1 Overview

Field data collections included North Carolina Stream Assessment Method (NC SAM) assessments, field habitat assessments, and measurements of channel dimensions at 69 field sites within North Carolina (NC). Results from NC SAM were then compared to existing water chemistry monitoring data, existing instream community assessments, the onsite habitat assessments and channel dimensions, and stream discharge characteristics to determine level of concurrence between the different metrics. Residuals analysis was performed to identify sensitivities of NC SAM and other assessment methods to select explanatory variables. NC SAM and other field assessments were repeated in the winter and summer at a subset of locations to examine the precision of NC SAM, which relies heavily on observational data and so may be affected by varying field conditions.

Spatial analysis of watershed–level factors was completed to identify linkages between the watershed's natural characteristics (drainage area, slope, and watershed width function), anthropogenic modifications (impervious surface), and temporal watershed hydrologic responses (runoff ratio and flashiness). These findings were used to guide further analyses of watershed condition and hydrologic responses, and their relationships with instream physical condition, habitat, water chemistry, benthic macroinvertebrate community status, and NC SAM functional ratings.

2.2 Study sites

Sites were selected based on the availability of existing data, including active USGS stream gaging stations, recently collected benthic macroinvertebrate community samples and bioclassifications, and current water chemistry monitoring. Recent (2006-2012) benthos sampling locations were obtained from the NC Division of Water Resources (NC DWR) Biological Assessment Unit monitoring programs and converted to ESRI ArcGIS 9.3 shapefile format. All current USGS NC stream gage locations were also obtained and converted to a shapefile. The two shapefiles were intersected using a 330m buffer and the results were manually reviewed to ensure that sampling locations from both monitoring programs corresponded to the same stream. The resulting list of 69 locations represented sites with recent benthos and current USGS gaging data.

The site list was reviewed further to determine the level of statewide spatial coverage based on region (NC SAM zone and ecoregion), range of bioclassifications, range of drainage areas, and land use (Table 2-1). Sites were predominantly located in the Piedmont (n=44) and Blue Ridge (n=15) ecoregions, and smaller sample sizes for Southeastern Plains (n=7) and Mid-Atlantic Coastal Plain (n=3) (Table 2-1, Figure 2-1). The latter two ecoregions were combined into a single zone, called the Inner Coastal Plain, and is the designation used throughout this project.

Table 2-1 Number of field sites by NC SAM zone, corresponding ecoregion, and latest bioclassification; and range and mean drainage area and land cover types.

NC SAM zone	Ecoregion (Level 3)	n	Number of sites by latest bioclassification: Excellent	Good	Good-Fair	Fair	Poor	DA (km^2) min-max (mean)	% land cover by category min-max (mean): Developed	Forested	Planted/ cultivated
Mountain	Blue Ridge	15	6	6	3	0	0	73-3449 (677)	1-24 (8)	68-99 (85)	0-19 (6)
Piedmont	Piedmont	44	8	8	9	16	3	9-2246 (271)	2-100 (33)	0-92 (48)	0-47 (14)
Inner Coastal Plain	Southeastern Plains	7	3	2	2	0	0	20-5846 (1926)	5-9 (7)	14-62 (43)	0-55 (26)
	Mid-Atlantic Coastal Plain	3	0	0	3	0	0	215-1564 (692)	4-6 (5)	19-41 (30)	24-44 (34)
ALL SITES		69	17	16	17	16	3	9-5846 (554)	1-100 (23)	0-99 (54)	0-55 (14)

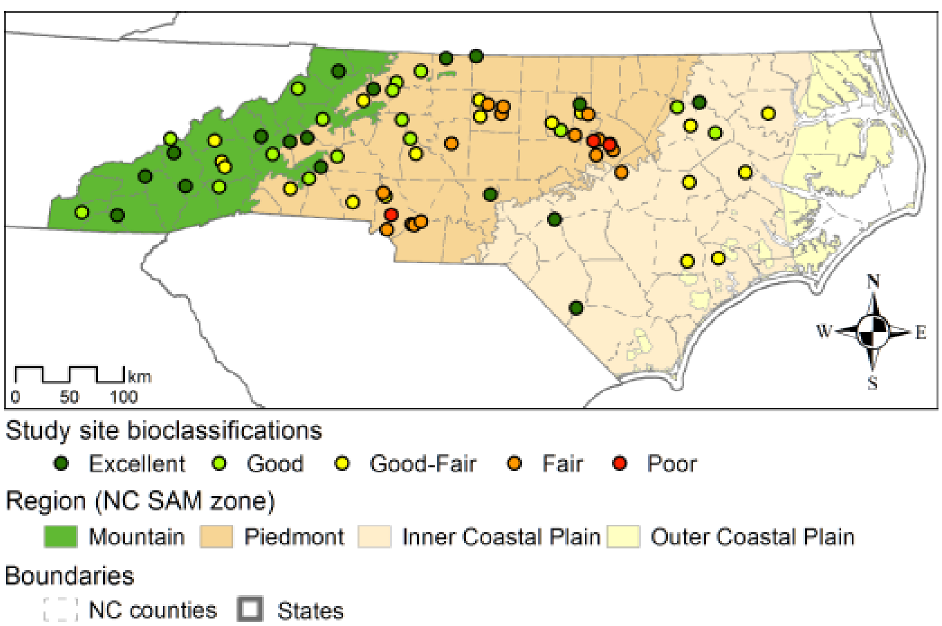


Figure 2-1 Locations and bioclassifications for study field sites.

2.3 Existing data

The selection of appropriate data for NC SAM validation was based on the four main functions defined by NC SAM: water quality, habitat, hydrology, and overall (NC Stream Functional Assessment Team 2013). Data sources included local, state, and federal government surface water monitoring programs (Table 2-2).

Table 2-2 NC SAM functions, existing standards, and sources for existing data.

NC SAM Function	Existing standard	Data source(s)
Habitat	Field habitat assessments	Field assessments
Hydrology	Watershed runoff ratio, stream flashiness	US Geological Survey stream gage data PRISM precipitation data
Water quality	Long-term water chemistry monitoring	NC DWR Ambient Monitoring System NC NPDES Discharger Coalitions Mecklenburg County, NC City of Greensboro, NC USGS NAWQA program
Overall	Benthic macroinvertebrate community bioclassifications	NC DWR Benthic macroinvertebrate community monitoring program

Benthic macroinvertebrate (benthos) data were obtained from the NC Division of Water Resources (NC DWR) (previously Division of Water Quality) web site (http://portal.ncdenr.org/web/wq/benthosdata). The original Excel spreadsheets were compiled into a single file and imported into a custom Microsoft Access 2010 database. The field site list was used to extract the latest sampling event for each field site. Site metadata included stream name, location description (e.g., road crossing), latitude and longitude (decimal degrees), sampling date, sampling method, bioclassification, and biotic indices scores.

Daily mean discharge data for the study sites were obtained for the period of 1/1/1993 through 12/31/2013 by creating a custom USGS instantaneous values web service URL using the REST Web Service Tool (http://waterservices.usgs.gov/rest/DV-Test-Tool.html). Results were downloaded in WaterML 1.1 format, opened in Microsoft Excel 2010, converted to a text file, and then imported into SAS JMP Pro 10.0 for calculation of distributions of stream discharge data for each site for the entire period of 1993-2012. Basic statistical moments describing the distribution shape were calculated, including mean, mode, standard deviation, mean standard error, variance, skewness, and kurtosis.

The T_{Qmean} represents the proportion of measured discharges that exceed the mean value in a given time period. It has been used in a number of studies in the Puget Sound area of Washington State as an indicator of stream flashiness (Greve 2012; Konrad and Booth 2002), with a purported benefit of allowing comparisons between sites with differing data record lengths based on relatively low interannual variability. Low values suggest flashy systems, or discharges of high magnitude but short duration. Higher values suggest more attenuated changes in discharge in response to storm events. Mean discharge for each site was calculated for each year as well as for the entire time period of 1993-2012, and then the proportion (0-1) of mean daily discharge readings exceeding the appropriate mean (annual or 1993-2012) were calculated for each site.

The RBI (Baker et al. 2004) is a modification of the Richards flashiness index, which is a measure of change in discharge taking into account the length of the time step between measurements. The RBI requires data to be reported at uniform increments, e.g., daily, and so simplifies the calculation. It is calculated as:

$$\frac{\sum |Q_i - Q_{i-1}|}{\sum Q_i}$$

Equation 1

where Q_i is the daily mean discharge for day i and Q_{i-1} is daily mean discharge for the previous day. It is essentially a rescaled or standardized mean of the absolute value of daily changes in mean discharge with lower values suggesting a less flashy flow regime. The RBI was calculated annually and for the entire period (1993-2012) for each site.

The runoff ratio or runoff coefficient represents the fraction of precipitation (water inputs) that leaves the watershed as stream discharge at the outlet. It is one of the most basic water balance accounting measures and is widely used for characterizing watersheds in terms of water fluxes (e.g., ET, soil and groundwater infiltration capacities) and storage (e.g., soil moisture). Changes to the runoff ratio in a given watershed can be due to loss of vegetation (and therefore reduced interception and ET), soil disturbance (such as compaction, which decreases infiltration), artificial drainage networks (e.g., storm sewer infrastructure), and increases in impervious surfaces.

Modeled values for U.S. total annual precipitation were obtained from the Oregon State University's PRISM Climate Group website (http://www.prism.oregonstate.edu/) for the calendar years 2002-2012. PRISM data are provided in a GIS-compatible format, greatly simplifying the computation of total precipitation inputs to a watershed and allows consideration of spatial variability of precipitation patterns. Additionally, the PRISM data have been developed through a more sophisticated modeling process than the basic spatial interpolation methods commonly used in the past (e.g., area-weighted average of nearby rain gauge data) that takes into account terrain and landscape features, such as elevation and aspect (Daly et al. 2008). Each year is provided as an individual ASCII file representing total annual precipitation (mm*100) at a resolution of 30 arc-seconds (~800m). Each raster was converted to ESRI GRID format, re-projected, re-sampled, and snapped to match other project rasters (e.g., DEM). A Python script was developed to automate calculation of total precipitation for each year and each watershed in mm. Likewise, USGS daily mean stream discharges (cfs) for each watershed outlet were summed by year and converted to total annual discharge (mm/year). Runoff ratios for each site were then calculated for the entire time period of 2002-2013 by dividing the sum of annual stream discharge by the sum of annual precipitation.

Water chemistry data were obtained for the 10-year period of 1/1/2003 through 12/31/2012 for 62 of the field sites. Data came from several sources: NC DWR (55 sites), USGS (2 sites), and local governments, including the City of Greensboro (2 sites) and Mecklenburg County (3 sites). Each agency's standard operating procedures or quality assurance project plan was reviewed to ensure comparability of analytical methods.

The state monitoring programs included the Ambient Monitoring System (AMS), Lower Neuse Basin Association (LNBA), and Yadkin-PeeDee River Basin Association

(YPDRBA). Data from the state programs were downloaded as text files from the EPA STOrage and RETrieval data management system (STORET) website (http://www.epa.gov/STORET) in accordance with guidelines provided by the NC DWR on their website (http://portal.ncdenr.org/web/wq/storethome). USGS data were obtained from the USGS National Water Information System (NWIS) as text files via the agency website (http://waterdata.usgs.gov/nc/nwis/qw). Local government data were obtained through direct contact of agency staff and provided as Excel spreadsheets. The parameters requested were specific conductance (SC; μS/cm at 25°C), water temperature (°C), pH, dissolved oxygen concentration (DO; mg/L), fecal coliform (FC; colonies/100 mL), total suspended solids or suspended solids concentration (TSS, SSC; mg/L), turbidity (NTU), nitrate + nitrite nitrogen (NO_x; mg/L as N), ammonia nitrogen (NH_3; mg/L as N), total Kjeldahl nitrogen (TKN; mg/L as N), and total phosphorus (TP; mg/L as P). Field measurements (SC, temperature, pH, DO) and TSS/SSC were available for all 62 sites; FC was available for 61 sites; turbidity was available for 60 sites; NO_x, TKN, and TP were available from 53 sites; and NH_3 was available from 51 sites. Where data were available, total nitrogen (TN; mg/L as N) was calculated as TKN + NO_x, total inorganic nitrogen (TIN) as NH_3 + NOx, and total organic nitrogen (TON) as TKN – NH_3. In all cases, non-detects were assigned a value equal to the reporting limit. Summary statistics for each site and parameter were calculated including minimum and maximum date, number of results, number of non-detects, and result percentiles (minimum, 10th, 25th, median, 75th, 90th, and maximum).

Water quality parameters often co-vary so the available parameters were examined for correlations. Spearman correlations identified significant correlations between SC and TN (0.7793; $p < 0.0001$) and TP (0.7149, $p < 0.0001$). It also showed significant correlations with physical measures, such as temperature (0.6911; $p < 0.0001$) and turbidity (0.3512, $p = 0.0069$). SC provided the additional benefit of containing only non-censored data (e.g., all SC results were above the detection limit), unlike the available nutrient data. Based on these correlations, the universal availability of SC from all sites, and lack of non-detect results, SC was selected as a single indicator for chemical contaminants. SC is a commonly used screening measurement for water quality and chemical contaminants, though it can vary widely due to natural conditions, such as geology. Recent work has been done to develop reference values for SC based on concentrations of naturally-occurring ions for U.S. ecoregions (Griffith 2014) and based on benthic macroinvertebrate community bioclassifications for NC ecoregions (Gale 2011). SC was $\log_{10}$-transformed ($\log_{10}(SC)$) prior to analysis due to the wide range of values associated with this parameter. Turbidity was also included in analyses as a water quality indicator due to the weaker correlation with SC and its utility for representing sediment transport processes (Gippel 1995).

2.4 Watershed delineations and watershed land use

The basic data required for performing watershed delineations were Digital Elevation Models (DEMs). While a 10m raster grid has been recommended for catchment-level analysis (Jencso et al. 2009; Zhang and Montgomery 1994), the wide geographic range and large number of field sites for this project limited the data resolution that would feasible for use due to impacts on processing efficiency. Higher resolution raster data (e.g., 1 m, 10 m raster grid cells) would have required more intensive processing power and increased time to complete the analyses. A 30 m resolution raster was selected as an appropriate balance between processing efficiency and landscape detail. This resolution also provided consistency with other GIS data used in this project, such as the Multi-Resolution Land Characteristic Consortium's (MRLC) National Land Classification Database (NLCD) (U.S. Geological Survey 2011), described below.

DEM rasters at a resolution of 1 arc-second (approximately 30m cell size) were obtained from the USGS National Elevation Dataset (NED) (U.S. Geological Survey 2009). NLCD data for the state were obtained in a TIFF format at a resolution of 1 arc-second (approximately 30 m). The DEM and NLCD data were mosaicked as necessary, clipped, re-projected to NC State Plane Meters, and snapped to a common alignment, to produce single rasters for each data type that covered the entire extent of the project area, namely the majority of NC and portions of southern Virginia. Watershed delineations, flow directions, flow path networks, and flow path lengths were calculated in ArcGIS 9.3.1 using ESRI ArcGIS Spatial Analyst tools and the ESRI ArcHydro extension.

The flow path lengths were used to build the frequency distributions (width functions) for each watershed. The width function has been proposed as the underlying mechanism for empirically-derived basin morphometry metrics, such as Horton's bifurcation ratio and Hack's Law. The lack of reliable stream maps for the study sites made application of Horton's laws impractical for this project, but the dimensionless basin form factor (BFF) was easily calculated for each watershed as A/L^2, where A is the watershed area and L is the length of the longest flow path within the watershed. The BFF was derived from Hack's Law (Rodríguez-Iturbe 1997), such that:

$$\frac{A}{L^2} \approx \frac{1}{2} A^{-0.2}$$ **Equation 2**

which suggests that as A increases, A/L^2 decreases, and watersheds elongate as area increases. The BFF was calculated using the GIS-derived drainage areas and the longest flow path for each watershed.

Land use for each watershed was calculated using ESRI ArcGIS zonal statistics tool as the percent contributions to total watershed area for the standard NLCD categories: Water, Developed, Barren, Forest, Shrubland, Herbaceous, Planted/Cultivated, and Wetlands (Fry et al. 2011). Average watershed slope as percent change was also calculated using the Zonal Statistics tool; while channel slope is often used for hydrologic analysis, it was believed that watershed slope would more accurately reflect combined effects of the channels and unchannelized hillslopes.

The site code, land use, and flow path length to the watershed outlet for each pixel (grid cell) in each study watershed were extracted to a text file using the Spatial Analyst Sample tool. Total impervious area (TIA) was calculated for each watershed using the NLCD 2006 raster and the following equation:

$$\sum ((.10*A21+.25*A22+.65*A23+.90*A24)/DA)*100$$ **Equation 3**

where A21, A22, A23, and A24 are subcategories of the NLCD Developed land cover class (open space, low density, medium density, and high density, respectively). The associated weighting factors for each class were obtained from the methods used by the USGS for development of the NLCD imperviousness layer and for calculating watershed TIA using the online StreamStats application (Fry et al. 2011). The broader “Developed” land cover class was therefore associated with a wide range of TIA (10-90%).

To understand the organization of land cover with respect to flow path lengths, a Python script for the Monte Carlo (MC) simulation was developed using the SciPy 0.13.0, NumPy v. 1.7.1, and pandas 0.13.0 libraries (McKinney 2010; McKinney 2013; Oliphant 2007) and run using iPython (Perez and Granger 2007). The mean, kurtosis, skewness, and 25^{th} and 75^{th} percentiles for flow path lengths to the outlet were calculated using the text file previously extracted from GIS. Statistics were calculated for each land use class within each site for the original conditions (run 0) of the MC simulation. The flow path lengths were then randomly shuffled 4999 times and the summary statistics of the flowpaths for each land use class were re-calculated for each iteration to build a frequency distribution of values for each statistic. The final data set of summary stats (initial conditions and 4,999 randomizations) for each watershed and land use class were output as a text file for further analysis in SAS JMP Pro.

2.5 Additional watershed and assessment site attributes

The majority of the additional watershed characteristics that are described in this section were specifically required by the NC SAM assessment methodology and included regulatory considerations and geographic or physical conditions of the watershed. Regulatory considerations included current NC DWR use assessment status (i.e., supporting or impaired as per the latest 303(d)/305(b) assessment (NC DENR Division of Water Quality 2012b)), NC DWR stream classification (e.g., C, Water Supply) and supplemental classifications (e.g., Trout), and presence of threatened/endangered aquatic species. Stream use support status and stream classifications were determined by snapping the shapefile of field sampling points to the closest stream feature in the shapefile representing the latest available NC DWR water quality assessments (NC DENR Division of Water Quality 2012a). The stream name from the project site list and the results of the intersection were compared to ensure that the point was snapped to the correct waterbody, and corrected if necessary. The use assessment protocol assigned a category to each stream reach ranging from 1-5. If a stream reach was assigned Category 4 or 5, it was considered impaired. Both of these categories represent waterbodies that are considered impaired based on existing monitoring data; the difference is whether or not a Total Maximum Daily Load (TMDL) is required or not (NC DENR Division of Water Quality 2012b) as per U.S. EPA regulations. Sites assigned to categories 1-3 were considered supporting their designated uses. Approximately half (50.8%) of the study sites were listed as impaired, and several were impaired based on two or more criteria. Reasons for impairments included violations of associated water quality standards for copper (5 sites), fecal coliform (6 sites), low pH (6 sites), nitrate nitrogen (1 site), PCBs (3 sites), turbidity (12 sites), water column mercury (2 sites), and zinc (3 sites). Sites were also listed as impaired based on the integrity of biological communities, including benthic macroinvertebrates (16 sites) and fish (5 sites).

The determination of the presence of threatened and endangered species was done through the NC Natural Heritage Program online map utility (NC DENR, Natural Heritage Program 2013) using the latitude and longitude for each site and the minimum available search radius (1 mile).

Geographic attributes that were required by the NC SAM method included ecoregions, county, and river basin, and these were obtained in ArcGIS by intersecting readily available GIS data products and assessment site locations. Ecoregions were derived from the Level IV ecoregion shapefile (US Environmental Protection Agency 2012). County and river basin were derived from GIS data files from NC agencies (NC Geodetic Survey 2012; NC North Carolina Natural Resources Conservation Service State Office 2013).

2.6 Field assessments

Field data collections occurred during three distinct sampling seasons: summer 2012 (preliminary; 15 sites), February-March 2013 (winter 2013; 13 sites), and summer 2013 (summer 2013; 67 sites). The summer seasons were defined as the period between May 1 and September 30, which is consistent with the standard sampling time frame used by the NC DWR bioassessment program (NC DENR Division of Water Quality 2011).

The sampling methods described below were consistent across all seasons. Not all data were collected during every field visit: in cases where the stream was not wadeable due to depth or high discharge certain assessments were either omitted (field habitat assessments) or estimated (channel dimensions). All data were entered into a Microsoft Access database for warehousing.

Selection of the specific assessment reach during field visits was based on local conditions. Where possible, reaches were located upstream of bridges to minimize effects that are often associated with these features, such as scour pools, rip rap, and stormwater inputs. A riffle/run and pool/glide section were included if both features were present; these are the types of areas that are preferred for NC SAM assessments (NC Stream Functional Assessment Team 2013) and are also the areas targeted for benthic macroinvertebrate collections and habitat assessments (NC DENR Division of Water Quality 2011). A specific reach length was not required, though a target length of three times the stream width was used as a guideline. Assessment reaches were selected such that they did not contain any tributaries confluences, point source dischargers, and had fairly consistent riparian conditions throughout, in accordance with the NC SAM method. Once the assessment reach was identified, the appropriate measurements or estimations were recorded on a Site Information Sheet developed for the project. General information included date of visit; coordinates of upper and lower end of assessment reach, length of assessment reach, and the mean, minimum, and maximum water depth within assessment reach. Coordinates were determined using a Garmin eTrex20 recreational grade GPS receiver (< 3m accuracy with WAAS).

The channel dimensions were measured at one riffle section and included width at bankfull (BF) and top of bank (TB), and height from the bottom of the thalweg to BF and TB elevations. It should be noted that many researchers use BF and TB interchangeably in referring to the stage at which a stream overflows its banks and accesses its floodplain, which occurs at a return interval of approximately 1-2 years (Leopold 1964). While this assumption is appropriate for relatively undisturbed and/or small streams, stream incision and relict floodplains are common in NC and have been in some areas since at least the early 20th century (e.g., Perkins et al. 1924). In such cases, the wide, flat area abutting a stream or river is more correctly referred to as a terrace rather than a functioning floodplain since it is rarely accessed by the stream. Instead, the BF discharge will occur at a lower stage and the stream will often widen and develop a new, accessible floodplain at the lower BF elevation and within the confines of the incised channel (Rosgen 1994). In these cases, field indicators of bankfull stage include features such as the back of benches or bars. These are the indicators that were used to identify BF elevation for this project, and TB corresponded to the elevation of the terrace/relict floodplain.

Water quality measures included conductivity (μS/cm) and water temperature (°C), which were recorded using an Oakton dual range ECTestr meter. The meter was calibrated weekly at room temperature using a two-point calibration method with distilled water (0 μS/cm) and purchased standard (447 μS/cm). Field conductivity readings were later converted to specific conductance (μS/cm at 25°C) using the conversion equation described in the USGS field manual (Radtke, Davis, Wilde 2005):

$$C_{25} = \frac{C_m}{1 + 0.0191(t_m - 25)}$$ **Equation 4**

where C_{25} is the conductivity value corrected to 25°C (µS/cm at 25°C), C_m is the field measured conductivity (µS/cm), and t_m is the water temperature (°C) at the time of conductivity measurement.

A qualitative survey of aquatic organisms was completed as per the NC SAM methodology (NC Stream Functional Assessment Team 2013), including kick samples in a riffle area using a D-frame net; net sweeps of undercut banks, root mats, and in pools; and visual inspection of large cobble, boulders, leaf/stick packs, and logs. Aquatic insects were identified to family on-site and representative specimens for each taxon were preserved in 70% ethanol or isopropanol as voucher specimens. The relative abundance (1; >1) for each insect taxon and for non-insect taxa (crayfish, fish, salamanders, tadpoles) was noted.

Instream and riparian habitat were assessed using the appropriate NC DWR habitat assessment based on the ecoregion where the assessment site was located (Mountain/Piedmont or Coastal Plain), in accordance with NC DWR standard operating procedures (NC DENR Division of Water Quality 2011), which provided a numerical score between 1-100, with better habitat conditions associated with the higher scores. As noted previously, this assessment was omitted for if the stream was not wadeable at the time of the field visit as key features (e.g., bed substrate) could not be assessed without access to the stream bed. Instream physical conditions considered by the field habitat assessment included substrate composition; variety of instream habitat available for fish and macrobenthos colonization; longitudinal pattern (riffle-pool sequences); channel modification (e.g., straightening); and bank stability. Additional measures included canopy cover provided by riparian vegetation and the extent and condition of the riparian vegetation. Riparian vegetation contributes to bank stability and is often considered protective of instream biota by regulating water temperature, serving as a source of carbon inputs, and providing local moderation of quickflow from the surrounding uplands. However, these are fairly localized impacts, and the primary focus of this project was the effect of watershed conditions on local instream conditions, so these factors were excluded from the habitat assessment score calculation for the majority of analyses. After riparian vegetation points were excluded, the habitat assessment score was re-standardized as a percent of the total possible points (80) using only instream measures.

The NC SAM field assessment was completed based on the data and observations previously described. Answers were recorded on the NC SAM Field Assessment Form and later entered into the NC SAM calculator (a custom Microsoft Excel-based application developed by the Stream Functional Assessment Team (SFAT), which converted field observations to final functional ratings using the NC SAM model and provided a single page report for each site.

CHAPTER 3. RESULTS AND DISCUSSION

3.1 NC SAM validation

This section provides a summary of significant findings from the comparison of results of 98 NC SAM assessments from 70 sites to stream use support status, bioclassifications, field habitat assessments, stream flashiness, watershed runoff ratio, and water chemistry as part of the NC SAM method validation, to address *Objective 1* of the study *Research Questions* (Section 1.4). An analysis of replicate NC SAM analyses was also performed to assess method precision. Details of these results are also provided Appendix C.

NC SAM results were visually skewed towards high values for the overall function and for the three subfunctions (habitat, hydrology, and water quality); a paucity of low- and medium-rated sites made statistical analysis impractical in some cases and limited analysis to comparisons of descriptive statistics and qualitative assessments for the NC SAM rating categories. The predominance of high-rated sites suggested that NC SAM may not be appropriately sensitive to conditions that would lead regulatory agencies to determine a stream is not supporting its designated uses. As noted previously (Methods, Section 2.5), more than half of the study sites were considered impaired due to exceedences of water quality standards or stressed biological communities. The percentage of these impaired sites that received a low rating by NC SAM were 29% (overall function), 46% (habitat and hydrology subfunctions), and 14% (water quality subfunction), and only 19 of the 34 officially impaired sites (56%) received a low rating for one or more subfunctions. The combination of predominantly high ratings and low rate of detection of stream impairment suggests that NC SAM was unable to identify water quality or habitat issues that were significant enough to cause use impairment. In several cases, information was provided on the NC SAM assessment form to indicate the presence of point source discharges, extremely high specific conductance at the time of assessment, high levels of developed land use in the watershed, and documented evidence of water quality impairment, but this information appeared to be under-weighted by the NC SAM model and resulted in functional ratings that were higher than the functioning level suggested by results from existing data or observation of onsite conditions. NC SAM functional ratings did show similar responses to watershed conditions such as land use, drainage area, and slope, as other measures (bioclassification, habitat assessment, water chemistry), with the exception of the NC SAM habitat function, and the exceptions as noted above.

Results from the repeated NC SAM assessments under varying seasons and conditions showed the importance of training and experience in using the method, as concurrence rates between repeated assessments increased and inter-assessment variability decreased over time. The concurrence rate of NC SAM was approximately 75% after additional training occurred in November 2012, suggesting that repeated training and experience were necessary to attain an acceptable level of repeatability. For the 25% of repeated assessments that did not show perfect agreement, differences in the NC SAM scores averaged 3.4 NC SAM rank units (with rank having a scale of 1-10), which was an improvement over differences in earlier repeated assessments, which averaged 4.7 rank units.

No seasonality effects were identified, suggesting that factors such as stream stage or discharge do not affect the precision of the method once the assessor is fully trained.

In many cases, NC SAM was found to be a promising tool for screening-level or similar studies (e.g., watershed characterizations), and in many cases will provide a rapid, cost-effective assessment. The observational nature of the method contributes to the speed of the assessments, though this advantage is countered by the extensive training and experience that is required for precise application of the method. Of primary concern were the site-specific issues that were noted at a number of study sites, which were not adequately captured by NC SAM. In some cases, local conditions, such as forested land cover at the assessment reach, appeared to outweigh watershed conditions, such as high TIA, or local conditions, such as extremely elevated specific conductance. NC SAM was developed primarily as a tool for site-specific assessments, with the results being tied to regulatory activities, such as compensatory mitigation ratios for stream impacts, with higher functioning streams requiring a higher ratio than lower functioning streams (US Army Corps of Engineers 2013). In this study, sites often received a higher NC SAM functional rating than would be suggested by other monitoring data or use support status, suggesting that the regulated community will be required, in some cases, to meet higher mitigation ratios than necessary. However, time- and cost-intensive monitoring of streams for definitive determination of stream function are unrealistic given the staff, funding, and time constraints of regulatory agencies and the regulated community. Any rapid assessment method will necessarily require a balance between acceptable accuracy and efficiency.

Accuracy of NC SAM as compared to instream conditions could be improved by greater weighting by the model of some of the information and observations already provided during assessments. It is believed that accuracy could also be improved by consideration of watershed conditions rather than simply local conditions; the only watershed condition considered by NC SAM is watershed TIA, and this variable is linked to stormwater regulatory programs' threshold of 24%. Further analysis of NC SAM, other local instream indicators, watershed hydrologic responses, and watershed structure and condition were completed to determine which watershed factors may be the strongest environmental drivers for local instream conditions and therefore deserve strong consideration in any future refinements to the NC SAM method.

3.2 Watershed hydrologic responses, topography, and land cover

Two basic watershed hydrologic responses (runoff ratio and stream flashiness) were compared to watershed slope and land cover to address, in part, *Objective 2* of the study *Research Questions* (Section 1.4). However, an initial examination of the two measures of flashiness used in this study, the RBI and T_{Qmean} was completed. The latter has been proposed to be relatively insensitive to interannual variability (Greve 2012; Konrad and Booth 2002), thereby allowing comparisons between sites with differing periods of record. To test this assertion, I compared the coefficient of variation (CV) for annual RBI and T_{Qmean} for all study sites using the non-parametric Wilcoxon signed rank test. No significant difference in CV values was found between the two different measures, suggesting that the T_{Qmean} will not provide the purported benefits over the RBI for this data set. Both measures were included in analyses to determine if there were differences in the relative sensitivity of each measure to watershed conditions.

Examination of scatterplots of stream flashiness and runoff ratios suggested a non-monotonic relationship (Figure 3-1) where high runoff ratios were observed in watersheds having both low flashiness (low RBI or high T_{Qmean}) and high flashiness (high RBI or low T_{Qmean}). Regional differences were also evident: Mountain (M) sites clustered by low flashiness/high runoff ratio, Piedmont (P) sites showed a wide range of both values, and the Inner Coastal Plain (ICP) sites exhibited low runoff ratios and moderate flashiness values.

Ecoregion is often used to control for natural variations in instream conditions due to in geology, climate, soils, and other physical factors. For example, the method for determining benthic macroinvertebrate bioclassifications based on biotic indices and metrics varies in NC based on the ecoregional location of the sampling site (NC DENR Division of Water Quality 2011). However, a more accurate control for natural regional variability would be one that considered the variability within the contributing watershed, would reflect the interactions between the physical and climatological conditions, and would vary continuously across the landscape. A characteristic meeting these criteria is mean watershed slope, which is also meaningful in terms of watershed hydrodynamics as it is a contributor to differences in energy gradients on the landscape that drive surface and subsurface flow.

The scatterplots shown in Figure 3-1 were modified so that the marker color indicated the associated watershed slope (Figure 3-2) and total impervious area (TIA) (Figure 3-3).

High runoff ratios were observed in watersheds with a combination of high slope/low TIA/low flashiness and in watersheds with low slope/high TIA/high flashiness.

Comparison of Figure 3-2 and Figure 3-3 suggested that the non-monotonic relationship between the runoff ratio and stream flashiness is due to the interactions between watershed slope and land cover, acting as two separate drivers on runoff generation and instream flashiness. Decreases in watershed slope, all of which were associated with relatively low TIA values, resulted in apparent decreases in runoff ratio and an apparent increase in stream flashiness. The trend was reversed at very low slope values, and the runoff ratio began increasing as stream flashiness continued increasing. However, these increases appeared tied to increases in TIA in these specific watersheds. For sites where slope and TIA were low, the runoff ratios were also low. An examination of TIA and slope indicated that high levels of TIA were primarily associated with a relatively narrow range of watershed slopes, primarily in the Piedmont (Figure 3-4).

These findings suggested that slope provides a significant control on watershed hydrologic responses, but that land cover may override that influence once a certain threshold is crossed. A direct comparison of slope and TIA versus runoff ratio and flashiness (Figure 3-5) suggested that once the threshold was exceeded, more predictable relationships appeared.

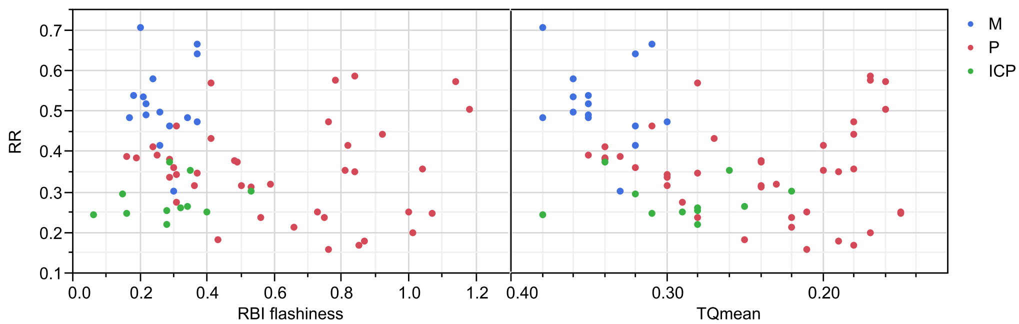


Figure 3-1 Scatterplots of runoff ratio (RR) versus two flashiness indices: Richards-Baker Index (RBI) and T_{Qmean}. Marker color denotes the region of NC, Mountains (M), Piedmont (P), or Inner Coastal Plain (ICP).

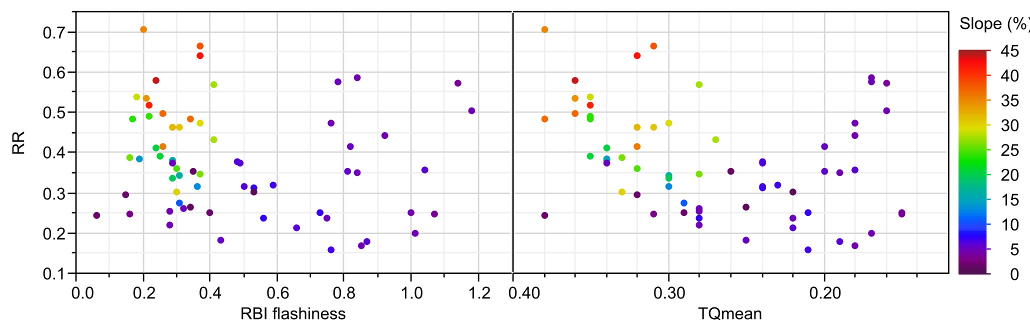


Figure 3-2 Scatterplots of runoff ratio (RR) versus flashiness indices (RBI and T_{Qmean}). Marker color denotes average watershed slope (%).

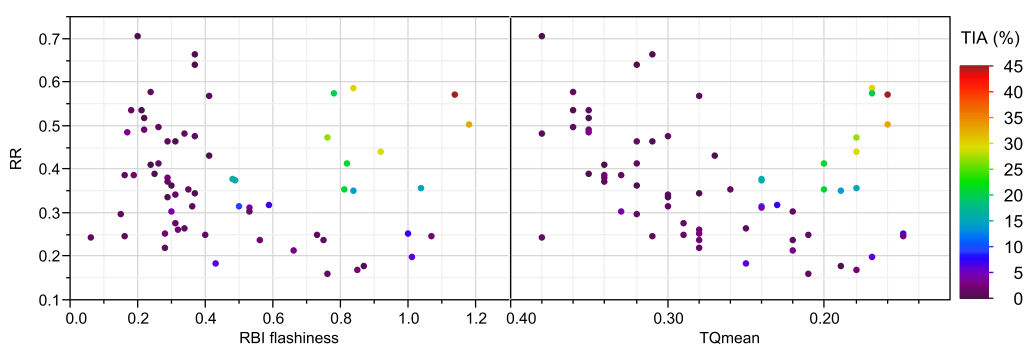


Figure 3-3 Scatterplots of runoff ratio (RR) versus flashiness indices (RBI and T_{Qmean}). Marker color denotes watershed TIA (%).

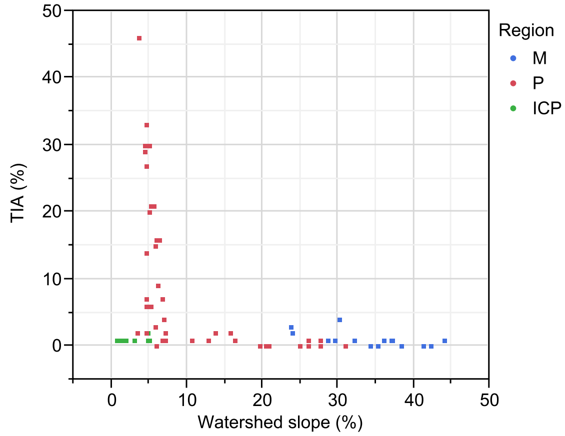


Figure 3-4 TIA (%) versus watershed slope (%). Marker color indicates region where sampling site was located.

A threshold of 10% slope was selected based on a visual assessment of the scatterplot. All M sites were above this threshold, ICP sites were below, and P sites displayed a gradient between the two extremes of M and ICP. There was a lack of significant correlations between slope and watershed hydrologic responses when slope was <10% (low slope), suggesting that in flatter terrain, slope was not a significant driver. For sites with watershed slope >10% (high slope), there was a significant positive correlation between slope and runoff ratio (Spearman's $\rho = 0.735$, $p < 0.0001$), which suggested that increases in slope result in increases in runoff generation. There was no significant correlation between RBI flashiness and slope in the high slope watersheds, and only a marginal correlation between T_{Qmean} and slope ($\rho = 0.360$, $p = 0.055$). Streams were significantly less flashy in high slope watersheds than in low slope watersheds ($p < 0.0001$ in Wilcoxon rank sum tests for both RBI and $T_{Qmean;}$ RBI medians: 0.29 for high slope, 0.66 for low slope; T_{Qmean} medians: 0.33 for high slope, 0.22 for low slope). High slopes may provide a physicallybased upper boundary that mediates the stream flashiness response, or alternatively, the impact of slope on stream flashiness may be masked by other factors, such as TIA.

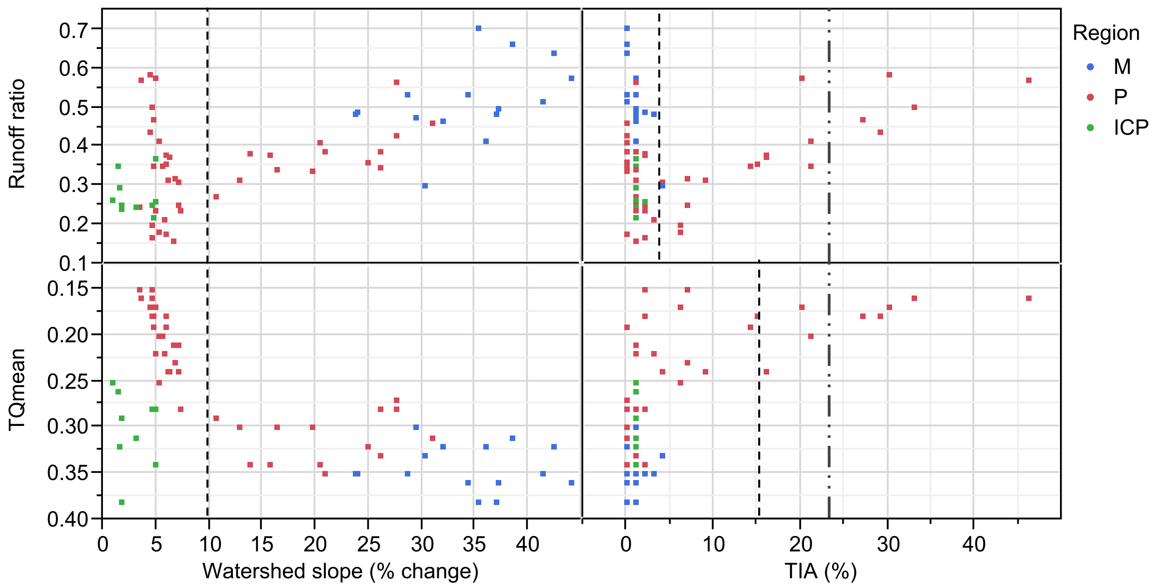


Figure 3-5 Scatterplots of watershed slope (%) and TIA (percent) versus runoff ratio and T_{Qmean}. Marker color denotes region. Dashed lines indicate identified thresholds for slope (10%) and TIA (4% for runoff ratio, 16% for flashiness). The dashed/dotted line indicates the 24% TIA threshold used by stormwater regulatory programs.

The threshold for TIA was identified by performing repeated Spearman correlation analyses systematically on growing subsets of data. The first correlation analysis was performed using the watershed response variables and TIA only for sites with TIA >29%; the second analysis included only sites with TIA >29%, etc. until the final analysis, which included all sites. The strongest correlations were found to occur at TIA = 4% for the runoff ratio (ρ = 0.904, p < 0.0001) and at TIA = 16% for the flashiness measures (RBI: ρ = 0.896, p = 0.0004; T_{Qmean}: ρ = -0.842, p = 0.0022). These thresholds are well below those currently used by regulatory stormwater programs; for example, the General Certifications issued as part of NC DENR 401 water quality certification program state that onsite stormwater management is not required for a development project until it exceeds a “high density development” threshold of 24% imperviousness. It was actually fairly rare for watersheds in this study to even exceed this regulatory threshold (only 5 of 69 sites), in spite of many of them being located in or near urban areas. These results suggest that there may have been significant changes to watershed hydrologic responses well before the 24% threshold is reached.

One additional notable characteristic of the scatterplots of watershed slope and hydrological responses was that ICP sites did not quite seem to fit the gradient exhibited by M and P sites; while ICP slopes and levels of TIA overlapped with P watersheds, the ICP exhibited low runoff ratios in a relatively narrow range. The unique responses of ICP watersheds could be attributable to the much higher percentage of wetland land cover in the ICP watersheds, which could provide additional storage within the watersheds. The median wetland cover (according to the NLCD data set) was 13% for ICP sites (range 4-23%), 1% for P sites (range 0-4%), and <1% for all M sites. Alternatively, underlying geology may facilitate higher rates of vertical drainage and deep groundwater recharge in ICP watersheds due to the unconsolidated sediments underlying these sites (Anderson and Emanuel 2008) as compared to the harder crystalline bedrock which underlies the P and M sites; an increase in storage and recharge would account for the relatively low flashiness and runoff ratios observed in ICP.

Lower runoff ratios could also be attributable to higher ET rates, which could be due to differences in the dominant vegetation type within different regions. All forest types were combined for land cover analysis in this study; however, it has been shown that watersheds dominated by evergreen forests (e.g., pine) have lower water yields than deciduous forests (Swank and Douglass 1974). Evergreen forest cover was more extensive in the ICP watersheds (mean = 24% of watershed area, range 9-54%) than in M watersheds (mean = 5%, range 0-15%) and P watersheds (mean = 7%, range 0-17%), suggesting that vegetation could also be a contributing factor to differences in runoff ratios in ICP watersheds.

In summary, these results suggest that in watersheds with a mean slope >10%, slope was a primary control for the watershed hydrologic response. Below 10% slope, slope exerted minimal control over either the runoff ratio or stream flashiness, and increases in these responses were, instead, due to the influence of land cover. There was a predictable increase in runoff ratio once TIA exceeded 4%, and stream flashiness was likely to show a predictable increase once TIA surpassed 16%. These values are in keeping with, or even below, the findings from a review of research on impervious area and its instream effects (Paul and Meyer 2001), where the authors found that changes to watershed hydrology (ET, runoff, and infiltration) were detectable once TIA reaches 10-20%.

Scatterplots of the hydrologic responses versus two additional land cover classes—forest cover and agriculture—were also examined (Figure 3-6). Large runoff ratios were observed in watersheds with both low and high percentages of forest cover, and the previously identified watershed slope threshold of 10% provided an adequate breakpoint for interpretation for this non-monotonic relationship. The runoff ratio had a significant negative correlation with forest cover in low slope watersheds ($\rho = 0.746$, $p < 0.0001$). From a water balance perspective, this pattern would be expected since greater forest cover may result in lower runoff ratios due to increased ET and interception. However, the inverse relationship was seen in high slope watersheds, with a significant positive correlation found between forest cover and runoff ratios ($\rho = 0.622$, $p = 0.0003$). This suggests that the energy gradient resulting from high slopes was a stronger driver for runoff generation than forest cover, which concurs with other studies (Julian and Gardner 2014; Yadav, Wagener, Gupta 2007). While ET and interception are an integral consideration of water balances, studies have demonstrated that in third-order or greater streams, such as those in this study, watershed topography is a stronger predictor of watershed hydrologic response than climate-vegetation interactions (Julian and Gardner 2014), and therefore the impact of the vegetation on hydrologic response is reduced. Given the importance of the interplay between TIA and slope discussed previously, the decreases in runoff ratios in low slope watersheds may be due to the concomitant decreases in TIA as forest cover increased rather than merely the increase in forest cover.

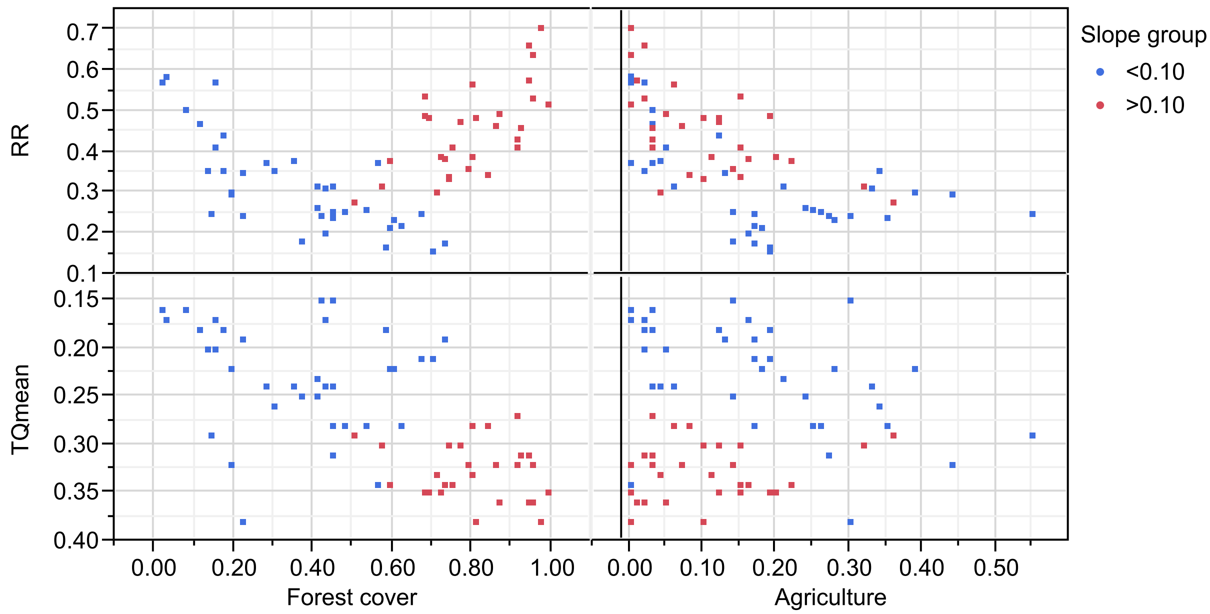


Figure 3-6 Scatterplots of watershed forest (%) and agriculture (%) cover with runoff ratio and stream flashiness (TQmean). Blue markers indicate watersheds with mean slope <10% and red markers indicate watersheds with mean slope >10%.

Conversely, forest cover and flashiness were only significantly correlated when using the full data set (RBI: ρ = -0.542, p < 0.0001; T_{Qmean}: ρ = 0.664, p < 0.0001), suggesting that regardless of watershed slope, increases in forest cover tended to be associated with decreases in stream flashiness. This more monotonic relationship contrasts with that seen for TIA, which exhibited a lack of correlation with stream flashiness at low values and a significant positive correlation was only detectable once the threshold of 4% TIA was exceeded. While high slopes determine the overall efficiency of transference of precipitation to stream discharge, forest cover moderated the rate of the transference through increases in surface roughness and deeper soils than found in non-forested landscapes, which increased lag times and resulted in less flashy streams.

In contrast to TIA and forest cover, agriculture showed a more linear relationship with the runoff ratio and seemed to show more independence from watershed slope; i.e., the two slope groups overlapped each other in the scatterplot of runoff ratio and agriculture. The correlation was fairly strong (ρ = -0.708, $p < 0.0001$), with higher levels of agriculture associated with lower runoff ratios. Agriculture was correlated with both measures of stream flashiness in the low slope watersheds (RBI: ρ = -0.465, p = 0.0037; T_{Qmean}: ρ = 0.505, p = 0.0014).

The sign of the correlations (higher percentages of agriculture were associated with lower runoff ratios and less stream flashiness) were contrary to published research. For example, Poff, Bledsoe, and Cuhaciyan (2006) found that instream hydrologic responses were similar for urban and agricultural watersheds. Agricultural land use is generally associated with the removal of woody or native vegetation from the landscape and is generally expected to increase surface or sub-surface runoff and storm flows, and decrease the potential for transpiration and baseflow (Allan 2004). Commonly used empirical measures for modeling runoff potential on the landscape confirm this idea that agriculture is associated with higher runoff. The curve number (CN) is widely used in stormwater modeling for discretely calculating the surface discharge Q for a given storm event (USDA Natural Resources Conservation Service, NRCS, 2001). CN has a range of 0-100, with higher values corresponding to greater potential for surface runoff, and the appropriate value is determined by land cover and hydrologic soil group (HSG). In (USDA NRCS 2001), the associated curve numbers for all categories of row crops ranged from 61-91 (depending on

HSG and cultivation method). This range is higher than that specified for “Woods, Good Condition” (i.e., “Woods are protected from grazing, and litter and brush adequately cover the soil.”), which has a range of 39-77 and are nearly equivalent to those for “Residential, ¼-acre lots” with an average impervious area of 38% (range 61-87).

This unanticipated relationship of positive correlations between increases in agricultural land use and better instream conditions was actually noted in several analyses throughout the project. Cursory detailed analyses of the study sites (data not shown) suggested that the relationship statewide may be due a geographic cluster of agricultural watersheds in certain river basins in the Piedmont, suggesting that there is something unique about terrain or watershed/stream management practices in these areas. Alternatively, the effects may not be due to the increases in agricultural use, but rather to the decreases in other land cover types, such as TIA.

Drainage areas (DA; km^2) were compared to the runoff ratio and flashiness among watersheds. No significant correlation was found between DA and runoff ratio, but DA did correlate significantly with RBI (ρ = -0.6321; p <0.0001) and T_{Qmean} (ρ = 0.5613; p <0.0001), confirming that smaller watersheds tend to have flashier stream discharge responses. DA and mean slope were not found to be significantly correlated.

One additional measure of stream hydrology was collected in this study, the NC SAM hydrology functional rating (low, medium, or high); low functional ratings should have been associated with disturbance. Based on the conclusions drawn from Figure 3-2, Figure 3-3, and Figure 3-5, high runoff ratios would be associated with either high slopes, suggesting a natural condition of the watershed, or with high TIA, which would represent an anthropogenic disturbance that increases total runoff. A combination of low slope and low TIA would suggest lower runoff ratios. It was therefore anticipated that sites rated high for the NC SAM hydrology function would likely have <4% TIA (exhibited low levels of watershed disturbance), and exhibit a wide range of values for the runoff ratio in a response to watershed slope (as the runoff ratio in undisturbed watersheds would respond to the energy gradient provided by slope, which varied greatly in the low TIA watersheds).

The majority of NC SAM hydrology ratings were high (41 of 67 sites), and the high-rated sites predominantly had low TIA and the widest range of slopes and runoff ratios (Figure 3-7), as was anticipated. Most of the 18 sites with TIA >4% were associated with low or medium NC SAM ratings, though there were four sites with higher TIA that received high NC SAM ratings: QB351 and QB352 (both on Goose Cr. near Charlotte, NC; 6-7% TIA); JB165 (Ellerbe Cr., which drains downtown Durham, NC; 20% TIA); and BB407 (N. Buffalo Cr., which drains Greensboro, NC; 27% TIA). These sites were all notable in that though they had large amounts of development in their respective watersheds, the assessment reaches were well forested. This suggested that local condition may be more heavily weighted than watershed condition by the NC SAM model and that in certain cases NC SAM did not adequately reflect the watershed hydrologic response variables used in this study.

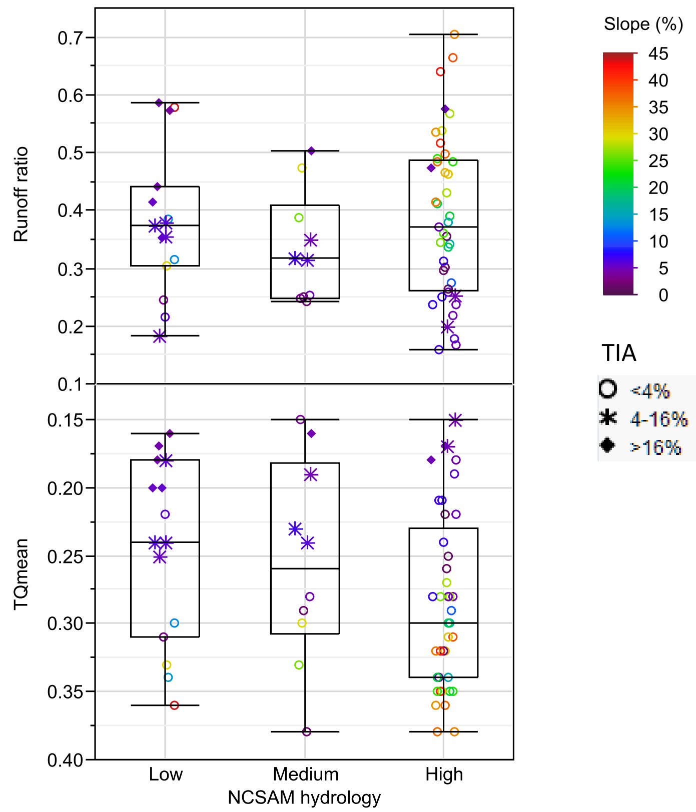


Figure 3-7 NC SAM hydrology ratings, runoff ratios, and stream flashiness (T_{Qmean}). Watershed slope is indicated by marker color and TIA class by marker shape.

Stream flashiness was expected to have a more straightforward, monotonic relationship with NC SAM hydrology, and it did show a weak but significant negative correlation with T_{Qmean} (ρ = -0.252, p = 0.0143) though not with RBI. Similarly to the runoff ratio, the high-rated sites predominantly had low TIA with the same four exceptions (sites QB351, QB352, JB165, and BB407). Interestingly, there were five sites that received a low NC SAM hydrology rating in spite of having low TIA and less flashy systems (defined as $T_{Qmean} \geq 0.30$): EB145 (Swannanoa R. in Asheville, 71% forested), GB011 (Oconaluftee R. near Cherokee, 94% forested), OB090 (Tar R. in Tarboro, 45% forested), QB112 (Yadkin R.

in Elkin, 73% forested), and QB206 (Hunting Cr. near Harmony, 57% forested). With the exception of QB206, the assessment reaches for these sites were located in urban areas and/or had major roads flanking the stream, but had highly forested watersheds. QB206 was located in a rural area, but the assessment reach was notable for an extremely large artificial berm or levee on one bank which severely inhibited local overland or shallow subsurface flow from reaching the stream. In this case NC SAM correctly identified this site as having locally impaired hydrologic function that would not necessarily be captured by the flashiness metrics used in this study. These conditions again suggest that NC SAM may weigh local conditions more strongly than watershed conditions when rating the stream hydrology function.

Results suggest that for many sites, NC SAM results were as anticipated in terms of the watershed hydrologic responses used in this study. NC SAM considers only one watershed condition factor-- TIA is greater or less than 24%-- and so the rating is primarily dependent on local factors, such as stream channel stability and modifications, streamside/floodplain access and condition (where appropriate), riparian vegetation extent and condition, instream substrate, and variety of aquatic life. These factors are used to determine the Flood Flow and Baseflow metrics, on which the hydrology functional rating is based. Results suggest that, in many cases, using these secondary indicators provided appropriate hydrology ratings as compared to the watershed hydrologic response variables used in this study. However, there were a number of cases (8 of 69 sites, or 12%) where local conditions conflicted with watershed condition, particularly land cover. Consideration of watershed conditions would likely increase the sensitivity of NC SAM to modifications of stream hydrologic functions.

3.3 Instream physical conditions

Objective 2 of the study *Research Questions* (Section 1.4) was further addressed by comparison of watershed hydrologic responses to measures of local instream physical condition, represented by channel dimensions, the field habitat assessment score, and NC SAM habitat function rating. Channel dimensions measured included bankfull (BF) height, width, and height/width ratio, which are often related to watershed characteristics. Top-of-bank (TB) height and width were also measured. For this analysis, only sites with channel measures that were directly measured were used, so extremely large rivers and non-wadeable sites where one or more dimensions had been estimated were excluded, reducing the number of sites for analysis to 47. A summary of the results are provided in Table 3-1. A Wilcoxon signed rank matched pairs analysis indicated that TB heights were significantly larger than BF heights, suggesting that stream incision was fairly widespread for the study sites.

Table 3-1 Minimum, maximum, and mean channel dimensions.

		BF (ft.) Min-Max (Mean)		**TB (ft.)** Min-Max (Mean)	
Region	**n**	**Height**	**Width**	**Height**	**Width**
Mountains	8	1.9-5.2 (3.6)	42-170 (81.5)	3.0-11.7 (7.1)	50-104 (80.1)
Piedmont	32	1.0-5.0 (2.6)	19-180 (53.3)	1.7-12.5 (6.7)	22-195 (62.5)
ICP	7	1.7-6.2 (3.4)	14-142 (56.9)	2.8-10.2 (6.4)	22.5-144 (69.4)
Statewide	47	1.0-6.2 (2.9)	14-180 (58.6)	1.7-12.5 (6.7)	22-195 (66.3)

Table 3-2 Results of Spearman correlations between channel dimensions and watershed DA, slope, runoff ratio, and flashiness. Significant correlations (p<0.05) are underlined and bold.

Variable	BF height		BF width		TB height		TB width	
	ρ	p	ρ	p	ρ	p	ρ	p
DA (km^2)	**0.397**	0.0057	**0.849**	<.0001	**0.344**	0.0179	**0.831**	<.0001
Mean slope	0.180	0.2254	**0.517**	0.0002	0.051	0.7343	**0.471**	0.001
Runoff ratio	0.222	0.1380	0.285	0.0550	0.156	0.3018	0.259	0.0855
RBI	**-0.380**	0.0092	**-0.581**	<.0001	-0.137	0.3648	**-0.580**	<.0001
T_{Qmean}	**0.374**	0.0105	**0.558**	<.0001	0.120	0.4286	**0.517**	0.0003

Comparisons between channel dimensions, DA, and watershed slope showed identical patterns for both BF and TB measurements, though the correlation was much stronger for width than height. BF and TB widths were anticipated to be correlated; incised channels with steep banks would be more likely to suffer bank failures and slumps, thus widening the TB measurement proportionally with the BF width. The similarities between heights was surprising, as differences between the two values would represent the extent of incision of a stream, which one would assume would be independent of DA and more dependent on factors such as current and legacy land use.

Runoff ratio did not exhibit any significant correlations with channel dimensions, though the correlation between runoff ratio and BF width was marginally significant (p = 0.0550). RBI and T_{Qmean} showed inverse correlations with BF dimensions, such that flashier systems (higher RBI or lower T_{Qmean}) were associated with narrower, deeper stream channels. Previously discussed results (Section 3.2) suggested that flashiness was more strongly associated with land use in lower slope watersheds, but repeating the flashiness/channel dimension analyses with further grouping by land use and/or slope weakened the correlation between the two sets of measures. However, stream incision may not necessarily be representative of current land use; it may instead reflect past disturbances, such as excess sediment loading from legacy agricultural uses (Jefferson and McGee 2012) or impoundments such as millponds (Walter and Merritts 2008).

Two channel dimension ratios were compared to the watershed hydrology measures: width/height ratio for BF, and height BF/height TB ratio. The BF width/height ratio significantly correlated with DA ($\rho = 0.5602$; $p < 0.0001$), though the strength of the correlation was less than that seen with BF width alone. The ratio of BF height to TB height is commonly used as an indicator of stream instability and disturbance (Rosgen 1994), where stable streams are expected to have a BF/TB height ratio close to 1.0, and incision increases as the ratio decreases.

Comparisons of channel dimensions and ratios were also made directly to watershed land use. Significant correlations are shown in Table 3-3, with higher development and TIA resulting in an increase in stream incision, as indicated by smaller BF/TB ratios. Increases in forest cover led to decreases in stream incision. There was a notable lack of correlations between channel dimensions and agricultural land use (results not shown). The correlations between channel dimensions and land cover that were identified were slightly weaker than those seen with DA. The correlations were also weaker than those seen between land use and stream flashiness, suggesting that land use affects the channel morphology via the mechanism of stream flashiness.

Table 3-3 Significant correlations between channel dimensions and watershed land cover.

	Land cover class					
	Development		TIA		Forest	
Channel measure	ρ	p	ρ	p	ρ	p
TB width/height ratio	**-0.299**	0.0417	**-0.307**	0.0358	**0.400**	0.0053
BF/TB height ratio	**-0.411**	0.0037	**-0.410**	0.0038	**0.326**	0.0238
BF width	NS[1]	NS[1]	**-0.291**	0.0446	**0.435**	0.0020

[1] No significant correlation

Interestingly, land use was significantly correlated with BF width but not with BF depth. While it would be anticipated that disturbance would lead to both widening and deepening of the channel, these results could be explained by the fact that width may be more directly tied to watershed hydrologic responses, in that flashier discharge patterns may lead to greater bank erosion. Stream depth, however, is a function of sediment transport, which entails both an erosive and a depositional component.

Field habitat assessments were completed at 55 of the 69 sites (Table 3-4). The exclusion of the riparian assessment questions resulted in minimal changes to the percent score in the P and ICP, but at the M sites the mean modified habitat score was six points greater than the mean full habitat assessment score. P sites showed the widest range of habitat conditions and approached a normal distribution (Figure 3-8). M and ICP sites were skewed towards higher values, suggesting better overall instream habitat for these sites.

Table 3-4 Summary of field habitat assessment results by region, showing the results from the full assessment, and results when riparian scores are excluded.

Region	n	Full habitat assessment (%) Min – max (mean)	Instream habitat only (%) Min – max (mean)
M	13	55 - 95 (77)	60 - 96 (83)
P	36	24 - 92 (65)	28 - 91 (62)
ICP	9	41 - 96 (78)	49 - 98 (79)

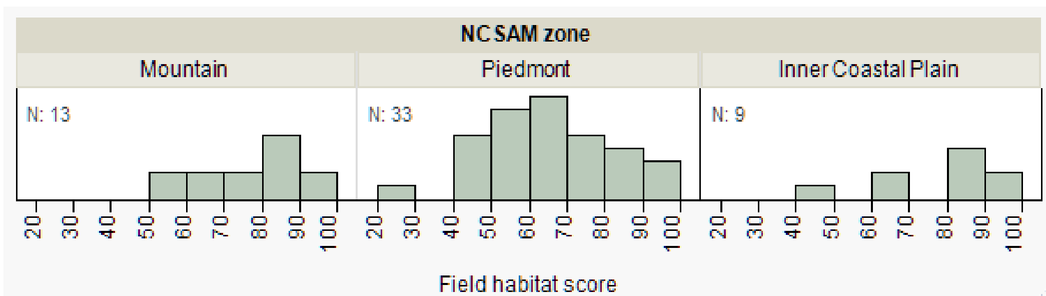


Figure 3-8 Distributions of modified field habitat assessment scores (riparian scores excluded) by region.

The modified field habitat assessment results were compared to the same watershed conditions and hydrology metrics that were previously compared to channel dimensions. It was anticipated that by limiting the habitat scores to physical instream condition, they would show a similar set of responses as channel dimensions. This assumption was confirmed by comparing the instream habitat score to the individual channel dimensions as well as the ratios described above, and significant correlations occurred between instream habitat and the BF/TB height ratio ($\rho = 0.3965$, p = 0.0053). DA did not show any significant correlations with the instream habitat assessment, suggesting that this habitat assessment field method was relatively insensitive to this factor and therefore applicable to the range of stream sizes and landscape positions seen in this study.

When comparing the runoff ratios from all sites and the habitat assessment scores, no significant correlations or other obvious patterns were identified. When split into high and low slope watersheds using the previously determined threshold of 10% (Figure 3-9), low slope watersheds showed a significant negative correlation between runoff ratio and instream habitat (ρ = -0.4940, p = 0.0026), whereas high slope watersheds exhibited a significant positive correlation (ρ = 0.5389, p = 0.0097), similar to what was seen for forest cover and runoff ratio. If the habitat score was driven solely by the runoff ratio, then a monotonic relationship should have been identified. However, results suggest that the habitat score responded to the same factors as the runoff ratio, namely land cover in low slope watersheds and increasing slopes in high slope watersheds. Relationships between stream flashiness and instream habitat were also significant (RBI: ρ = -0.5101, p < 0.0001; T_{Qmean}: ρ = 0.5484, p < 0.0001), confirming the widely held belief that degraded instream habitat is associated with flashy flows, regardless of watershed slope.

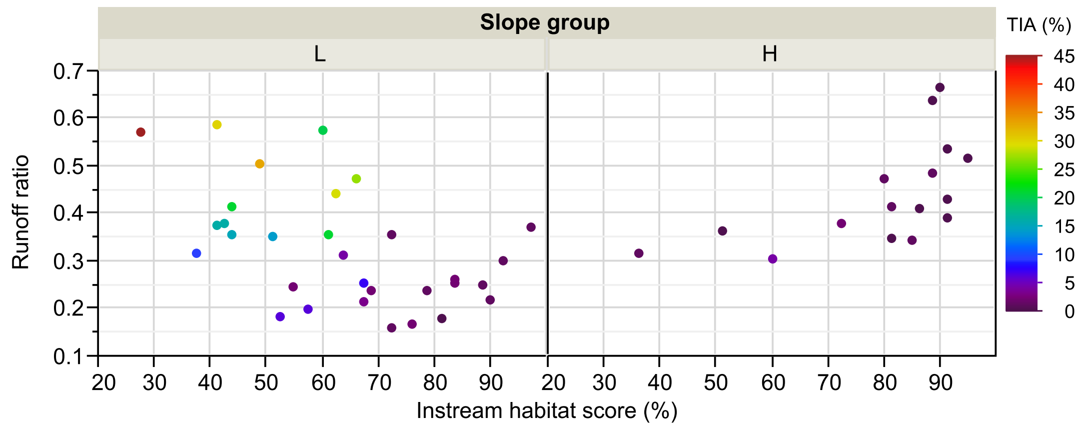


Figure 3-9 Scatterplots of field instream habitat assessment score (%) and runoff ratio by watershed slope group (L: slope <10%; H: slope >10%.

Correlations between instream habitat and land use were strongest when watersheds were not grouped by slope, at least for development, TIA, and forest cover. Agricultural land use is often considered detrimental to instream physical conditions, primarily due to increased mobilization of soil and sediment, and increases in stream flashiness due to the loss of vegetative cover. However, agriculture was only significantly correlated with instream habitat when sites were grouped by slope groups. The correlation was positive among low slope watersheds, providing another example of an unanticipated increase in a measure of instream condition when agricultural land use increased. The correlation was negative among higher slope watersheds, where increased agricultural land cover was associated with lower instream habitat scores.

Correlations between land use and instream habitat were much stronger than those seen for the watershed hydrology metrics, suggesting that instream condition was more strongly linked to land cover, possibly via a different mechanism than runoff ratio or stream flashiness. Other possible mechanisms could include sediment transport processes (as was suggested previously for channel depth), which would drive substrate composition, or the availability of woody debris from local or upstream sources. Neither of these would necessarily be reflected in the watershed hydrology responses, though both are important in terms of providing appropriate habitat for benthic macroinvertebrates and fish. This underscores the importance of considering the many complex processes involved in determining the ecological status of a given stream reach, and provides support for the multi-metric approach of assessment methods such as NC SAM.

Table 3-5 Results of Spearman correlation analysis of instream habitat assessment score and watershed land use.

Land use category	All watersheds		Slope <10%		Slope >10%	
	ρ	**p**	***ρ***	**p**	***ρ***	**p**
Development	**-0.707**	<0.0001	**-0.716**	<0.0001	**-0.488**	0.0211
TIA	**-0.713**	<0.0001	**-0.684**	<0.0001	**-0.576**	0.0051
Agriculture	0.022	0.8729	**0.494**	0.0022	**-0.428**	0.0467
Forest	**0.633**	<0.0001	**0.519**	0.0012	**0.542**	0.0092

NC SAM ratings from the same 55 sites where full habitat assessments were conducted were compared to the watershed characteristics (slope, DA), hydrologic response variables (runoff ratio, flashiness), channel dimensions (BF width, BF height, ratio of BF height/ToB height), and land cover (TIA, forest cover, agriculture) to determine if these variables were significantly different between the low-rated habitat and high-rated habitat sites; however, due to small sample size for medium-rated sites, correlation analyses were not performed and analyses were limited to comparisons of group medians using the

Wilcoxon test. Significant differences between low- and high-rated sites were only identified for two factors: TIA (p = 0.0302) and BF height (p = 0.0362). Low-rated sites had a median TIA of 4%, whereas high-rated sites had a median of 3%. BF height, which has not been shown to be a reliable response variable for other watershed condition or hydrologic response variables, was higher in low-rated sites (median = 3.8) than in high-rated sites (median = 2.8). As compared to the field habitat assessment scores, NC SAM habitat showed a much weaker relationship to non-local, watershed factors and to other measures of local condition.

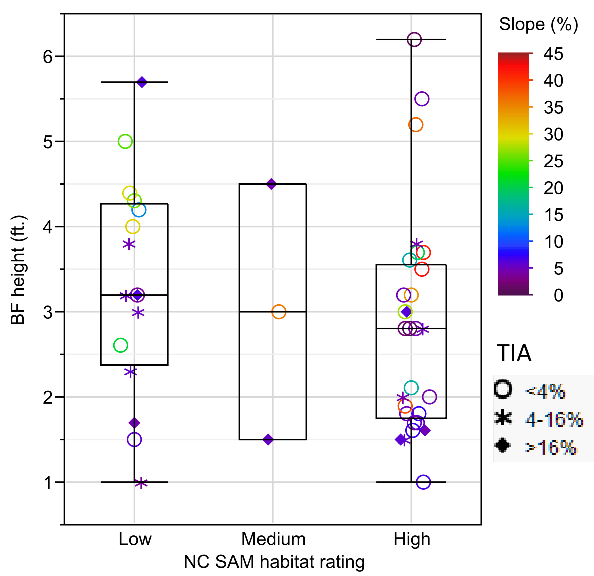


Figure 3-10 Distributions of BF height and TIA by NC SAM habitat rating.

While higher NC SAM habitat ratings were generally associated with slightly lower impervious area (TIA), there were seven sites with high NC SAM ratings that had high TIA (15-30%). All of these sites were located in the Piedmont region, and were notable for very high levels of development (55-96%), low field habitat assessment scores (45-69), and low

BF/TB height ratios (0.20-0.58). The sites were BB312 (E. Fork Deep R. in Greensboro), BB407 (N. Buffalo Cr. in Greensboro), CB139 (McDowell Cr. near Charlotte), CB146 (L. Sugar Cr. near Charlotte), JB052 (Swift Cr. in Cary), JB156 (Crabtree Cr. in Raleigh), and JB165 (Ellerbe Cr. in Durham). These sites had highly urbanized watersheds but the assessed reaches had high forest cover or wide (50-100 foot) vegetated riparian buffers. All were characterized by fine sand/silt substrates with high embeddedness with only small amounts of cobble or gravel present, and issues with bank erosion/scour and bank failures were widespread. This is another line of evidence that suggested that by ignoring watershed conditions and relying solely on local conditions, particularly riparian vegetation, NC SAM may be overestimating instream conditions in certain cases.

While the NC SAM habitat rating did not respond as expected to watershed condition or hydrological characteristics, the analysis was confounded by the inclusion of riparian habitat condition in the NC SAM functional assessment method. As with the field habitat assessment, NC SAM may reflect other watershed processes, such as sediment transport, that were not considered in this project. The modified field habitat assessment used for this study did exhibit stronger correlations to land cover than watershed hydrologic response measures, suggesting that instream assessment methods that account for additional watershed and stream processes, such as sediment transport, may provide a more complete picture of the ecological instream conditions.

3.4 Water quality

While water is important as a vector for contaminants, water quality problems necessarily depend on the presence of a contaminant source, and therefore it was anticipated that only weak relationships, if any, would be identified between watershed hydrologic responses and instream water quality during this next set of analyses addressing *Objective 2* of the study *Research Questions* (Section 1.4). It was anticipated that water quality would be more closely tied to land cover.

The distributions of the mean values for $\log_{10}$-transformed SC (µS/cm at 25°C) and turbidity (NTU), calculated from the long-term monitoring data collected by other programs for 58 of this project's study sites, are shown in Figure 3-11, along with the distribution of NC SAM water quality ratings from field assessments, presented by region. Scatterplots of SC and turbidity as a response to forest and agriculture land cover among sites are shown in Figure 3-12. The region of the state (M, P, or ICP) and watershed TIA group (<4%, 4-16%, >16%) are indicated by marker color and shape, respectively. Different land cover classes co-vary with each other when calculated for a discrete land area, so the inclusion of TIA assisted with determining if effects may be due to the increased prevalence of one land cover class or the decreasing prevalence of another. For example, forest cover appeared to have a fairly strong correlation with SC, regardless of the level of TIA in the watershed, with higher forest cover associated with lower SC ($\rho = -0.849$, $p < 0.0001$). However, the relationship between turbidity and forest cover was more complex. In this case, there was a linear, inverse correlation between the two factors using all data ($\rho = -0.451$, $p = 0.0004$), but the clustering of certain TIA groups, particularly watersheds where TIA was >16%, suggest that in spite of the significant correlation, turbidity cannot be reliably predicted using forest cover alone.

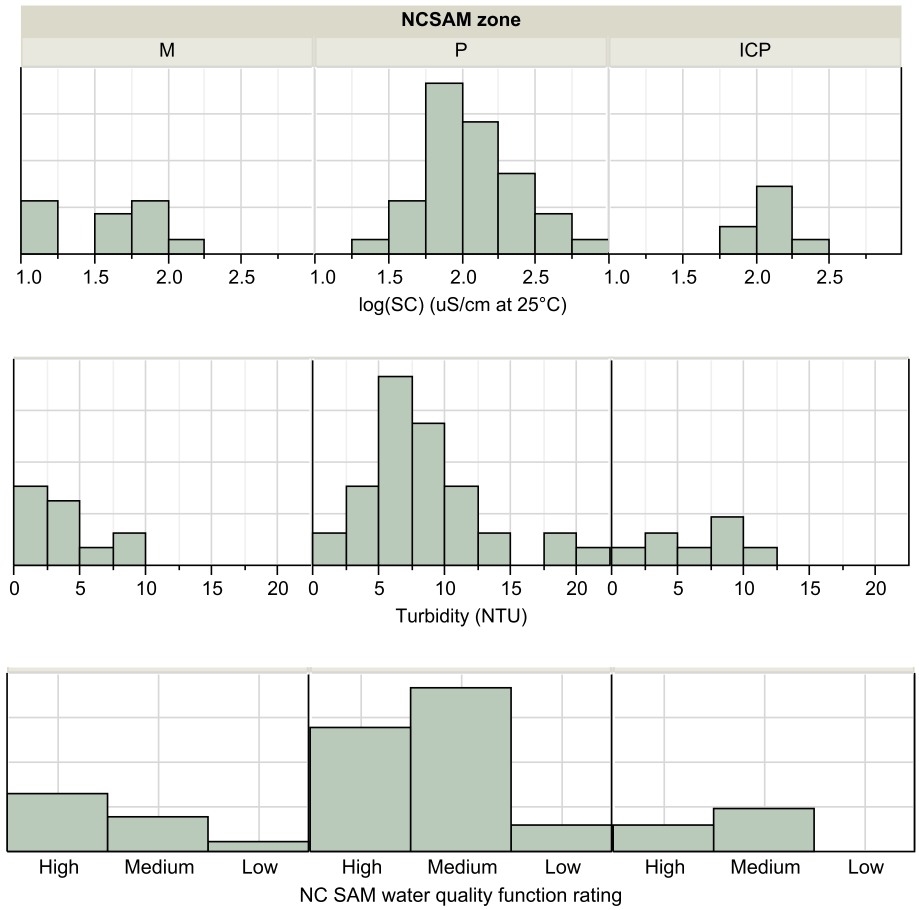


Figure 3-11 Distributions of mean SC (top, uS/cm at 25°C) and turbidity (middle, NTU) for study sites from long-term monitoring programs by region. Bottom graph depicts the distribution of results from NC SAM assessments for this study, with high ratings associated with low SC and low turbidity.

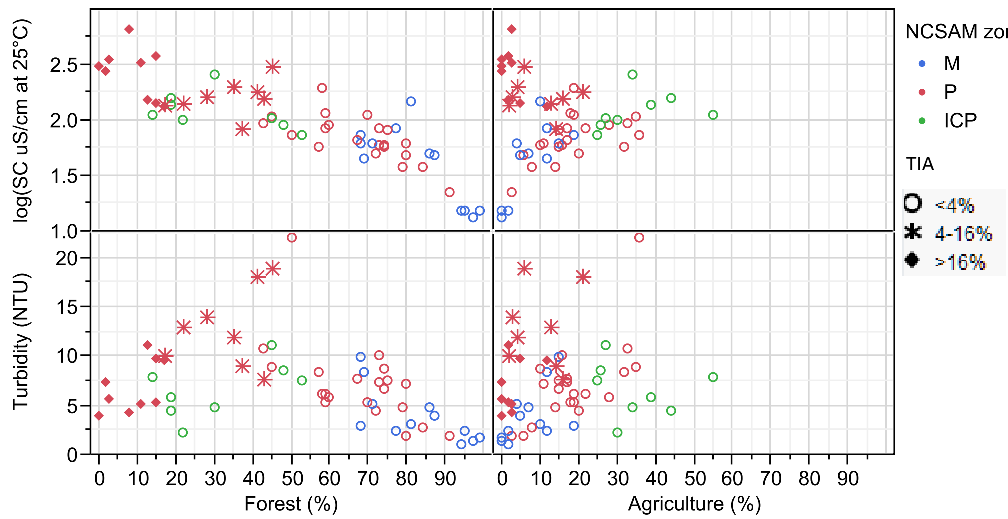


Figure 3-12 Scatterplots of SC and turbidity as a function of forest and agriculture land cover. The marker color indicates region (M, P, or ICP) and marker shape indicates the TIA of the watershed.

The watersheds with moderate (4-16%) and high (>16%) TIA were somewhat clustered in the scatterplots of agriculture/SC and agriculture/turbidity. While excluding the moderate and high TIA sites from the forest/turbidity correlation did not appreciably increase the strength of that correlation, excluding these data from the agriculture/water quality analyses markedly improved the strength of the correlations, with both SC and turbidity increasing with increasing agriculture. For SC, there was not a significant correlation using the entire data set, but for the low TIA watersheds only $\rho = 0.751$, $p < 0.0001$. For turbidity, the correlation using all data was marginally significant using all data ($\rho = 0.257$, $p = 0.0512$), but increased to $\rho = 0.538$, $p = 0.0003$ when using only the low TIA watersheds. These results suggest that the effects of impervious surface may quickly overwhelm the effects of competing land uses within the watershed, including the beneficial effects of forest cover.

Interestingly, watersheds with highest levels of TIA (>16%) had lower levels of turbidity than those with moderate TIA (4-16%) (Figure 3-13), and comparison of turbidity values by watershed TIA group confirmed that instream turbidity values for the lowest TIA (<4%; median = 5.8 NTU) and highest TIA (>16%; median = 5.7 NTU) watersheds were very similar; a Wilcoxon test indicated no significant differences between the medians. The watersheds with moderate TIA (4-16%) had a higher median turbidity (12.5 NTU), which was significantly different from the low TIA ($p = 0.0002$) and high TIA ($p = 0.0061$) watersheds.

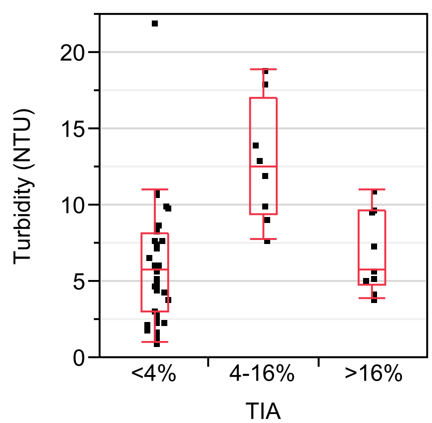


Figure 3-13 Box plots of mean turbidity (NTU) by watershed TIA.

The effects of increases in impervious surface and urbanization are often thought to be monotonic on instream water quality, but these data suggest that turbidity was actually most elevated within an intermediate range of TIA. One explanation may be that moderate development may provide more available bare soil for mobilization; excessive impervious surfaces may in fact limit the amount of sediment that is available for transport during storm events and therefore limit instream turbidity. However, given that instream turbidity in lotic systems is often a storm-driven event, the conditions under which the samples were taken are crucial to full interpretation of results. The turbidity data used in this analysis did not indicate the conditions under which the samples were taken, for example, baseflow or on the rising or falling limb of a stormflow event, and so further examination of this source of variability in the data set could not be completed.

Another notable feature of the scatterplot of turbidity and forest cover was an unusual clustering of five ICP sites in the lower left corner this graph that were associated with low forest cover and low turbidity. The sites were BB126 (NE Cape Fear R), IB039 (Lumber R.), JB106 (Nahunta Swamp), OB107 (Chicod Cr.), and PB004 (New R.), and their watersheds were entirely located within the ICP region; the remaining three ICP sites that seem to fit the gradient of low-TIA M and P sites had outlets located in ICP, but large portions (approximately 40-95%) of the watersheds were actually located in P region. This provides further evidence that controlling for regional variability by grouping stream assessment sites by the ecoregion in which the *outlet* is located instead of where the majority of the watershed area is located may provide misleading results.

Scatterplots of SC and runoff ratio (Figure 3-14) suggested two different water quality responses due to the watershed slope and TIA. One response exhibited increasing SC with increasing runoff ratio and TIA; the other showed decreasing SC in response to increasing runoff ratio, increasing slope, and decreasing TIA. These are similar to the patterns seen between runoff ratio and watershed conditions (Figure 3-2, Figure 3-3, Figure

3-5), and suggests that runoff ratio and SC are responding to similar stimuli, rather than runoff ratio driving SC. SC showed a more linear relationship with flashiness (T_{Qmean} and RBI, not shown), with the highest SC values associated with the highest TIA (>16%) and highest flashiness.

Turbidity exhibited moderate correlations with runoff ratio (ρ = -0.386, p = 0.0030) and T_{Qmean} (ρ = -0.354, p = 0.0069), but not with the RBI flashiness metric. Higher runoff ratios were associated with lower turbidity, due to high-slope watersheds; this may likely be due to the tendency for the high slope watersheds to be more heavily forested.

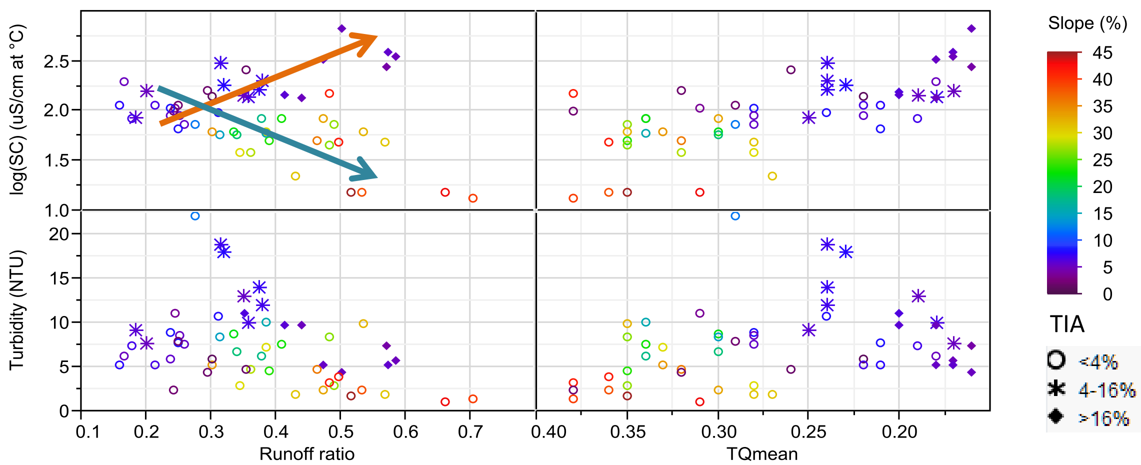


Figure 3-14 Scatterplots of turbidity (NTU) and $\log_{10}$(SC (μS/cm at 25°C)) as a function of the runoff ratio and stream flashiness (T_{Qmean}). Marker color indicates mean watershed slope (%), and marker shape indicates watershed impervious area (TIA, %). The arrows are conceptual, and represent the direction of influence of TIA (brown arrow) and watershed slope (blue arrow).

Comparison of $\log_{10}$(SC) and turbidity to slope and TIA (Figure 3-15) shows the patterns observed in Figure 3-2, Figure 3-3, and Figure 3-5, when these watershed characteristics were compared to runoff ratio and flashiness: below certain thresholds, water quality indicators varied widely, but once the threshold was crossed a more distinct pattern was evident. Given the similar responses of SC, runoff ratio, and stream flashiness to watershed slope and TIA, SC may be helpful as an relative indicator of not only water quality (e.g., nutrient concentrations), but also of watershed hydrologic responses, particularly stream flashiness.

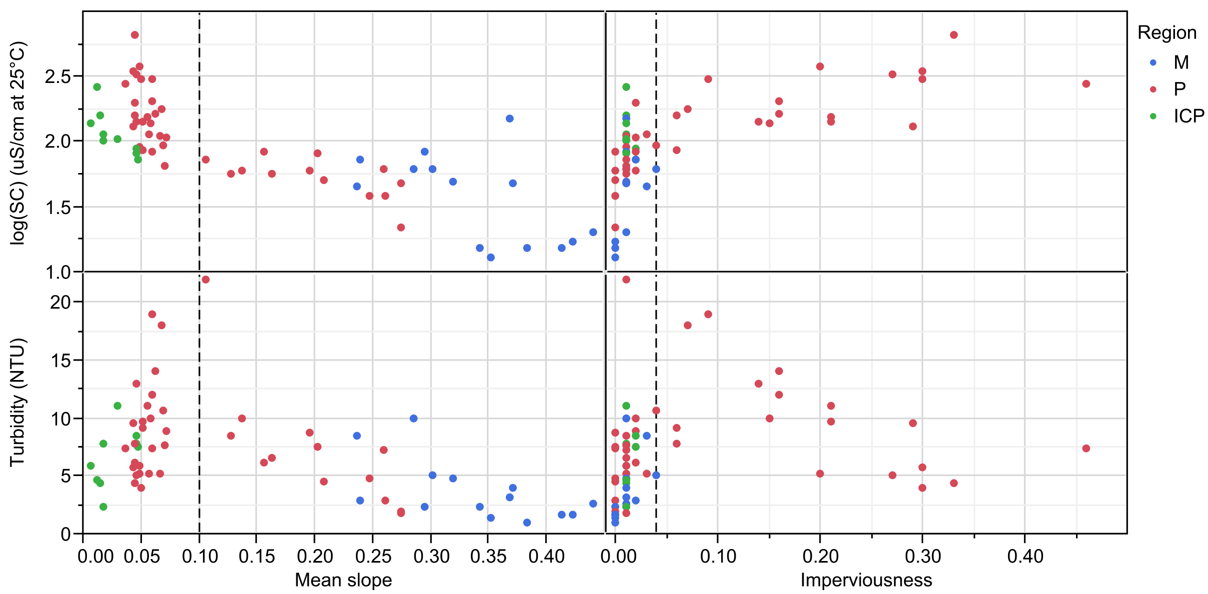


Figure 3-15 Scatterplots of SC and turbidity against mean watershed slope and TIA. Dashed lines indicate previously identified thresholds for slope (10%) and TIA (4%).

The SC data also suggested sharp decreases in response to very small changes in slope below the previously identified thresholds (10% slope, 4% TIA), SC dropped sharply in response to very small changes in slope, and increased sharply in response to very small changes in TIA and declines were more gradual at higher slopes and increases more gradual for TIA >4%. This suggests that water quality degradation may begin occurring even before the 4% TIA threshold and certainly well below the 24% threshold commonly used by stormwater regulatory programs.

Comparison of NC SAM water quality functional ratings to watershed hydrologic responses and conditions was hindered by a paucity of sites rated low by NC SAM; only 4 of the 58 sites used for the water quality analyses rated low, compared to 29 that rated medium and 25 that rated high. Correlation analyses were not performed and analysis was limited to qualitative assessments and statistical comparison of medians for each associated independent factor (e.g., runoff ratio, land cover) by NC SAM category using the Wilcoxon rank sum to determine if the medians were significantly different. Visual examination (Figure 3-16) indicated that all sites rated high by NC SAM for water quality were associated with low TIA (<4%) and a wide range of watershed slopes, with higher slopes associated with higher runoff ratios, as expected. Sites receiving a medium or low NC SAM rating were associated with a mix of TIA and slope values, including watersheds with high slopes and low TIA that would be expected to have minimal water quality problems, though the majority of the watersheds did have increased TIA and low slopes. However, comparison of the watershed hydrologic responses found no significant differences in medians for the different NC SAM water quality rating with the exception of T_{Qmean}. For this measure of flashiness, the high rated sites were significantly less flashy (indicated by a higher T_{Qmean} value) than the medium rated sites ($p = 0.0496$; high, median = 0.3; medium, median = 0.25).

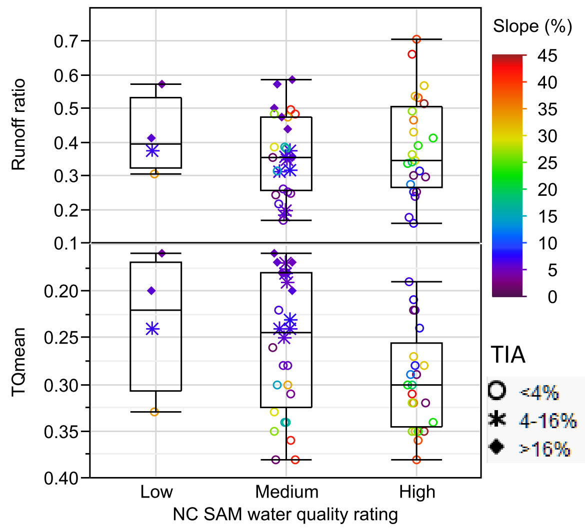


Figure 3-16 Boxplots of T_{Qmean} and runoff ratio by NC SAM water quality rating. Marker color indicates watershed slope (%) and marker shape indicates watershed TIA.

The distributions of log_{10}SC, turbidity, land cover, TIA, and watershed hydrologic responses were compared based on their regulatory use impairment status (see Section 2.5 for a description of use support assessments). A discussion of results from the analysis of NC SAM water quality ratings and stream impairment was previously provided in Section 3.1, with more detailed results provided in Appendix C: Impairment ("303(d) listing"). Impairment is determined by the NC Department of the Environment and Natural Resources, and can be based on the frequency of violations of water quality standards, health of instream communities, fish tissue analyses, or other information that suggest conditions threaten designated use of a waterbody, such as aquatic life support, recreation, or water supply. Only sites which were impaired based on water quality standard violations were used in this analysis.

Medians of impaired and supporting sites were compared using the non-parametric Wilcoxon method (Table 3-6). Impaired streams had significantly higher turbidity, SC, TIA, were flashier (had higher RBI and lower T_{Qmean}), and had lower forest cover, suggesting a coupling of land cover, water quality, and watershed hydrologic responses.

The median value for TIA for impaired streams was relatively low (6%), but had a wide range (0-46%). The TIAs for supporting streams were all ≤ 4%, providing additional evidence that better water quality is associated exclusively with very low levels of TIA. However, low TIA was also no guarantee of good water quality, as nearly half of impaired streams had low levels of TIA, a reminder that there are multiple reasons for poor water quality. Reviewing the impaired sites with TIA <4% indicated that the majority of streams were impaired due to high turbidity (8 sites) and six were impaired due to low pH. All of these sites were located in the western Piedmont or Mountain regions of the state, areas that have historically been subject to impacts from acidic deposition (Reuss and Johnson 1986), which may explain the prevalence of low pH values, and would therefore not be captured by a spatial analysis. Several of these impaired sites were designated as trout waters (supplemental classification Tr); while turbidity has a maximum allowable level of 50 NTU in most waters of the state, the standard for Tr waters is much more stringent (10 NTU) (NC Administrative Code, 15ANCAC 02B), and therefore may be more likely to be exceeded.

Table 3-6 Results of Wilcoxon analysis for differences in medians for land cover, water quality, and watershed hydrologic response variables by regulatory use support status (Supporting or Impaired).

Variable	Z	p-Value	Supporting	Impaired
Turbidity (NTU)	2.4483	**0.0144**	5.2	8.4
SC (µS/cm at 25°C)	1.9643	**0.0495**	82	119
TIA	4.1282	**<0.0001**	1%	6%
Forest	-3.4183	**0.0006**	71%	43%
Planted/Cultivated	-1.1993	0.2304	15%	10%
Runoff ratio	1.1285	0.2591	0.34	0.38
RBI	2.6748	**0.0075**	0.31	0.50
T_{Qmean}	-2.3439	**0.0191**	0.30	0.24

In summary, results indicate that specific conductance, as an indicator of water quality, was influenced by watershed land cover and slope, two factors were also correlated with the watershed hydrologic responses, the runoff ratio and stream flashiness. The thresholds for TIA (4%) and watershed slope (10%) that were previously identified appear to provide an acceptable breakpoint for the non-monotonic responses seen for this water quality indicator. However, scatterplots suggested once the threshold of 4% TIA is crossed, SC is likely to be high. Low TIA, however, was not a guarantee that all water quality standards will be met, as many study streams considered impaired due to standard violations had very low watershed TIA values. Relationships between turbidity and watershed condition diverged from the patterns seen with specific conductance and the watershed hydrologic response variables, with the highest turbidity values associated with moderate TIA (4-16%). Full assessment of the impact of watershed condition on turbidity, however, may require additional consideration of sediment transport processes and stormflow dynamics, which were not addressed in this study.

3.5 Bioclassification and NC SAM overall function

Bioclassifications and the NC SAM overall function ratings were intended to reflect integrated interactions of local and watershed conditions, including physical, chemical, and biotic conditions, and to provide the most comprehensive information on the relationships between instream ecological conditions and the watershed conditions, as part of addressing *Objective 2* of *Section 1.4*. The analysis used 63 of the total 69 study sites and focused on the relationships between bioclassifications, NC SAM overall rating, watershed hydrology variables, and the two major influences on flow regimes (TIA and mean watershed slope). The NC SAM overall function was converted to a rank value of 1-10, with lower values indicating the highest overall levels of stream function (see Appendix C: for details on calculation of the rank values).

Bioclassifications (Figure 3-17) were visually skewed towards healthier communities for M sites, and towards moderately stressed communities in ICP. Communities at P sites were skewed towards moderately and highly stressed conditions. NC SAM rated the majority of sites (35 of 63 sites) as ranks 1-2, which correspond to the highest levels of overall stream function. Bioclassifications were assumed to be roughly equal to the NC SAM overall rank values, such that Excellent ≈ 1-2; Good ≈ 3-4; Good-Fair ≈ 5-6; Fair ≈ 7-8; and Poor ≈ 9-10.

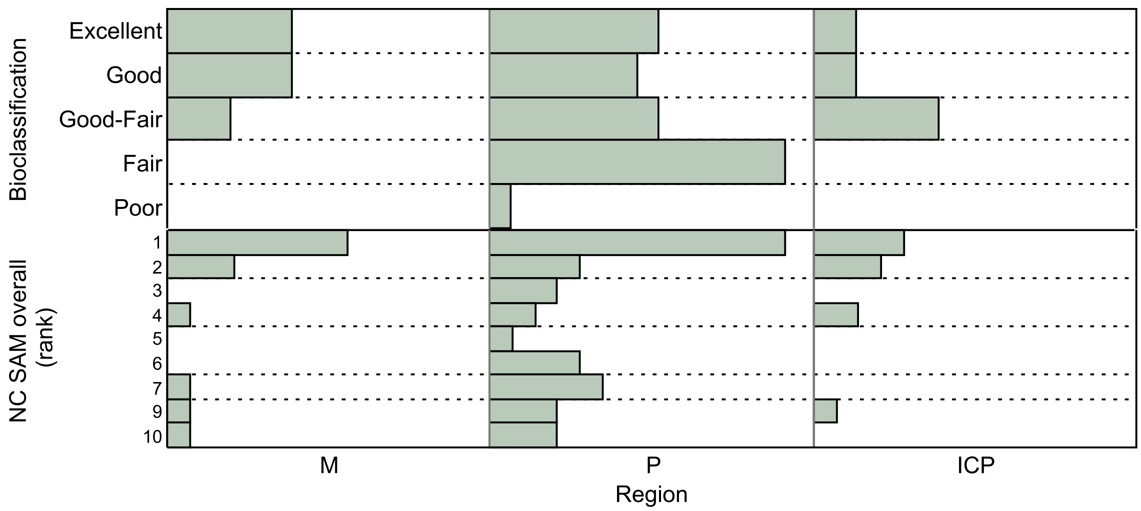


Figure 3-17 Distributions of bioclassifications and NC SAM overall function ranks by region for 63 study sites. Best instream conditions are indicated by Excellent bioclassification and NC SAM rank of 1.

Direct comparison of bioclassification to watershed slope and TIA (Figure 3-18) strongly suggested that when watersheds exceeded the previously identified threshold of 4% TIA, it was extremely likely that benthic macroinvertebrate communities would exhibit extreme stress, indicated by a bioclassification of Fair or Poor. One exception was noted: site BB312 (E. Fork Deep R. near Greensboro), which received a bioclassification of Good-Fair when it was last sampled in 2008. Field notes from the current study indicated that this site had very poor habitat, extremely widespread bank failures and erosion, and that the bed had been scoured to bedrock by high flows. Previous bioclassifications for this site from 1993, 1998, and 2003 rated this site as Fair. These results suggest that the latest bioclassification from may not have accurately represented conditions encountered during site visits in 2013.

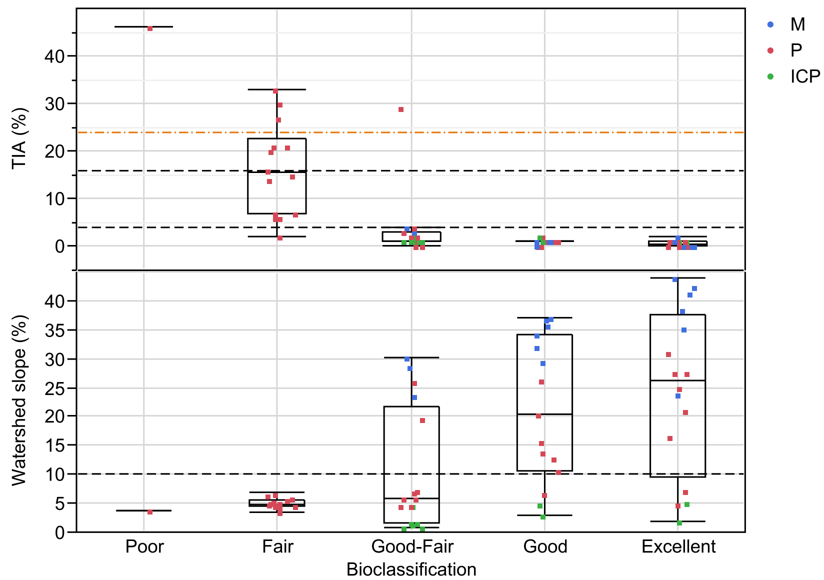


Figure 3-18 Distributions of watershed slope (%) and TIA (%) by bioclassification. Marker color indicated region of the state where sampling site was located. Dotted lines represent previously identified thresholds of 10% slope, 4% TIA, and 16% TIA, above which watershed hydrologic responses (runoff ratio or stream flashiness) are affected. The brown dotted line indicates the TIA threshold used by regulatory stormwater programs.

Bioclassifications also exhibited a predictable response to the previously identified threshold of 10% watershed slope, with watersheds above that value consistently rating Good-Fair or better. For watersheds below that value, the bioclassification was determined by TIA. This reflects the linkage seen between TIA and slope in determining the runoff ratio (see Section 3.2), where runoff ratio exhibited significant positive correlations with slope, and stream flashiness was consistently moderate or low when slopes were >10%. Higher TIA was only seen in very low slope watersheds and so for slopes <10%, the watershed hydrologic responses were more affected by TIA, with no significant correlation between TIA and watershed hydrologic responses when TIA was <4%.

The competing factors of TIA and watershed slope on bioclassifications were obvious when bioclassification and runoff ratio were directly compared (Figure 3-19). Higher runoff ratios were associated with both the Excellent and the Poor/Fair bioclassifications, with Excellent bioclassifications associated with higher slope watersheds with low TIA, and the stressed communities associated with low slope watersheds with high TIA. In both cases, a large proportion of the precipitation received by the watersheds was converted to stream discharge, but the rate of that conversion (as indicated by stream flashiness) was the key difference, with higher levels of TIA resulting in water traveling by a relatively quick hydrological pathways, such as overland flow or through engineered systems, that encourages rapid delivery of the relatively large volume of water to streams in response to storm events. Previous results from the water quality analysis (Section 3.4) suggest that the water transferred is also likely to be of poor quality.

Bioclassification showed a more monotonic relationship with stream flashiness (Figure 3-20), and again reflects the combined influence of slope and TIA on both response variables. Similarly to the flashiness/slope scatterplot (Figure 3-5), watersheds with slopes >10% appeared to have an upper boundary to stream flashiness, the Good and Excellent bioclassifications had very similar flashiness values. Both bioclassifications had a median T_{Qmean} of 0.32. The similarities of results for the runoff ratio, stream flashiness, and bioclassification suggest that all three variables respond similarly to an anthropogenic stressor, TIA and to a physical watershed characteristic, slope.

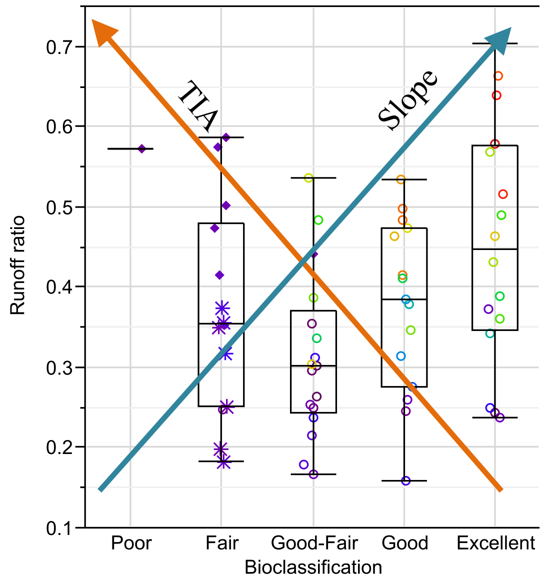


Figure 3-19 Distributions of runoff ratio by bioclassification. Marker color denotes watershed slope (%) and marker shape denotes TIA group (<4%; 4-16%; >16%). Arrows indicate conceptual directions of influence of TIA (brown) and slope (blue) on bioclassifications.

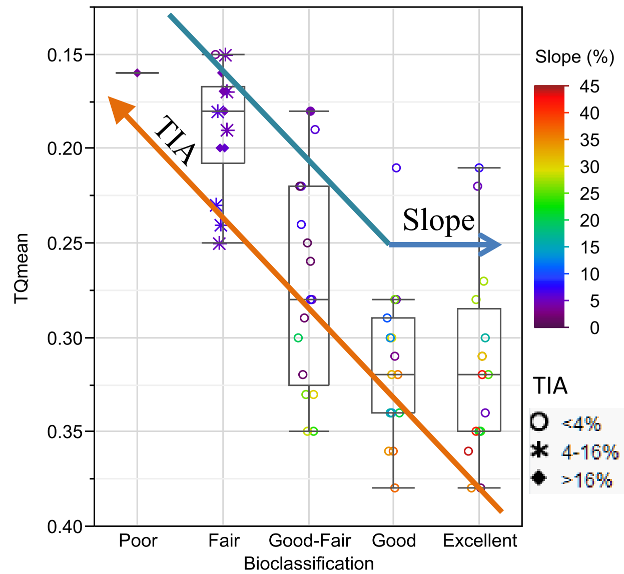


Figure 3-20 Distributions of T_{Qmean} by bioclassification. Marker color denotes watershed slope (%) and marker shape denotes TIA group (<4%; 4-16%; >16%). Arrows indicate conceptual directions of influence of TIA (brown) and watershed slope (blue) on bioclassifications.

The NC SAM overall rank did not show the expected response to TIA or watershed slope, nor did it seem to have a predictable relationship with either the runoff ratio or stream flashiness as represented by T_{Qmean} and RBI (data not shown), with the exception of all sites receiving the highest NC SAM rank of 1 having low levels of TIA (<4%). Comparison of median runoff ratio, RBI, and T_{Qmean} by NC SAM rank using the Wilcoxon method found very few differences between ranks; only rank = 6 showed significant differences with more than one other rank. Rank 6 had a significantly larger runoff ratio than rank 9 ($p = 0.0195$) and significantly lower runoff ratios than ranks 1 ($p = 0.0017$), 2 ($p = 0.0318$) and 3 (0.0497). It would be anticipated that moderate ranks would have lower runoff ratios than both the high and low rank values, and that pattern was not evident in the data. These results suggest that NC SAM does not exhibit the same strong coupling with watershed hydrologic responses as seen with bioclassification.

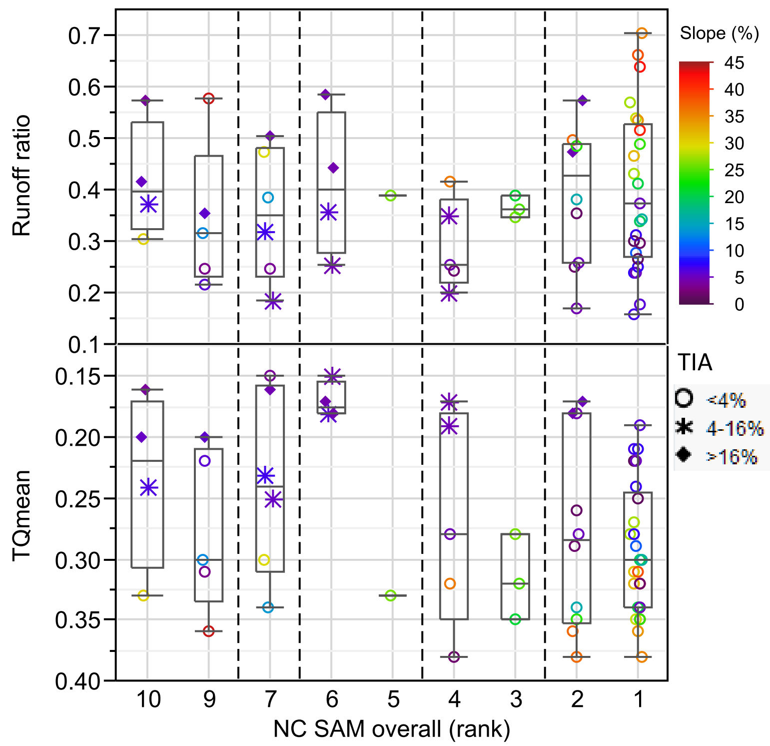


Figure 3-21 Distributions of runoff ratio and stream flashiness (TQmean) by NC SAM overall function rank value, with smaller rank values associated with better stream function. Dotted lines partition the NC SAM ranks into groups that were assumed to be roughly equal to bioclassifications (from left to right, Poor, Fair, Good-Fair, Good, and Excellent).

3.6 Flow path network structure, basin morphometry and land cover

Results thus far have focused on watershed slope and land cover as significant drivers for watershed hydrologic responses, as well as for water quality, channel dimensions, habitat, and biological community conditions at the watershed outlet. This final results section will examine relationships between the watershed stream network structure, as represented by the geomorphic width function and in the context of the geomorphic instantaneous unit hydrograph (GIUH) theory, and spatial patterns of land use on watershed hydrologic responses using hydrologic distance from the watershed outlet, and will complete assessment of *Objective 2* from *Section 1.4*.

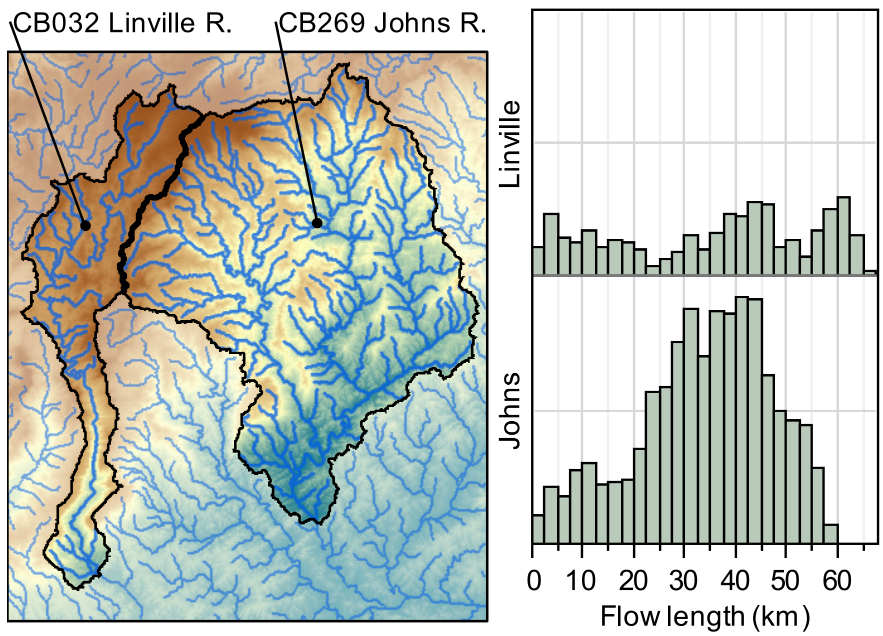


Figure 3-22 Examples of basin morphometry and associated width functions for two study watersheds.

Examples of the relationship between the width function and watershed morphometry are presented in Figure 3-22 using two watersheds assessed in this study, CB032- Linville R. (Watershed A) and CB269- Johns R. (Watershed B). The Linville R. (Watershed A, on left in map) has a more complex, multi-modal frequency distribution with a larger interquartile range (IQR), whereas the Johns R. (Watershed B, on right in map) has a smoother, more regular frequency distribution with a much smaller IQR. It would therefore be anticipated that these two watersheds would transfer precipitation to runoff differently. In the case of the Linville R., stream discharge would be anticipated to arrive at the watershed outlet over an extended period of time, resulting in less flashy stream flows. The width function for the Johns R. would suggest that half of the precipitation would be delivered to the watershed outlet in a much shorter time span based on its much smaller width function IQR of its width function, suggesting flashier flows at the watershed outlet. Since the instream conditions at the watershed outlet reflect the net effects of variable stream discharge events over time, it seems a natural extension of the GIUH theory that the width function would also predict the instream conditions from the cumulative effects of many storm events.

Several moments of the width function within each watershed were examined: mean, kurtosis, skewness, and IQR. Since the magnitude of the mean and IQR would be related to drainage area, both of these values were standardized through rescaling by dividing each watershed's mean and IQR by its longest flow path to control for watershed size. The standardized mean (s-mean) and standardized IQR (s-IQR) were found to be acceptable alternative measures of skewness and kurtosis, respectively. One main advantage of the s-mean and s-IQR over the traditional statistical moments was that they were more intuitive, particularly due to the limitation of possible values to the range of 0-1, and therefore more likely to be easily interpretable by water quality or watershed managers. The s-mean and skewness were strongly negatively correlated (ρ = -0.947, p < 0.0001), as were the s-IQR and kurtosis (ρ = -0.739, p < 0.0001).

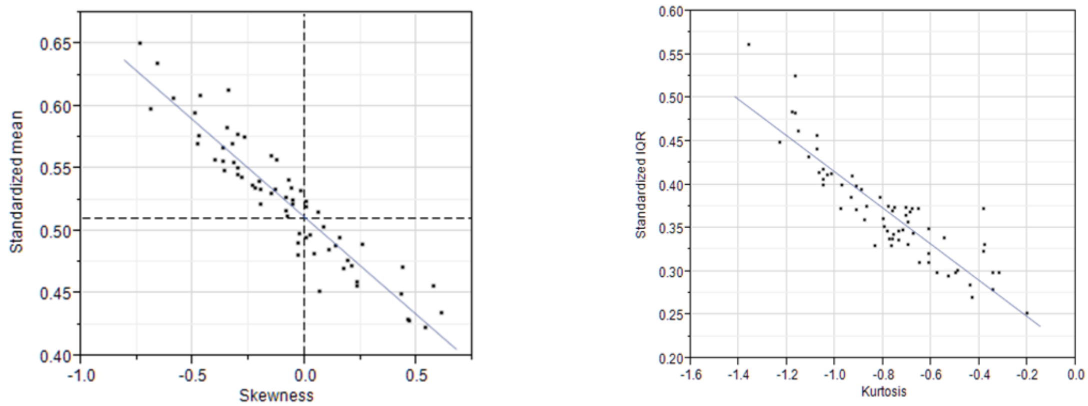


Figure 3-23 Scatterplots of standardized mean (s-mean) and skewness (left) and the standardized IQR (s-IQR) and kurtosis (right) of watershed width functions. The line shown is from the Pearson regression, but non-parametric Spearman correlation analyses resulted in significant correlations.

Based on the scatterplot, negative skewness was associated with S-mean values >0.51, and s-means <0.51 were generally associated with positive skewness. S-mean values for the study watersheds had a range of 0.42 – 0.65, with a median of 0.53. High s-mean values would suggest a watershed morphometry where there were few short flow paths and a dominance of longer flow paths, implying that runoff from within the watershed would be routed through a relatively small number of flow pathways near the watershed outlet, resulting in a bottleneck that would be conducive to flashy stream flows.

All watershed width functions had negative kurtosis and the corresponding s-IQR values ranged from 0.25 – 0.56 (median 0.37). Smaller s-IQR values corresponded with higher kurtosis values, and so would suggest more “peakedness” in the distributions, or relatively high frequency of similar flow path lengths within a relatively narrow range. Extrapolating from GIUH theory, watersheds with width functions that had smaller s-IQR values would therefore result in flashier stream flow regimes. As examples, the elongated

Linville R. watershed presented earlier had one of the highest s-IQR's (0.54; >97th percentile) whereas the more rounded Johns R. had one of the lowest (0.30; equal to the 10th percentile).

Since lower values of the s-IQR would suggest more rounded basins, they would be anticipated to be associated with higher values of the BFF, and these watersheds would be anticipated to have flashier stream flows. Higher s-IQR and lower BFF values would represent more elongated basins with a lower tendency for stream flashiness. The BFF for watersheds in this study ranged from 0.0358 – 0.4540, with a median of 0.1581. The previous watershed examples (Figure 3-22) had BFF values of 0.0404 (CB032, Linville R.) and 0.1500 (CB269, Johns R.). The highest BFF value was associated with the most upstream assessment site on Goose Cr., near Charlotte, (QB351) (BFF = 0.4540). This site was nested within two other study watersheds, and when combined they provide an example of the basin elongation that occurs as drainage area increases, as BFF dropped from 0.4540 to 0.2313 and finally to 0.1312, going from upstream to downstream, though this was not a universal finding for all nested study watersheds.

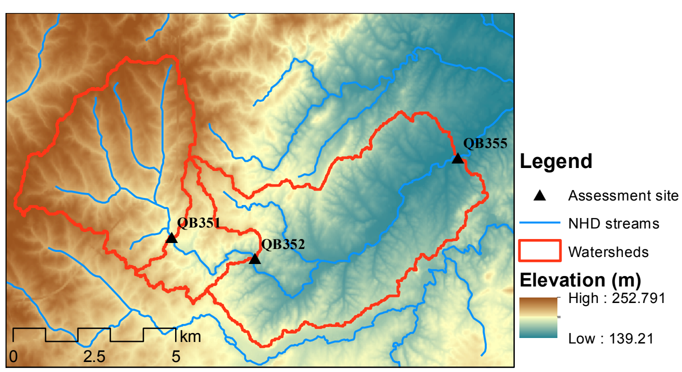


Figure 3-24 Nested Goose Cr. watersheds near Charlotte, NC provide an example of basin elongation as drainage area increases. The basin form factor (BFF) decreases as you move from upstream to downstream from 0.4540 (QB351) to 0.2313 (QB352) to 0.1312 (QB355).

Watershed slope was compared to BFF, s-mean, and s-IQR and no significant correlations were identified, suggesting that slope was an independent measure of basin structure (data not shown). BFF was compared to watershed hydrologic responses (Figure 3-25), s-mean, and s-IQR (<10% or >10%) (Table 3-7). The BFF showed no correlation with the runoff ratio, as was anticipated, as the overall efficiency of a watershed in terms of transfer of precipitation to stream flow is more strongly driven by factors such as land cover, soils, and watershed slope. The BFF was anticipated to be more closely linked to stream flashiness and did show significant correlations with RBI and TQmean, with higher BFF values (i.e., rounder basins) associated with higher stream flashiness (e.g., higher RBI and lower TQmean values), as expected. The BFF also exhibited the expected significant negative correlation with s-IQR. The relationship between s-mean and BFF was monotonic, with a significant positive correlation (Table 3-7). This finding was as expected, since high s-means as well as high BFF values would be more likely to produce flashy flows. Comparison of s-mean and s-IQR directly to watershed hydrologic responses (Table 3-8, Figure 3-26) resulted in significant correlations only between the s-mean and stream flashiness, suggesting that this characteristic of the width function may be a more accurate predictor for stream flashiness than s-IQR.

These findings demonstrate that there was a detectable linkage between watershed morphometry and the specific organization of flow paths within the watershed, which supports the hypothesis that the GIUH theory may serve as an underlying mechanism that explains the empirically-derived relationships between basin form and stream flow regimes that has been commonly used in classic watershed hydrology. The basin form has also been tied to instream benthic macroinvertebrate community condition and water quality in NC (Potter et al. 2004), providing additional evidence of the linkages between watershed hydrologic processes and instream ecological conditions.

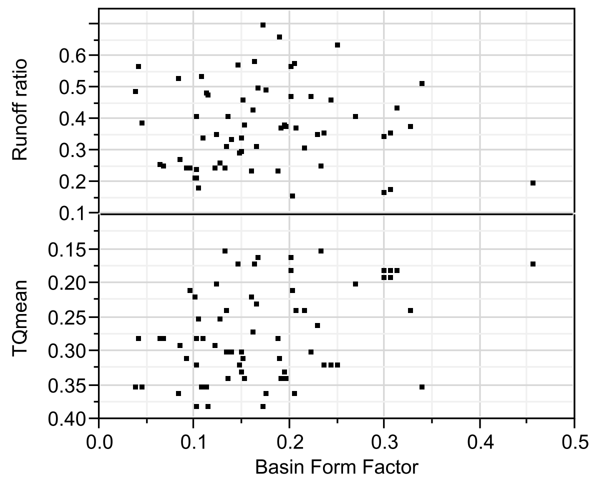


Figure 3-25 Scatterplots of the Basin Form Factor (BFF) and watershed runoff ratio and flashiness (T_{Qmean}).

Table 3-7 Results of Spearman correlations between Basin Form Factor, and watershed hydrologic responses and width function metrics.

Response variable	ρ	p
Runoff ratio	0.110	0.3838
RBI	**0.409**	0.0007
T_{Qmean}	**-0.306**	0.0132
s-IQR	**-0.271**	0.0293
s-Mean	**0.368**	0.0026

Table 3-8 Results of Spearman correlation analyses for watershed hydrologic response variables (runoff ratio, RBI flashiness, TQmean flashiness) and summary statistics of the watershed width functions.

Hydrologic Response	s-Mean		s-IQR	
	ρ	p	ρ	p
Runoff ratio	-0.115	0.3624	-0.231	0.0644
RBI	**0.307**	0.0128	0.171	0.1728
T_{Qmean}	**-0.253**	0.0423	-0.170	0.1747

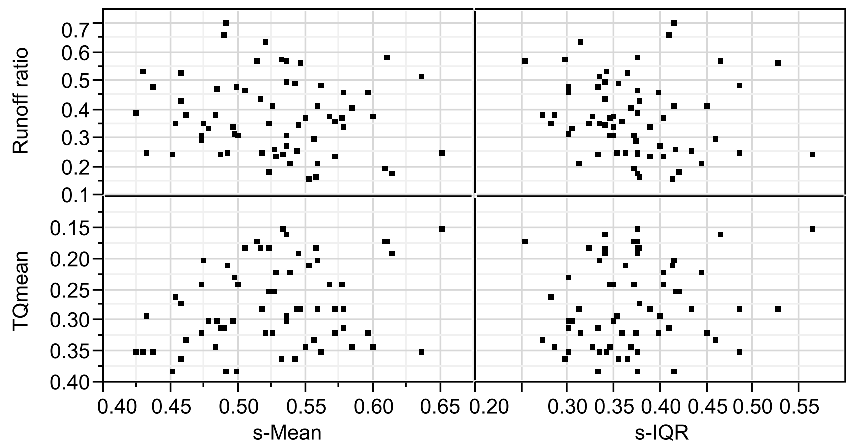
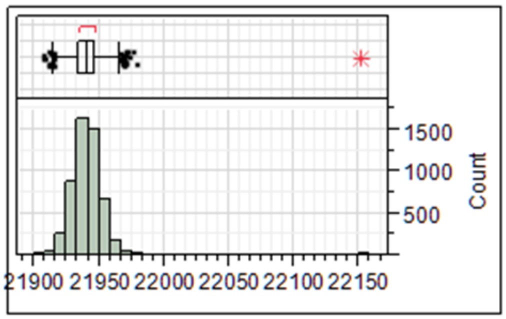


Figure 3-26 Scatterplots of width function summary statistics (standardized mean [s-mean]) and standardized IQR [s-IQR]) and watershed hydrologic responses (runoff ratio, RBI flashiness, T_{Qmean} flashiness).

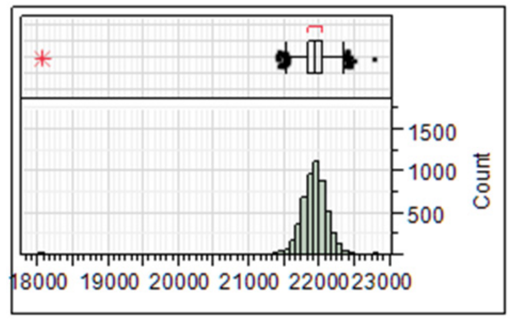
This initial watershed width function analysis identified significant correlations with watershed hydrologic responses, but it did not take into account heterogeneity of watershed surficial conditions. Land cover has a profound effect on the speed and efficiency of precipitation transfer to stream discharge, and is the basis of many of the hydrologic modeling methods currently in use, for example, for predicting storm runoff potential using the curve number or rational method, with higher values assigned to the land cover coefficients for modified land cover, such as development or agriculture, that have lower infiltration rates (U.S. Department of Agriculture, Natural Resources Conservation Service 1986).

The final set of analyses examined the importance of the spatial proximity and concentration of a given land cover type within watersheds. The width function metrics for each major land cover type were used, but in this case the s-mean and s-IQR represented the relative dispersion or clustering within the watershed. Low s-IQR corresponded to high kurtosis and strong clustering of the land cover type. Low s-mean suggested that the land cover type was predominantly situated closer to the watershed outlet. Spatial location within the watershed as compared to its location if the land use was randomly located with the watershed was determined by comparison to the Monte Carlo (MC)-derived frequency distributions (see Section 2.4).

To illustrate these concepts, results for forest and development from site AB021 (First Broad River) are shown in Figure 3-27. This watershed was located in the southern Piedmont area of the state, and was primarily forested (84%), with some agriculture (8%) and development (2%), with smaller amounts of other land cover classes (wetlands, herbaceous, barren, and water). However, the forest cover tended to be located in the upper portions of the watershed (indicated by mean path length for forested pixels being >90th percentile of the MC-derived distribution) and development was located closer to the watershed outlet (indicated by the mean distance of developed pixels being <10th percentile of the MC distribution). It would be anticipated that any hydrologic benefits provided by forest cover would be degraded due to its long distance from the watershed outlet, and the impact of development would be greater due to its spatial proximity to the outlet.



(a) Forested land use



(b) Developed land use

Figure 3-27 Examples of results from Monte Carlo analysis for site AB021 (First Broad River) and mean distance from watershed outlet for forest (a) and development (b). Red asterisk indicates actual mean distance of the land cover class from the watershed outlet.

This site had some of the most extensive forest cover and minimal developed land use among Piedmont sites (Figure 3-28). The last bioclassification for this site was Good, whereas other sites with less forest and/or more development rated Excellent, suggesting that more than the absolute percentage of land cover should be considered when examining its role as an environmental driver for instream conditions.

The spatial relationships of watershed flow path lengths (FPL), mean MC FPL, mean developed FPL, and actual location of raster cells assigned to the developed land cover class are shown in Figure 3-29. The purple and pink colors indicate the areas associated with the mean MC FPL and mean developed FPL and represent the hydrologic flow path centroids for, respectively, the watershed and the developed land cover class.

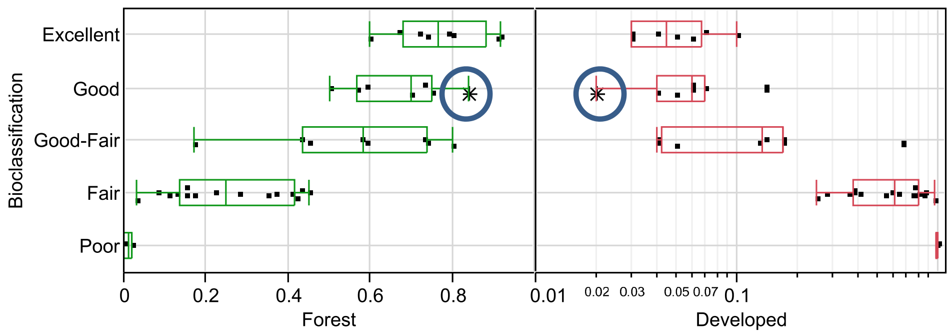


Figure 3-28 Bioclassification and corresponding percent forest and development cover for Piedmont sites. Site AB021 (First Broad River) is indicated with asterisks (circled). In spite of high levels of forest and low levels of development, this site received a bioclassification of Good, whereas sites with less forest and more development rated Excellent.

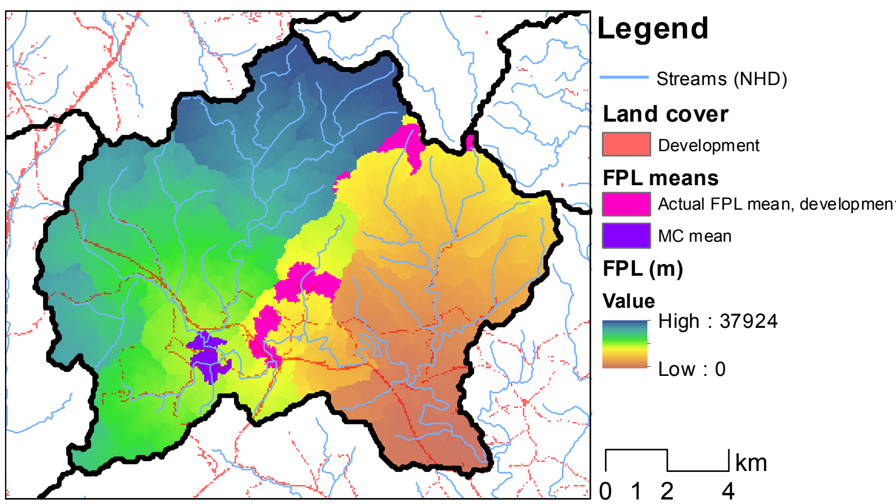


Figure 3-29 Site AB021, First Broad R. as an example of watershed flow path lengths (FPL), the mean developed FPL, the MC simulation-derived FPL mean, and spatial relations to the location of development and the stream network.

Comparison of actual mean FPL to the MC mean FPL showed that land use was not randomly distributed within the study watersheds (Table 3-9). Forested land use was slightly more common in the upper portions of these watersheds; development occurred approximately equally in the upper and lower portions of the watersheds; and agriculture was more likely to occur in the lower portion of the watersheds, similar to findings by Rice and Emanuel (2014). Only four watersheds had any land cover class fall within the random-equivalent range of the 10^{th}-90^{th} percentile of the watershed's MC distribution. Fewer sites were evaluated for agriculture (n = 63) since two of the study watersheds did not contain this land cover type.

Table 3-9 Study watersheds in which land cover classes were located closer than predicted by MC simulation ("near"), further than predicted by MC simulation ("far"), or an equivalent distance as the MC simulation ("random").

Land use	Near	Far	Random
Forest	26	39	0
Development	32	31	2
Agriculture	41	20	2

For each land cover type, correlations were run between the watershed hydrologic responses (runoff ratio, flashiness), s-mean, and s-IQR using all watersheds and grouped by *near* and *far* watersheds. The analyses were repeated using the actual s-mean and s-IQR and the MC-derived s-mean and s-IQR to determine if the actual conditions would show improved correlations as compared to randomly arranged land use. Results were summarized by development and forest land cover classes (Table 3-10).

The most consistent improvements in correlations were seen for s-mean for development land cover class. For all three response variables—runoff ratio, RBI, and T_{Qmean}—correlations not only improved when the actual land use configuration was considered, but all were significant for the *near* watersheds only, and correlation strength decreased for all of the *far* watersheds, demonstrating that developed land use had a more detectable effect on watershed hydrology when situated closer to the watershed outlet, such that when development was spatially closest (low s-mean in *near* watersheds), runoff ratio increased but flashiness decreased. Forested land cover for the *far* watersheds mirrored the results seen for developed land cover, with correlations between s-mean and watershed hydrologic responses for forest cover being identical to developed land cover in terms of direction, though the strength of correlations were slightly lower for forest cover.

Table 3-10 Summary of correlations between s-IQR, s-mean, and watershed hydrologic responses for actual land cover configurations and randomized land use configuration.

Response	**Location**	**Land cover class s-IQR**				**Land cover class s-Mean**			
		Actual		**MC**		**Actual**		**MC**	
		ρ	**p**	**ρ**	**p**	**ρ**	**p**	**ρ**	**p**
Land cover class: **DEVELOPMENT**									
Runoff ratio	near	-0.293	0.1033	-0.161	0.3789	**-0.359**	0.0435	-0.312	0.0822
	far	-0.306	0.0945	-0.314	0.0857	0.044	0.8161	0.204	0.2719
RBI	near	0.323	0.0710	**0.488**	0.0046	**0.499**	0.0037	**0.426**	0.0150
	far	-0.326	0.0732	-0.078	0.6788	0.123	0.5110	0.194	0.2946
T_{Qmean}	near	**-0.399**	0.0235	**-0.398**	0.0242	**-0.451**	0.0096	-0.228	0.2093
	far	0.177	0.3397	0.012	0.951	-0.148	0.4259	-0.256	0.1641
Land cover class: **FOREST**									
Runoff ratio	near	-0.200	0.3273	**-0.497**	0.0098	-0.250	0.2182	-0.100	0.6263
	far	0.025	0.8785	-0.069	0.6763	**-0.317**	0.0493	-0.133	0.4203
RBI	near	0.038	0.8539	-0.235	0.2486	-0.085	0.6791	0.290	0.1501
	far	-0.031	0.8495	**0.324**	0.0440	**0.364**	0.0226	0.292	0.0710
T_{Qmean}	near	0.010	0.9615	0.190	0.3533	0.191	0.3489	-0.219	0.2819
	far	0.167	0.3111	-0.227	0.1647	**-0.312**	0.0533	-0.184	0.2632

The observed decreasing flashiness associated with increasing developed land cover in *near* watersheds and forest cover in *far* watersheds seemed counterintuitive, but this analysis did not control for the overall fraction of any given land use within each watershed, only the relative distance from the watershed outlet. Further comparison of the characteristics of the *near* and *far* watersheds indicated that *near* watersheds had significantly more forest cover ($Z = -3.2114$, $p = 0.0013$; median *near*: 71%, median *far*: 17%), significantly higher slopes ($Z = -2.653$, $p = 0.0080$; median *near*: 18%, median *far*: 5%), and significantly less development ($Z = 2.472$, $p = 0.0134$; median *near*: 6%, median *far*: 10%). Based on these differences between the *near* and *far* watersheds, it is likely that results are due more to the influence of watershed slope rather than land use, as higher slopes were previously demonstrated to be associated with higher runoff ratios (Figure 3-2, Figure 3-5), low to moderate flashiness (Figure 3-2, Figure 3-5), and higher forest cover (Figure 3-6). Also, there was also only one significant correlation between s-IQR (which represents the relative clustering of the land cover class) and watershed hydrologic responses, though several significant correlations were identified between the MC-derived s-IQR and watershed hydrologic responses. As the MC-derived statistics were found to be equivalent to the statistics derived from the whole-watershed width function, these relationships would be attributable to the watershed morphometry characteristics previously discussed. In summary, these results were inconclusive in terms of identifying a clear linkage between the hydrologic distance of land cover from the watershed outlet and watershed hydrologic responses.

CHAPTER 4. CONCLUSIONS AND IMPLICATIONS

This thesis describes the coupling of reach-level stream function and watershed function, and the same factors, whether natural or anthropogenic, were associated with similar measurable responses for water quality, instream habitat, health of biological communities, and simple response measures of watershed hydrologic processes. Natural conditions, such as watershed slope or characteristics of the geomorphic width function, and anthropogenic impacts, such as increases in impervious surface in the watershed resulted in similar responses in multiple stream and watershed hydrologic function indicators. Degradation of instream conditions parallels degradation of watershed hydrologic functions, and therefore water resource management and rehabilitation of impaired streams must necessarily address the dysfunctions present in the hydrologic functions within the watershed.

The low threshold of TIA (4%) identified in this study is similar to thresholds identified by other researchers. These results suggest that current regulatory policies that rely on the 24% TIA threshold are not sufficiently protective of stream and watershed function. Stormwater management programs of varying extents have been in place in NC and the rest of the U.S. since the original 1972 Clean Water Act legislation and the 1987 amendments that addressed non-point source pollutants. That degradations of stream and watershed conditions associated with TIA are still detectable after several decades of non-point source management is further evidence that current stormwater management activities are providing insufficient protection against degradation of watershed, and therefore stream, hydrologic processes. This may be due to the primary focus in stormwater management on reducing peak flows due to excess surface runoff based on typical or idealized storm events. These results suggest merely addressing runoff generation may not be sufficient to mitigate for the multiple effects to watershed hydrologic processes that occur due to increases in TIA and corresponding losses of vegetation; examples include less available soil surface, decreased infiltration due to compaction and disturbance of soils, lower interception of precipitation due to vegetation loss, changes to ET rates, and lowered capacity for storage and groundwater recharge. Alternatively, the current benefits provided by stormwater management may essentially be negated by the persistent effects of pre-CWA development. Finally, it should be noted that while the effects of TIA on multiple stream and watershed response variables were detectable in this study, the data used represented a snapshot of recent conditions and these may, in fact, represent an improvement over past conditions. Additional analyses would be required to determine if there was a temporal trend of improved conditions; this would certainly be a feasible project given the extensive history of stream monitoring in NC.

Water resource management has been slowly shifting focus towards management of the watershed over the last several decades. The low-hanging fruit—point source discharges—has largely been addressed. The current focus is on non-point sources and the effects of excess surface runoff. However, these results suggest that current approaches may be insufficient, and additional dysfunctions in watershed hydrologic processes exist that are not being addressed.

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APPENDICES

Appendix A: Study sites

Study site locations are given in Table A.1, including project site ID, stream name, location (latitude/longitude in decimal degrees), NC SAM zone, level IV ecoregion code (L4 eco), drainage area (DA, km^2), impairment status (303(d); "X" indicates impaired), USGS site number, NC Div. of Water Resources Ambient Monitoring System (AMS) site number, and season(s) sampled (S12: summer 2012; W13: winter 2013; S13: summer 2013). Ecoregion codes and corresponding names are provided at the end of the appendix.

Table A-1 Study site locations

Site ID	Stream name	Latitude	Longitude	NCSAM zone [1]	L4 eco[2]	DA (km^2)	303(d)	USGS site #	AMS site #	S12	W13	S13
AB021	First Broad R	35.4931	-81.6822	P	45e	156	X	02152100	A4800000			X
AB044	Second Broad R	35.4042	-81.8719	P	45a	225		02150495	A2700000			X
BB126	NE Cape Fear R	34.8278	-77.8333	ICP	63h	1564		02108000				X
BB146	Morgan Cr	35.9236	-79.1156	P	45c	21		02097464		X		X
BB218	Flat Cr	35.1828	-79.1783	ICP	65c	20		02102908				X
BB238	New hope Cr	35.8847	-78.9667	P	45g	197	X	02097314	B3040000	X		X
BB241	Cane Cr	35.9869	-79.2064	P	45c	20		02096846		X		X
BB312	E Fork Deep R	36.0375	-79.9458	P	45b	38	X	02099000	B4240000		X	X
BB362	Reedy Fk	36.1728	-79.9533	P	45e	53	X	02093800				X

Table A-1 Continued

Site ID	Stream name	Latitude	Longitude	NCSAM zone [1]	L4 eco[2]	DA (km^2)	303(d)	USGS site #	AMS site #	S12	W13	S13
BB406	S Buffalo Cr	36.0597	-79.7256	P	45b	89	X	02095000	B0750000			X
BB407	N Buffalo Cr	36.1203	-79.7083	P	45b	96	X	02095500	B0540000			X
BB427	Horsepen Cr	36.1364	-79.8608	P	45e	38	X	0209399200				X
CB011	Catawba R	35.6858	-82.0611	M	66l	328		02137727	C0250000			X
CB032	Linville R	35.7947	-81.8903	P	45e	174		02138500	C1000000			X
CB133	Gar Cr	35.3611	-80.8981	P	45b	9		0214266080			X	X
CB139	Mcdowell Cr	35.3897	-80.9214	P	45b	69	X	0214266000				X
CB146	L Sugar Cr	35.085	-80.8822	P	45b	127	X	02146530	C9210000			X
CB192	Jacob Fk	35.5906	-81.5672	P	45e	66	X	02143040	C4370000			X
CB224	Long Cr	35.3056	-81.2322	P	45b	81	X	02144000	C5900000			X
CB269	Johns R	35.8339	-81.7117	P	45e	521		02140991				X
CB319	L Sugar Cr	35.2051	-80.8369	P	45b	30	X	02146409				X
EB090	French Broad R	35.6089	-82.5781	M	66j	2446	X	03451500	E4280000			X
EB145	Swannanoa R	35.5683	-82.5450	M	66j	337		03451000	E4170000		X	X
EB167	Mills R	35.3986	-82.595	M	66j	172	X	03446000	E1490000			X
EB194	French Broad R	35.7861	-82.6608	M	66j	3449	X	03453500				X
EB250	Pigeon R	35.7853	-83.1131	M	66g	1398		03460795	E6500000			X
EB273	W Fork Pigeon R	35.3961	-82.9381	M	66d	73		03455500			X	X

Table A-1 Continued

Site ID	Stream name	Latitude	Longitude	NCSAM zone [1]	L4 eco[2]	DA (km^2)	303(d)	USGS site #	AMS site #	S12	W13	S13
EB294	S Toe R	35.8311	-82.1844	M	66d	112	X	03463300	E8200000		X	X
EB320	Cataloochee Cr	35.6672	-83.0728	M	66g	127		03460000	E6450000			X
FB010	Valley R	35.1389	-83.9806	M	66j	268	X	03550000	F4000000			X
GB011	Oconaluftee R	35.4614	-83.3536	M	66g	476		03512000				X
GB042	Nantahala R	35.1269	-83.6192	M	66d	135		03504000				X
IB039	Lumber R	34.4431	-78.9606	ICP	65p	3205	X	02134500	I5690000		X	X
JB007	Eno R	36.0719	-78.9081	P	45g	367		02085070	J0770000	X		X
JB018	Little R	36.1417	-78.9206	P	45c	203		0208521324	J0820000	X		X
JB037	Crabtree Cr	35.8111	-78.6111	P	45f	314	X	02087324				X
JB052	Swift Cr	35.7192	-78.7531	P	45f	50	X	02087580				X
JB055	Swift Cr	35.5747	-78.4989	P	45f	294	X	0208773375				X
JB063	Walnut Cr	35.7583	-78.5833	P	45f	77	X	02087359				X
JB106	Nahunta Swp	35.4889	-77.8061	ICP	65m	207		02091000				X
JB156	Crabtree Cr	35.8453	-78.7247	P	45f	198	X	0208726005	J3000000	X		X
JB165	Ellerbe Cr	36.0589	-78.8331	P	45g	55	X	02086849	J1330000	X	X	X
JB291	Pigeon House Br	35.8039	-78.6189	P	45f	11	X			X		X
KB003	S Fork New R	36.3931	-81.4072	M	66c	530		03161000	K3250000	X		X
LB012	Watauga R	36.2383	-81.8228	M	66d	236		03479000	L4700000		X	X

Table A-1 Continued

Site ID	Stream name	Latitude	Longitude	NCSAM zone [1]	L4 eco[2]	DA (km^2)	303(d)	USGS site #	AMS site #	S12	W13	S13
NB008	Dan R	36.5147	-80.3031	P	45e	315	X	02068500	N0150000			X
NB028	Mayo R	36.5355	-79.9906	P	45e	672		02070500	N1400000			X
NB076	Cashie R	36.0478	-76.9856	ICP	63n	296		0208111310			X	X
OB056	Swift Cr	36.1117	-77.9211	ICP	65m	428		02082770	O3870000			X
OB058	Tar R	35.9542	-77.7872	ICP	65m	2405		02082585	O3180000			X
OB090	Tar R	35.8939	-77.5333	ICP	65p	5846		02083500	O5250000			X
OB101	Fishing Cr	36.1508	-77.6931	ICP	65p	1373		02083000	O4680000		X	X
OB107	Chicod Cr	35.5456	-77.2106	ICP	63e	118		02084160	O6450000		X	X
PB004	New R	34.8489	-77.5197	ICP	63h	215		02093000	P0600000		X	X
QB042	Roaring R	36.2497	-81.0442	P	45e	329		02112120	Q0660000	X		X
QB055	Yadkin R	36.1525	-81.1458	P	45e	1302		02112000		X		X
QB088	Mitchell R	36.3114	-80.8067	P	45e	205		02112360		X		X
QB112	Yadkin R	36.2417	-80.8469	P	45e	2246	X	02112250	Q0810000	X		X
QB118	Ararat R	36.4044	-80.5619	P	45e	596	X	02113850	Q1780000			X
QB206	Hunting Cr	36	-80.7456	P	45e	401	X	02118500	Q3484000			X
QB215	N Second Cr	35.7181	-80.5956	P	45b	305	X	02120780	Q4120000		X	
QB223	S Yadkin R	35.8444	-80.6594	P	45b	791	X	02118000	Q3460000			X
QB233	Abbotts Cr	35.8067	-80.2353	P	45b	457	X	02121500	Q5930000			X

Table A-1 Continued

Site ID	Stream name	Latitude	Longitude	NCSAM zone [1]	L4 eco[2]	DA (km^2)	303(d)	USGS site #	AMS site #	S12	W13	S13
QB351	Goose Cr	35.1306	-80.6312	P	45c	23	X	0212467451	Q8360000	X		X
QB352	Goose Cr	35.125	-80.6028	P	45c	29	X	0212467595				X
QB355	Goose Cr	35.1539	-80.535	P	45c	63	X	02124692				X
QB403	Little R	35.3864	-79.8322	P	45c	274		02128000	Q9200000			X
QB448	Yadkin R	35.9917	-81.5581	M	66l	74		02111000				X

[1] NC SAM zones (NC Stream Functional Assessment Team 2013)

Code	Zone name
M	Mountains
P	Piedmont
ICP	Inner Coastal Plain

[2] Ecoregion codes and names (Griffith et al. 2002)

Code	Level III name	Level IV name	Number of sites
45a	Piedmont	Southern Inner Piedmont	1
45b	Piedmont	Southern Outer Piedmont	11
45c	Piedmont	Carolina Slate Belt	7

45e	Piedmont	Northern Inner Piedmont	14
45f	Piedmont	Northern Outer Piedmont	7
45g	Piedmont	Triassic Basins	3
63e	Mid-Atlantic Coastal Plain	Mid-Atlantic Flatwoods	1
63h	Mid-Atlantic Coastal Plain	Carolina Flatwoods	2
63n	Mid-Atlantic Coastal Plain	Mid-Atlantic Floodplains and Low Terraces	1
65c	Southeastern Plains	Sand Hills	1
65m	Southeastern Plains	Rolling Coastal Plain	3
65p	Southeastern Plains	Southeastern Floodplains and Low Terraces	3
66c	Blue Ridge	New River Plateau	1
66d	Blue Ridge	Southern Crystalline Ridges and Mountains	4
66g	Blue Ridge	Southern Metasedimentary Mountains	3
66j	Blue Ridge	Broad Basins	5
66l	Blue Ridge	Eastern Blue Ridge Foothills	2

Appendix B: Results for watershed and instream characteristics

The following table summarizes the associated watershed and instream characteristics collected or developed for this project, including mean watershed slope (Slp; % change), runoff ratio (RR), stream flashiness (RBI, T_{Qmean}), bankfull width and height (BF wd, BF ht; feet); top-of-bank width and height (TB wd, TB ht; feet); instream habitat assessment score, and the basin form factor (BFF). Characteristics of the watershed width function include mean and maximum flow path lengths (mean FPL, max FPL; meters), skewness (Skew), kurtosis (Kurt), standardized mean (s-mean), standardized IQR (s-IQR).

Table B-1 Watershed and instream characteristics for study sites

Site	Slp	RR	RBI	T_{Qmean}	BF wd	BF ht	TB wd	TB ht	Hab	BFF	Mean FPL	Max FPL	Skew	Kurt	s-Mean	s-IQR
AB021	26	0.35	0.37	0.28	51	4.3	63	10.5	83	0.11	21940	37966	-0.31	-0.94	0.58	0.39
AB044	20	0.34	0.29	0.3	51		56			0.14	19333	40499	0.19	-0.49	0.48	0.3
BB126	1	0.3	0.15	0.32	85		95			0.15	48732	103386	0.17	-0.91	0.47	0.37
BB146	7	0.16	0.76	0.21	20	1	22	3.2	76	0.2	5661	10273	-0.3	-0.93	0.55	0.41
BB218	5	0.37	0.29	0.34	14	1.8	22.5	4.8	96	0.19	5596	10183	-0.36	-0.68	0.55	0.34
BB238	6	0.32	0.5	0.24	45	5.7	44	8.5	44	0.13	18272	38690	0.43	-0.77	0.47	0.37
BB241	6	0.18	0.87	0.19	23	1.7	27.2	3.9	81	0.3	4932	8039	-0.35	-0.71	0.61	0.37
BB312	4	0.44	0.92	0.18	40	3.8	44.5	10.8	63	0.31	5708	11058	0.05	-0.78	0.52	0.34

Table B-1 Continued

Site	Slp	RR	RBI	T_{Qmean}	BF wd	BF ht	TB wd	TB ht	Hab	BFF	Mean FPL	Max FPL	Skew	Kurt	s-Mean	s-IQR
BB362	7	0.31	0.53	0.24	25	1.6	28.3	4.4	64	0.21	7883	15794	-0.03	-0.76	0.5	0.34
BB406	4	0.5	1.18	0.16	48	3.8	52	6.8	55	0.17	12358	23098	-0.06	-0.74	0.54	0.34
BB407	5	0.47	0.76	0.18	70	2.8	79.5	7.8	69	0.2	11054	21934	0.08	-0.98	0.5	0.37
BB427	5	0.41	0.82	0.2	33	3.2	58	5.5	47	0.27	6619	11875	-0.13	-1.04	0.56	0.41
CB011	32	0.46	0.29	0.32	91		98			0.24	21921	36842	-0.5	-0.9	0.59	0.4
CB032	27	0.57	0.41	0.28	130	2.2	130	2.2	91	0.04	35802	65596	-0.31	-1.17	0.55	0.53
CB133	4	0.17	0.85	0.18	19	2	28	7	77	0.3	3075	5521	-0.37	-0.78	0.56	0.38
CB139	5	0.35	0.84	0.19	32	1.6	38.5	6.5	57	0.3	8283	15221	-0.29	-0.76	0.54	0.34
CB146	4	0.59	0.84	0.17	48	2	62	10	45	0.16	17049	27990	-0.47	-0.69	0.61	0.37
CB192	28	0.43	0.41	0.27	41	3	52	7.7	92	0.16	9290	20339	0.23	-0.87	0.46	0.38
CB224	7	0.24	0.56	0.28	34.5	1.8	45	6.8	73	0.19	11945	20921	-0.33	-0.82	0.57	0.39
CB269	31	0.46	0.31	0.31						0.15	34020	58947	-0.48	-0.35	0.58	0.3
CB319	4	0.57	1.14	0.16	23	1	36.5	4.5	24	0.2	6544	12236	-0.2	-1.16	0.53	0.46
EB090	24	0.48	0.17	0.35	290		300			0.11	64688	148503	0.61	-0.32	0.44	0.3
EB145	30	0.3	0.3	0.33	71	4	104	10	58	0.15	26581	47801	-0.32	-1.08	0.56	0.46
EB167	34	0.53	0.21	0.36	66	3.2	74	10.2	88	0.08	20931	45831	0.57	-0.81	0.46	0.36
EB194	29	0.54	0.18	0.35	270	5	290	9.5	81	0.11	77499	180688	0.46	-0.55	0.43	0.34

Table B-1 Continued

Site	Slp	RR	RBI	T_{Qmean}	BF wd	BF ht	TB wd	TB ht	Hab	BFF	Mean FPL	Max FPL	Skew	Kurt	s-Mean	s-IQR
EB250	37	0.48	0.34	0.38	170	5.2		11.7	83	0.11	55323	111095	0.02	-0.84	0.5	0.33
EB273	42	0.64	0.37	0.32	73	3.5	82	6.8	88	0.25	8885	17088	0	-0.65	0.52	0.31
EB294	38	0.66	0.37	0.31	95	1.9	96	3	89	0.19	11914	24367	0.13	-1.05	0.49	0.41
EB320	41	0.52	0.22	0.35	62	3.7	78	5	95	0.34	12304	19383	-0.67	-0.38	0.63	0.33
FB010	37	0.5	0.26	0.36	60	4.8	64	6.5	66	0.17	21186	39152	-0.21	-0.8	0.54	0.35
GB011	44	0.58	0.24	0.36	130	7.5	140	10.5	55	0.2	25731	48437	-0.15	-0.53	0.53	0.3
GB042	35	0.7	0.2	0.38	75	2.8	75	2.8	90	0.17	13745	28051	0.26	-1.02	0.49	0.41
IB039	2	0.24	0.06	0.38	105		105			0.1	80257	178177	0.42	-0.76	0.45	0.37
JB007	6	0.21	0.66	0.22	95	1.5	111	6.6	64	0.1	32658	60717	-0.24	-1.08	0.54	0.44
JB018	5	0.24	0.75	0.22	54	2.8	54	2.8	82	0.16	18895	35797	-0.09	-1.06	0.53	0.4
JB037	6	0.37	0.49	0.24	66	3.2	77	11.7	40	0.21	22504	39074	-0.27	-0.98	0.58	0.4
JB052	6	0.36	1.04	0.18	46	1.5	50	5.5	53	0.3	6661	12750	-0.2	-0.61	0.52	0.32
JB055	5	0.18	0.43	0.25	54	1.5	57	5.7	57	0.1	27867	53375	-0.06	-1.05	0.52	0.42
JB063	6	0.35	0.81	0.2	32	2.3	39	6.8	67	0.12	11851	25051	0.2	-0.7	0.47	0.33
JB106	2	0.25	0.4	0.29	40	2.8	48.6	6	88	0.12	17914	41626	0.46	-0.62	0.43	0.35
JB156	6	0.38	0.48	0.24	62	3	66	5.2	49	0.33	13963	24623	-0.37	-0.72	0.57	0.35

Table B-1 Continued

Site	Slp	RR	RBI	T_{Qmean}	BF wd	BF ht	TB wd	TB ht	Hab	BFF	Mean FPL	Max FPL	Skew	Kurt	s-Mean	s-IQR
JB165	5	0.57	0.78	0.17	50	1.5	56	7	66	0.14	9999	19488	-0.08	-0.21	0.51	0.25
JB291	5				32	3	41	8	61							
KB003	24	0.49	0.22	0.35	135		142		73	0.04	68237	121583	-0.15	-1.17	0.56	0.48
LB012	29	0.47	0.37	0.3	73	4.4	77	5.2	67	0.22	15782	32675	0.03	-0.58	0.48	0.3
NB008	21	0.39	0.25	0.35	69	2.6	87	5.6	92	0.04	36147	85317	0.53	-0.66	0.42	0.37
NB028	16	0.34	0.31	0.3	180	2.1	195	3.4	86	0.15	33350	67301	0.15	-0.74	0.5	0.35
NB076	1	0.26	0.34	0.25	24	2	24	2	62	0.13	25437	48418	-0.06	-1.07	0.53	0.41
OB056	5	0.26	0.32	0.28	55	1.7	62	5	83	0.06	45039	83097	-0.07	-1.12	0.54	0.43
OB058	5	0.25	0.28	0.28	142	5.5	144	8.5	83	0.07	98765	191028	-0.09	-1.18	0.52	0.48
OB090	3	0.25	0.16	0.31	110	6.5	125	15	41	0.09	123388	254084	0.1	-0.77	0.49	0.33
OB101	5	0.22	0.28	0.28	77	3.2	129	10.2	89	0.1	64938	116410	-0.4	-0.61	0.56	0.31
OB107	1	0.3	0.53	0.22	26	2.8	26	2.8	93							
PB004	1	0.35	0.35	0.26	44	6.2	54	7.2	68	0.23	13902	30720	0.06	-0.35	0.45	0.28
QB042	25	0.36	0.3	0.32	104	5	114	10	57	0.23	21390	37473	-0.49	-0.7	0.57	0.36
QB055	26	0.39	0.16	0.33	75		90			0.19	37695	81934	0.23	-0.44	0.46	0.27
QB088	20	0.41	0.24	0.34	60	3.7	60	4.8	83	0.13	22878	39205	-0.35	-0.71	0.58	0.37
QB112	14	0.38	0.19	0.34	85		92			0.15	58755	121939	-0.03	-0.44	0.48	0.28

Table B-1 Continued

Site	Slp	RR	RBI	T_{Qmean}	BF wd	BF ht	TB wd	TB ht	Hab	BFF	Mean FPL	Max FPL	Skew	Kurt	s-Mean	s-IQR
QB118	16	0.38	0.29	0.34	109	3.6	123	9.6	71	0.19	33129	55299	-0.69	-0.39	0.6	0.32
QB206	13	0.31	0.36	0.30	82	4.2	112	12.5	42	0.13	29535	55152	-0.23	-0.78	0.54	0.35
QB223	11	0.28	0.31	0.29						0.08	52203	97626	-0.14	-0.92	0.53	0.4
QB233	7	0.32	0.59	0.23	104	5.1	110	8.9	54	0.16	26228	52880	0	-0.5	0.5	0.3
QB351	4	0.2	1.01	0.17	23.5	4.5	26	7.5	59	0.45	4280	7045	-0.59	-0.69	0.61	0.37
QB352	5	0.25	1	0.15	30.5	1.7	30.5	1.7	71	0.23	7257	11148	-0.74	-0.39	0.65	0.37
QB355	3	0.25	1.07	0.15	32	3.2	37	5.2	60	0.13	11675	21905	-0.02	-1.36	0.53	0.56
QB403	7	0.25	0.73	0.21	68	3.1	93	8.5	79	0.09	26531	53955	-0.03	-0.88	0.49	0.36
QB448	36	0.41	0.26	0.32	42	3	50	5	73	0.1	14252	27139	0	-1.24	0.53	0.45

Appendix C: Validation of the NC SAM rapid stream assessment method

Methods

The methods used for site selection, field assessments, and watershed condition are detailed in the main text of this document in CHAPTER 2: Methods.

A total of 98 NC SAM assessments were completed at 70 locations between May 2012 and September 2013. Of these, 15 were during the summer 2012 (S12) sampling season (May-September, 2012; 15 sites), 13 were during the winter 2013 (W13) sampling season (February-March, 2013; 13 sites), and 71 were completed during the summer 2013 (S13) season (May-September 2013; 67 sites). The list of sites sampled during each sampling season is provided in Appendix A.

Method precision analysis: duplicate assessments

Multiple assessments (two for most sites, three for one site) were completed at individual sites across sampling seasons and, in the case of 2 sites, within the summer 2013 season. The NC SAM overall functional ratings for each pair of repeated assessments were compared, both by the relative rating (High, Medium, or Low) as well as by using a rank value developed for this project. The rank value was determined by the combination of the habitat, hydrology, and water quality ratings; these three subfunction ratings determine the overall rating based on majority-rules logic, with no consideration of order. For example, if two of the lower-level function rate high and the third is low or medium, the overall function also rates high. The ten possible combinations were determined and ordered, then assigned a rank value from 1-10 (Table C-1) representing the range of highest to lowest function. Using the ten rank categories allowed basic statistical analysis, such as regression, to be performed. The small number of categories in the native NC SAM ratings combined with small sample sizes for some ratings precluded more rigorous statistical analysis and in cases where this was necessary (i.e., comparison of the subfunction ratings to existing monitoring data), analyses were primarily limited to descriptive statistics.

Table C-1 NC SAM Ranks as determined by subfunction ratings

NC SAM Overall rating	Subfunction 1	Subfunction 2	Subfunction 3	NCSAM rank
H	H	H	H	1
	H	H	M	2
	H	H	L	3
M	H	M	M	4
	M	M	M	5
	L	M	H	6
	L	M	M	7
L	L	L	H	8
	L	L	M	9
	L	L	L	10

For comparison to other stream assessment measures (bioclassification, habitat, water chemistry, hydrograph characteristics), only the summer 2013 data were used. All four NC SAM ratings (major subfunctions and overall) for each site were compared to the stream's

NC use support status, defined as the regulatory determination of waterbody impairment. Overall ranks were compared to the latest NCDWR bioclassifications from each site. Habitat functional ratings were compared to the field habitat assessment scores. Hydrology functional ratings were compared to two measures of stream flashiness (the Richards-Baker Index and TQmean) and the runoff ratio.

Results are presented as tabular summaries, descriptive statistics, and as box plots. Where appropriate, non-parametric tests (e.g., Wilcoxon rank sum) were used to test for significant differences in response variables for the NC SAM functional ratings.

Results and discussion

NC SAM assessment results are presented in Figure C-1 (S12), Figure C-2 (W13), and Figure C-3 (S13). Deviations from the original sampling plan occurred during the S13 season. Two sites were not sampled due to active utility construction during the summer, bringing the total number of assessed sites to 67. Assessments were repeated at four sites during the season. For two, the abbreviated NC SAM Large/Dangerous River (LDR) method had been used during the first site visits due to non-wadeable conditions, while the second assessments were performed at lower stream stage using the full assessment method. The remaining two sites were re-visited and re-sampled during the summer due to low confidence in the original results due to marginally wadeable conditions. In both of these cases, only the second assessments for these sites were used for analysis.

NC SAM ratings, preliminary sites (2012)

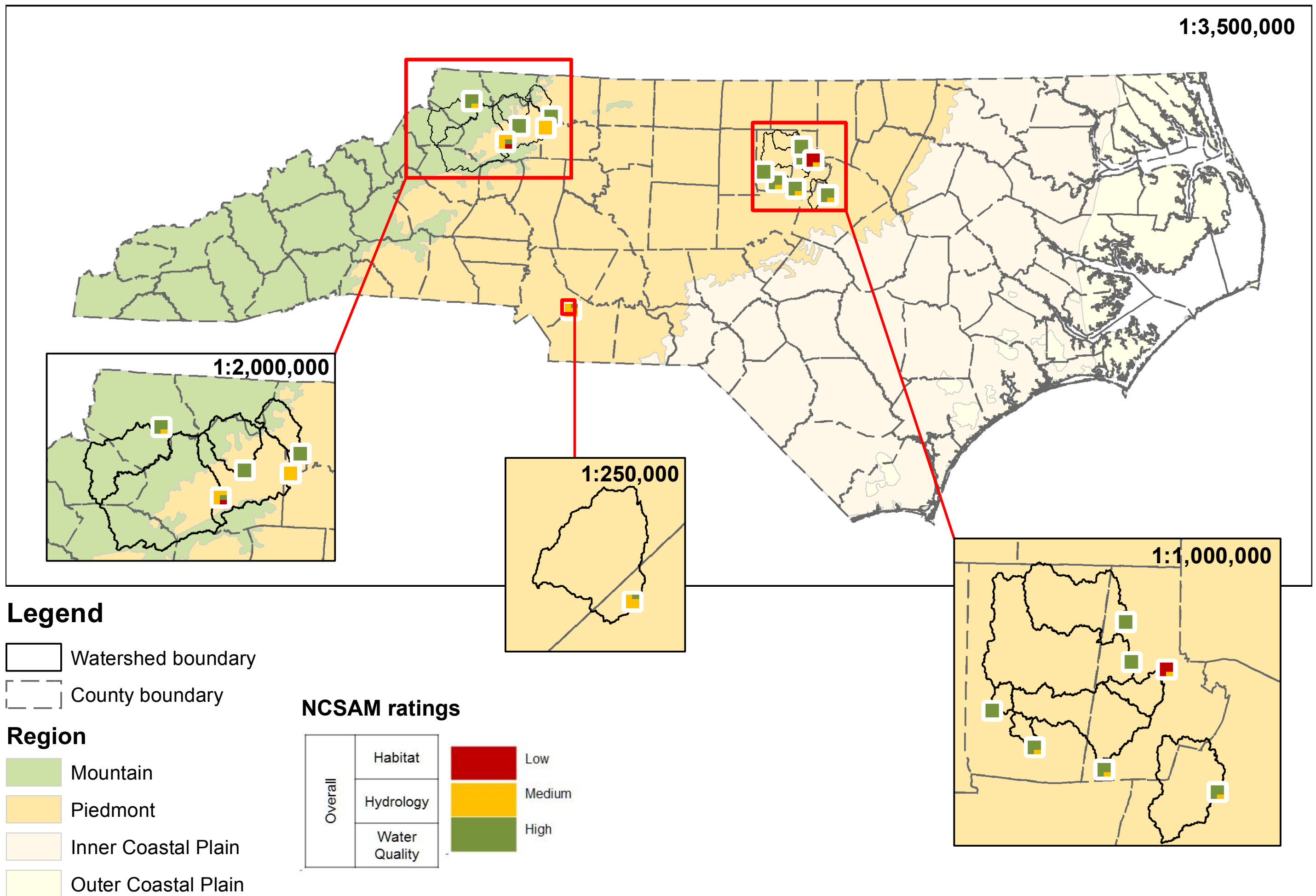


Figure C-1 NC SAM results for preliminary sites, sampled May-September 2012

NC SAM ratings, winter sites (2013)

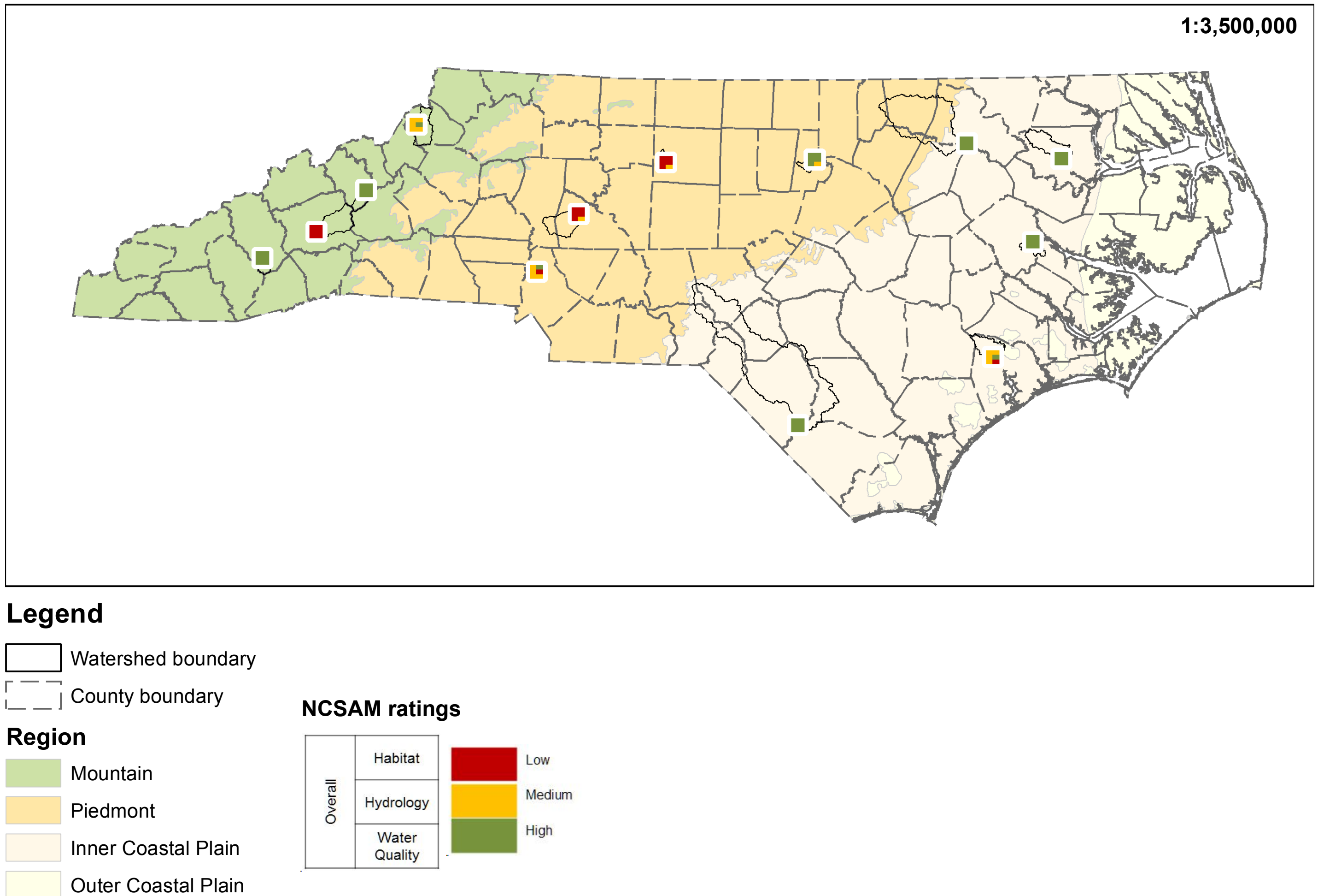


Figure C-2 NC SAM results for winter sites, sampled February-March, 2013

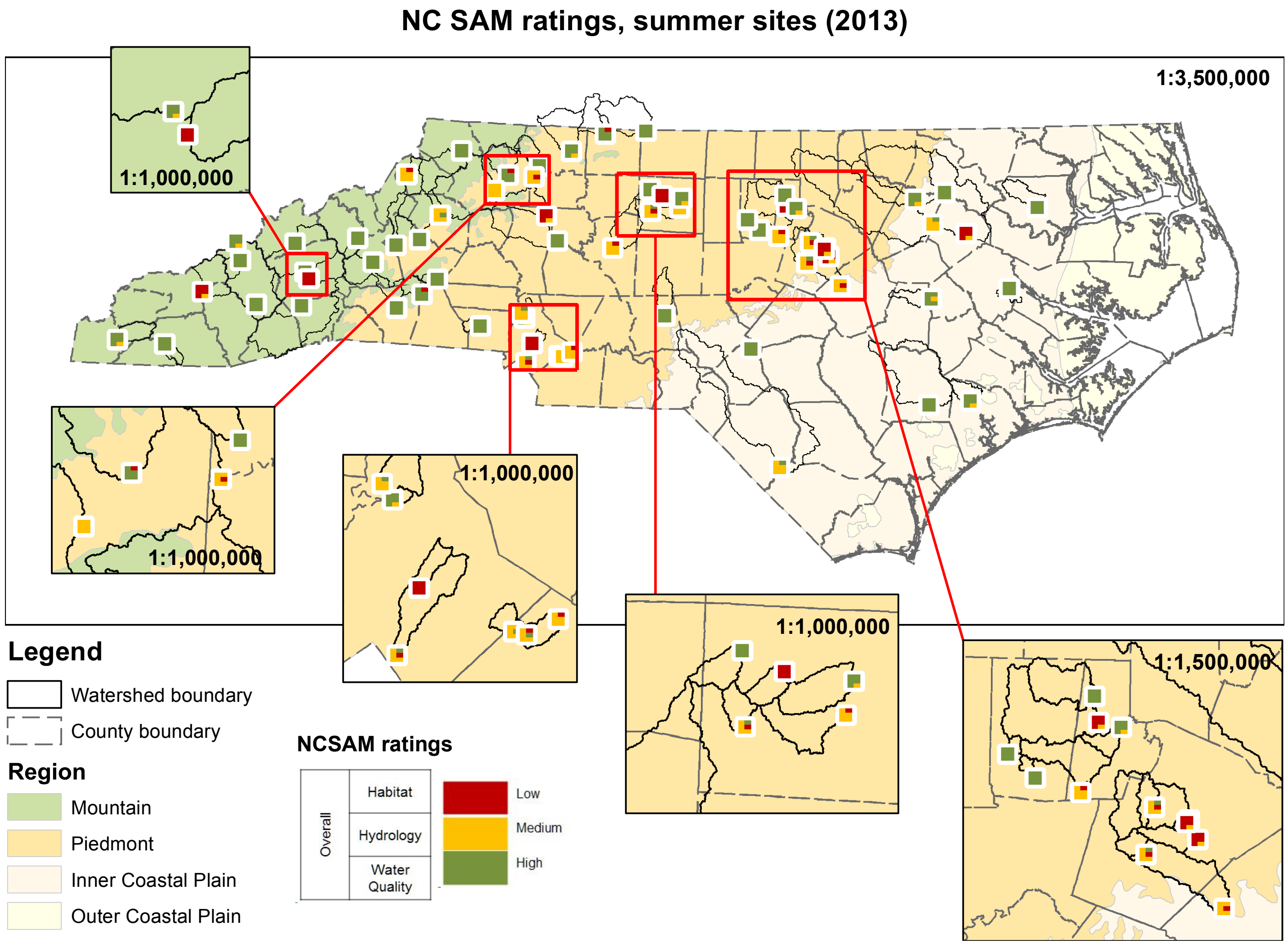


Figure C-3 NC SAM results for summer 2013 sites, sampled May-September 2013

When distributions of all assessments were examined statewide and by region (Figure C-4) and by sampling season (Figure C-5), NC SAM ratings appeared skewed towards high for the overall function and two of the three major subfunctions (habitat and hydrology) and to a lesser extent for the water quality function.

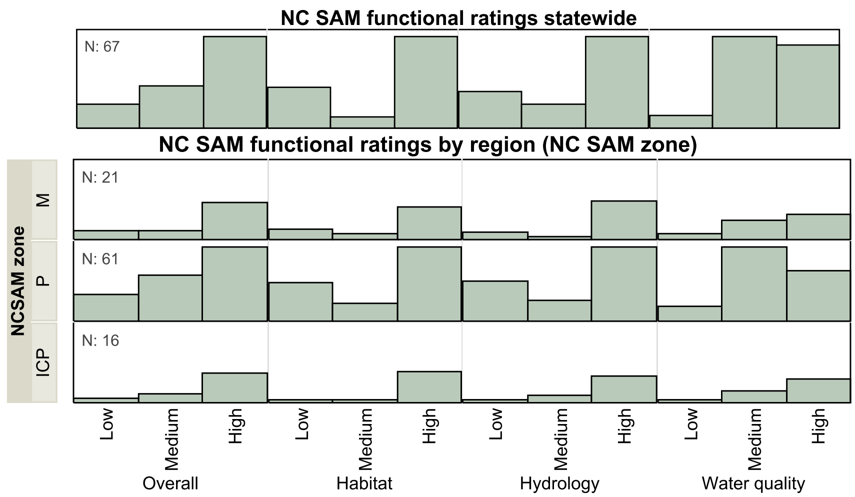


Figure C-4 NC SAM functional ratings statewide and by NC region

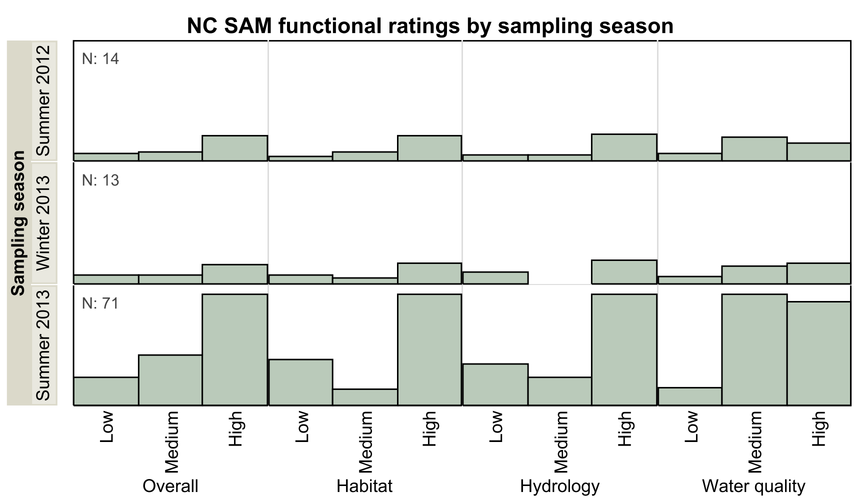


Figure C-5 NC SAM functional ratings by sampling season.

NC SAM functional ratings from S13 were compared to current use support status (impaired or supporting) (Figure C-6). Since regulatory impairments are determined based on multiple criteria, not just numerical standard exceedences or biological communities (NC DENR Division of Water Quality 2012b), it would be anticipated that one or more NC SAM measures should reflect the current use support status of a given stream. Supporting streams (Impaired = False) had predominantly high ratings for all NC SAM functions. I anticipated that impaired streams would receive predominantly low ratings, particularly for water quality function, but the percentage of impaired sites receiving a low rating were 29% (overall), 46% (habitat and hydrology), 14% (water quality).

While the NC SAM method specifically requests information on use support status in the header/informational section, this information is not considered by the underlying model used to calculate the functional ratings. However, the model does consider the answer to a question on water quality stressors (questions 7 on the NC SAM form). For all impaired streams, answer E was selected ("Current published or collected data indicating degraded water quality in the assessment reach") but results suggest that this information is not very heavily weighted in the model.

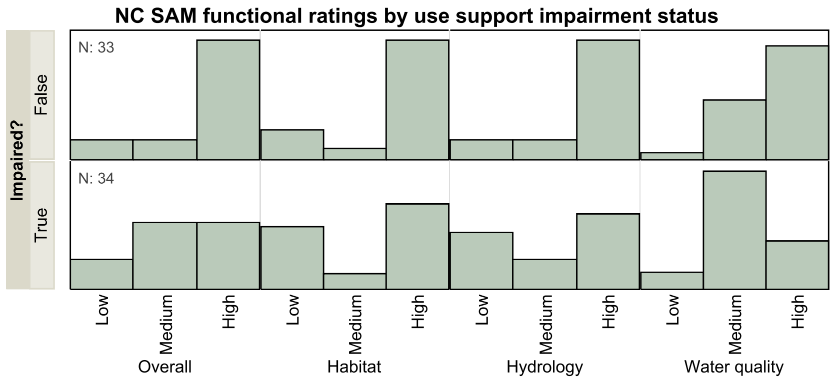


Figure C-6 Distributions of NC SAM functional ratings by use support status (True = impaired; False = supporting) for S13 samples.

NC SAM overall rank values were compared to bioclassifications, as both purport to reflect the combined effects of physical, chemical, and biological conditions within NC streams. Three sites were excluded from this analysis. The benthos sampling location was determined to be located at a non-equivalent point in the stream for New Hope Cr. (BB238); specific conductance readings during sampling visits were found to be outside of the historic range at Fishing Cr. near Enfield (OB101); and the age of the benthos data, combined with recent development in the watershed, was of concern for a Crabtree Cr. site (JB156). The total number of samples analyzed was 64.

NC SAM ranks appeared visually skewed towards high function (represented by lower rank values) and bioclassifications tended to have a fairly uniform distribution, whether examined by region or statewide (Figure C-7, Figure C-8, Figure C-9, Figure C-10). Bioclassifications in the Mountain region were also skewed towards higher values (Good, Good-Fair). However, Piedmont and Inner CP sites seemed to show more stressed communities (Good-Fair, Fair). Piedmont bioclassifications showed a marked peak for the Fair category but that hallmark was not evident in the NC SAM data.

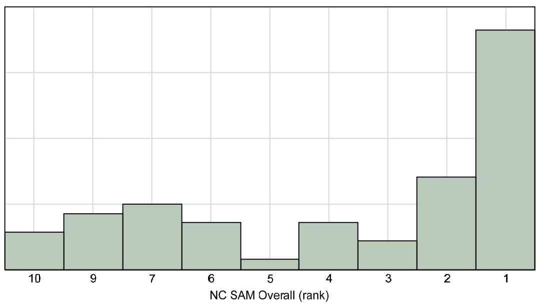
<table>
<tr><th></th><th>NC SAM Overall ranks</th><th>Bioclassifications</th></tr>
<tr><td>Statewide</td><td>

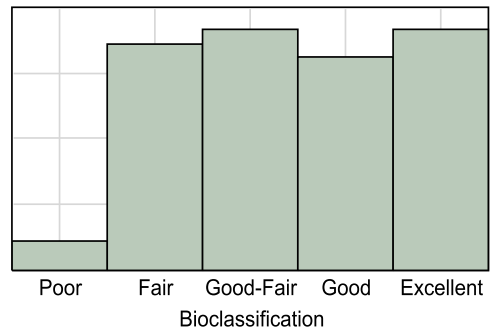
Figure C-7 Distribution of NC SAM overall ranks for summer assessments (total n=64). Lower rank values correspond to higher stream function.</td><td>

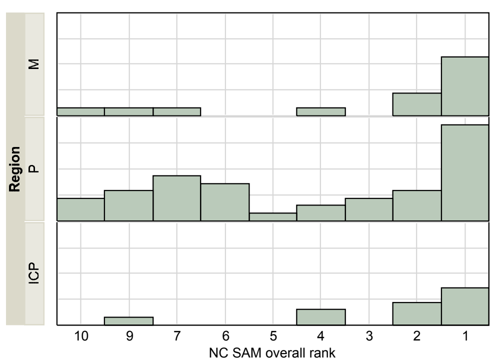
Figure C-8 Distribution of last bioclassifications for summer assessment sites (n=64).</td></tr>
<tr><td>By region</td><td>

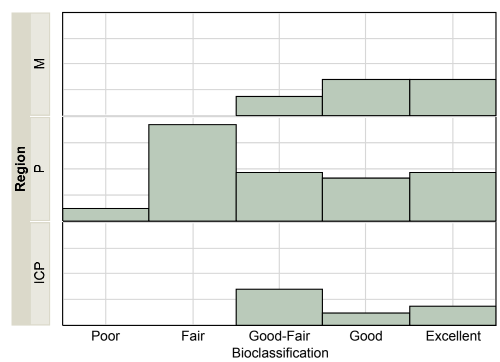
Figure C-9 Distribution of NC SAM overall ratings for summer samples by region (NC SAM zone).</td><td>

Figure C-10 Distribution of bioclassifications by region (NC SAM zone).</td></tr>
</table>

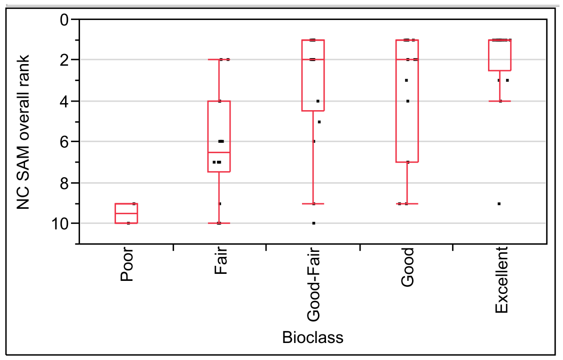


Figure C-11 NC SAM rank by bioclassification, statewide. Lower rank values correspond to higher stream function.

NC SAM ranks and bioclassifications (Figure C-11) showed a similar gradient, with Poor bioclassifications associated with higher ranks (indicating more degraded instream conditions. Wilcoxon rank-sum tests indicated that the distributions of ranks were significantly different ($p<0.05$) for most pairs (7 of 10 comparisons), with the exception of Fair and Poor; Good-Fair and Excellent; and Good-Fair and Good. Median NC SAM rank values by bioclassification were 9.5 (Poor), 6.5 (Fair), 2 (Good-Fair and Good), and 1 (Excellent). This suggests that in many cases differences in bioclassification can be detected using the NC SAM rank.

When compared by region (Figure C-12), differences in NC SAM ranks and bioclassification did not occur in the Mountain and Inner CP, though these regions had fairly small sample sizes (15 and 10, respectively, with only three bioclassification categories). There was a possible outlier identified in the data set, a Mountain site that received a rank of 9 yet had a bioclassification of Excellent. In this case, NC SAM may have provided a more accurate assessment. This site (Oconaluftee R, GB011) was downstream of the town of

Cherokee, which has been undergoing development since the last benthos sample was performed in 2009. Field notes noted poor quality habitat, including numerous bank slumps and excessive fine sediments. The river was closely flanked by roads on either side, one of which was a major highway (US-19). Excluding this site from analysis improved results, with Good and Excellent bioclassifications being significantly different (p=0.0439).

The Piedmont had a more robust sample size, and all five bioclassification categories were present, but only three bioclassification categories were found to be significantly different in the Wilcoxon rank-sum tests: Excellent—Fair (p = 0.0004), Excellent—Poor (p = 0.0208), and Good-Fair—Fair (p = 0.0268). Good-Fair—Poor (p = 0.0585) and Good—Fair (p = 0.0592) were marginally significant.

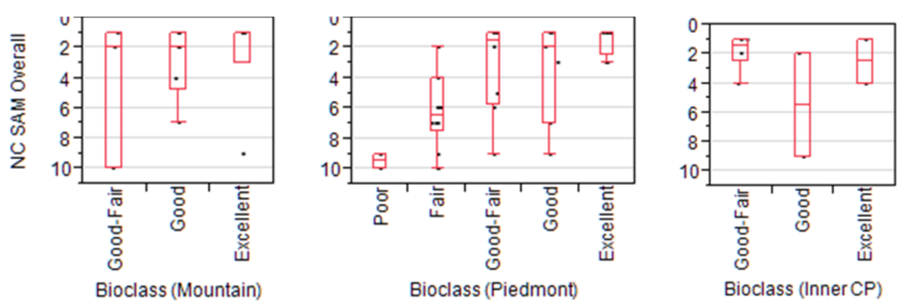


Figure C-12 NC SAM rank by bioclassification for NC SAM zones (Mountain, Piedmont, and Inner CP).

Field habitat assessments and NC SAM habitat functional ratings from S13 site visits were compared. Non-wadeable conditions precluded field habitat assessments at a number of sites. Analysis was based on the assessments from 55 sites.

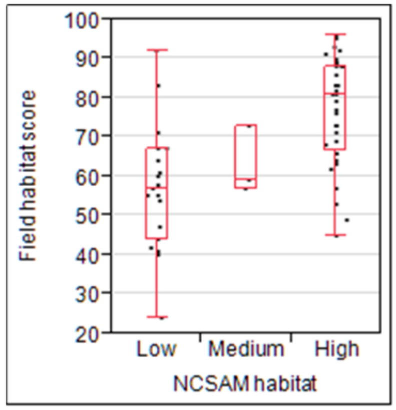


Figure C-13 Distributions (box and whisker plots) of field habitat assessment scores and NC SAM habitat function ratings.

As with bioclassifications, NC SAM and field assessments seemed to follow a similar trend when comparing distributions (Figure C-13). Median habitat scores by NC SAM rating were 57 (low), 59 (medium), and 88 (high). Wilcoxon rank-sum comparisons indicated that distributions of habitat field assessment scores for low and high rated sites were significantly different ($p<0.0001$), suggesting that NC SAM can discern between sites with the best and worst habitat conditions.

Several non-conforming sites were identified (i.e., sites with high field habitat scores but low NC SAM ratings, or sites with low field habitat scores and high NC SAM ratings). Three sites had habitat assessment scores in the top quartile, yet rated low on NC SAM. Nine sites with habitat scores in the bottom quartile received high NC SAM ratings. The majority of these non-conforming sites were located in the Piedmont, with only one each in the Mountains or ICP. One shared characteristic of these sites was that they have all been impaired for one or more designated uses, with the exception of the site in the Inner CP.

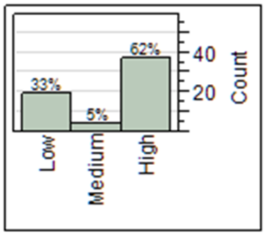
<table>
<tr><th></th><th>NC SAM habitat rating</th><th>Field habitat assessment score</th></tr>
<tr><td>Statewide</td><td>


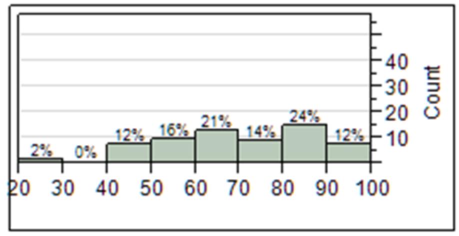
Figure C-14 Distribution of NC SAM habitat ratings for summer assessments statewide (total N = 58).</td><td>


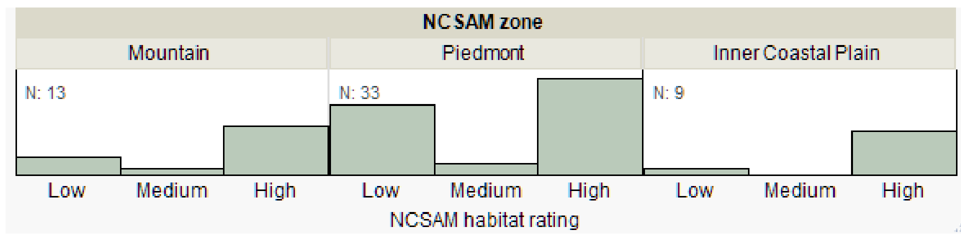
Figure C-15 Distribution of field habitat assessment scores statewide (total N = 58).</td></tr>
<tr><td rowspan="2">By region</td><td colspan="2">


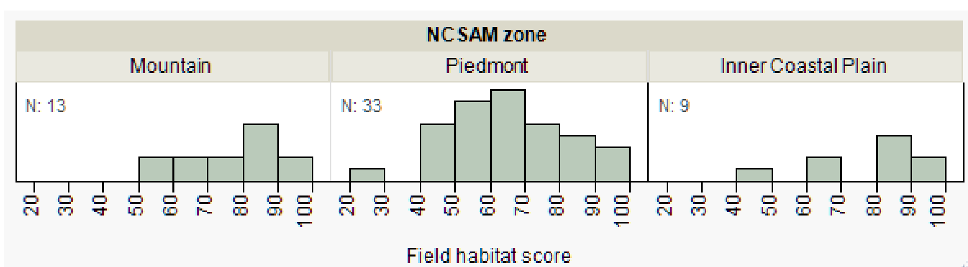
Figure C-16 Distributions of NC SAM habitat ratings by region</td></tr>
<tr><td colspan="2">


Figure C-17 Distributions of field habitat assessment scores by region.</td></tr>
</table>

As compared to the conforming sites, the nine sites with low habitat scores and high NC SAM ratings tended to have smaller drainage areas (median = 26.7 km^2; conforming sites = 87 km^2; p-value = 0.0467), more developed area in the watershed (median = 59%; conforming sites = 7%; p-value = 0.0069), less forest area (median = 22%; conforming sites = 60%; p-value = 0.0134), and more impervious area (median = 15%; conforming sites = 0.01; p-value = 0.0062). These sites also had higher SC (median = 312 μS/cm at 25°C); conforming sites = 84; p-value = 0.0463) and higher flashiness as measured by the RBI metric (median = 0.78; conforming sites = 0.35; p = 0.0251) and the T_{Qmean} (median = 0.19; conforming sites = 0.30; p = 0.0187). That these conditions were significantly different than those seen at conforming sites suggests that NC SAM may not appropriately assess particularly stressed sites with high watershed impervious area and low forest cover.

The NC SAM hydrology rating was compared to the watershed runoff ratio, RBI flashiness index, and T_{Qmean} metric by region. These metrics vary naturally with terrain characteristics, such as slope, and using the regional groupings was intended to control for terrain differences. Higher values for the runoff ratio indicated that a larger proportion of precipitation was routed to the stream and watershed outlet rather than stored within the watershed or lost through other fluxes (e.g., ET). Higher RBI and lower T_{Qmean} flashiness values are associated with flashier discharge, i.e., sharper increases in the streamflow response to storm events followed by a quicker return to baseflow conditions.

Distributions of the runoff ratio, RBI, and T_{Qmean} by NC SAM rating and region (Figure C-18) suggested that NC SAM ratings did not show any significant correlations to any of these measures, nor were there any significant differences between distributions when grouped by NC SAM rating when tested using Wilcoxon rank-sum.

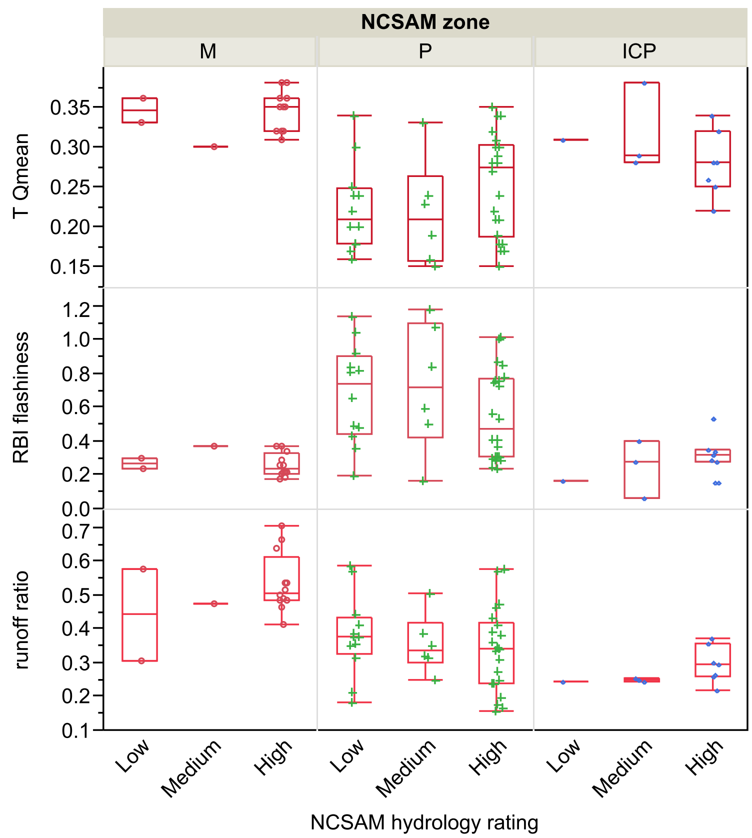


Figure C-18 Distributions of two flashiness metrics (T_{Qmean} and RBI) and the watershed runoff ratio by NC SAM hydrology rating and region.

Though certain groupings had very small sample sizes (e.g., low-rated Mountain and Inner CP streams), it appears that the NC SAM hydrology rating, as compared to these particular measures, may actually respond differently across ecoregions. Due to small sample sizes in the Mountain and ICP regions, Spearman correlation analysis was only performed on Piedmont sites, and no significant correlation was found. No differences in the distributions of the runoff ratio or flashiness metrics were found sites were grouped by NC SAM rating. Because no identifiable trends or relationships were identified, further analyses of residuals and/or anomalous observations were not performed.

While the NC SAM hydrology rating did not appear to reflect the quantitative watershed hydrology indicators used here, this may be due to an inappropriate selection of indicators for this study. The NC SAM manual does not provide a full description of what is specifically considered to be the “hydrologic functions” of a stream; additional guidance in the manual would have been helpful for selection of a more appropriate response variable.

The water quality monitoring parameters used for comparison to NC SAM water quality functional rating included physical measures (SC, temperature, and turbidity), microbiological indicators (FC), and nutrients (TN, and TP). Both SC and FC were $\log_{10}$-transformed based on the wide range of values seen for these measurements. As with the habitat and hydrology analyses, there was a paucity of low-rated sites: there were none in the ICP, one in the Mountains, and four in the Piedmont.

The strongest correlations were seen in the Piedmont, with lowest NC SAM scores associated with the highest SC, temperature, FC, and nutrients. In the Mountains, the only significant correlation was with temperature. Many of the original nutrient results were at or below the detection limit, and this was particularly common in the Mountain data. This minimized the ability to identify trends or patterns in the data. No significant correlations were seen in the ICP.

Table C-2 Results of Spearman correlations between NC SAM water quality function and physical, microbiological, and chemical parameters.

Parameter	M		P		ICP	
	ρ	p	ρ	p	ρ	p
$\log_{10}$(SC)	0.510	0.0623	**0.667**	<.0001	-0.260	0.4996
Temperature	**0.703**	0.0108	**0.632**	<.0001	0.130	0.7380
Turbidity	0.442	0.1134	0.314	0.0550	0.087	0.8239
$\log_{10}$(FC)	0.573	0.0514	**0.336**	0.0421	-0.087	0.8247
TN	0.529	0.1161	**0.566**	0.0004	0.144	0.7575
TP	0.296	0.4067	**0.371**	0.0282	-0.364	0.4220

Table C-3 Summary of duplicate assessments across sampling seasons.

Seasons compared	Total	Method	
		Same	Mixed
S12 - W13	1	1	0
S12 - S13	14	12	2
W13 - S13	12	11	1
S13 - S13	2	0	2
All seasons	29	24	5

Several non-conforming data points were noted in the scatterplots for nutrients (not shown). For both TN and TP, the highest nutrient concentrations were associated with sites rated medium by NC SAM. These sites were above the 75th percentile for both parameters (>3.0 mg/L for TN and >0.21 mg/L for TP), and all were located in urbanized areas, though the range of development (41-96%) and TIA (9-33%) were seen in other project watersheds. Sites included Little Sugar Creek (CB146, located in downtown Charlotte); South Buffalo and North Buffalo Creeks (BB406 and BB407; both drain the city of Greensboro); New Hope Creek (BB238, located downstream of the city of Durham), and Ellerbe Creek (JB165, drains the city of Durham and at the upper end of Jordan Lake). All sites, except S. Buffalo Cr., were downstream of major (>1MGD) municipal WWTP discharges though none were closer than about 2km. All of these sites have been impaired due to violations of water quality standards (including Cu and Zn) and stressed biological communities. Additionally, N. Buffalo is listed as impaired for violating the nitrate nitrogen standard for water supplies. While the presence of the WWTP effluent discharge was noted on the NC SAM assessments, it appears that this information may be underweighted by the NC SAM model, resulting in an over-rating of water quality for these sites.

Analysis of method repeatability relied on 29 duplicate assessments that were completed (Table C-3) across all sampling seasons. Matched pairs analysis by site was performed based on: the entire duplicate sample data set; sites where the same NC SAM assessment method was used for both sampling visits (full assessment or Large/Dangerous River [LDR] method); sites where both NC SAM assessment methods were used (full and LDR, “Mixed” methods); and by season. Seasonal comparisons were completed only for S12-S13 and W13-S13 due to the small sample size for the S12-W13 and S13-S13 data sets.

When comparing agreement of the NC SAM ratings for overall and lower-level functions (habitat, hydrology, water quality) (Figure C-19), the rate of concurrence for duplicate assessments was moderate (67% for all functions/all methods). Rate of agreement increased when the same assessment method was used (73% for all functions) as compared to when mixed methods were used (65% for all functions). This pattern was also seen for the overall, hydrology, and water quality functions, but the relationship was reversed for habitat. Duplicate assessments for S12-S13 and W13-S13were also reviewed (Figure C-20). In this case, the S12-S13 showed the lowest levels of concurrence, ranging from 50% to 71%. The W13-S13 comparisons showed improvement, with a 75% overall agreement.

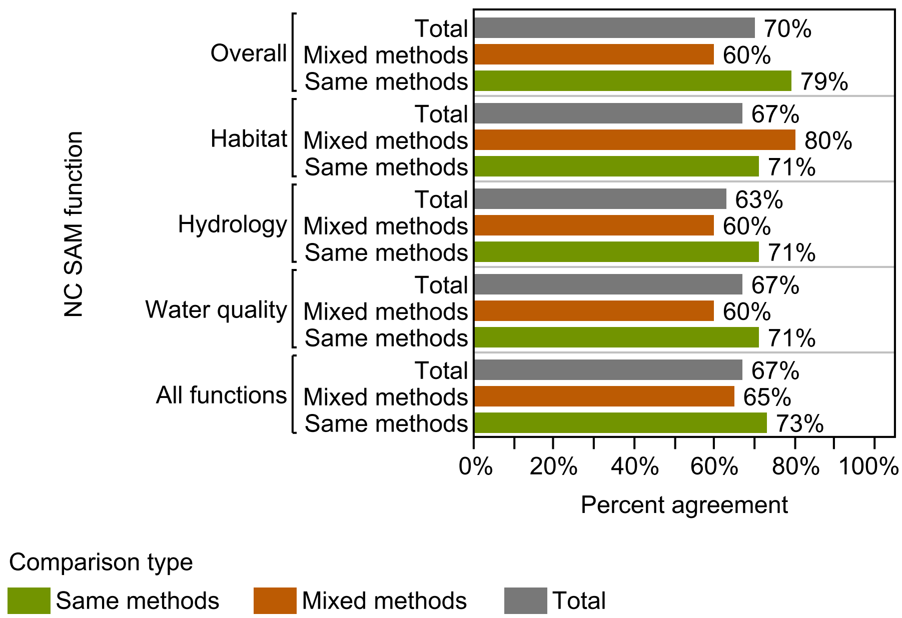


Figure C-19 Percent concurrence of repeated assessments by NC SAM functions for all duplicate assessments.

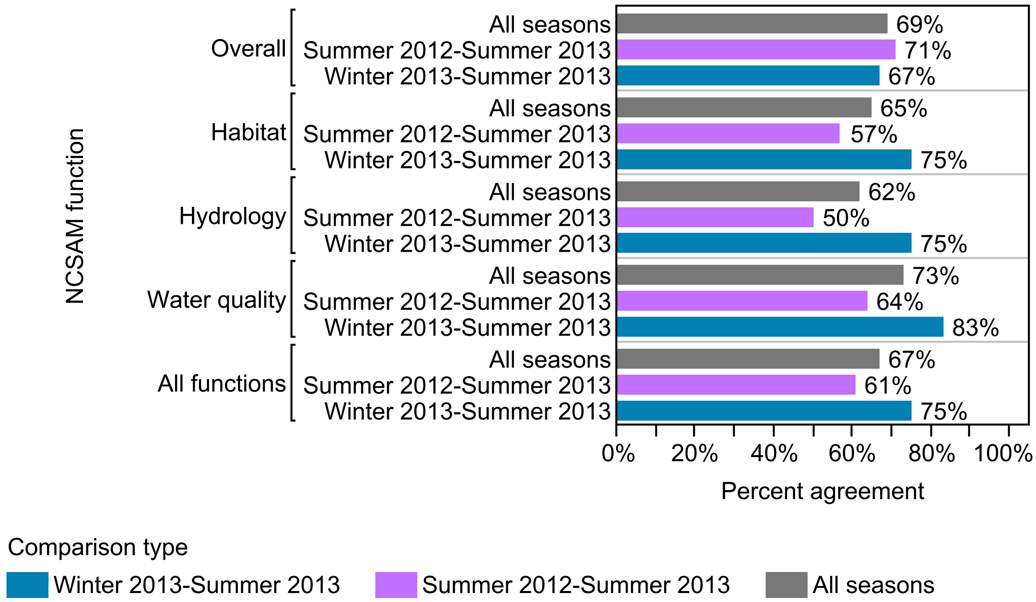


Figure C-20 Percent concurrence of repeated assessments by NC SAM functions by seasons.

Residuals of the replicates were generally smaller for the winter-summer comparisons (Figure C-21) than for the summer-summer comparisons (Figure C-22), i.e., repeated assessments showed less overall variability for W13-S13 than for S13-S13. Several factors may contribute to this: I participated in additional training with SFAT members between S12 and W13 sampling seasons; I gained more experience applying the NC SAM method; there may have been memory bias due to the shorter time period between W13 and S13; or high streamflows in S13 may have introduced bias, in that S13 streamflows were more similar to typical winter stream stages than those seen in the summer.

The influence of stage and discharge was examined using a matched pairs analysis (Wilcoxon signed rank) to determine if significant differences in daily mean discharge were present between sampling visits at each site. In spite of an unusually wet spring and summer, discharges on the days of sampling were not significantly different between S12 and S13 (S=21.000, p(S13>|S12|)=0.1509, p (S13>S12)=0.0754, p(S13<S12)=0.9246) but were different between W12 and S13 (S=-39.0, p(S13>|W13|)=0.0005; p(S13>W13)=0.9998, p(S13<W13)=0.0002). This suggests that differences in replicate assessments are likely due more to training, experience, and memory bias than subjective observer bias due to high flows.

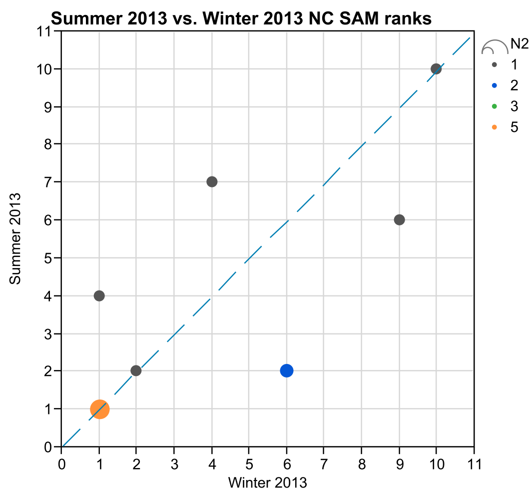


Figure C-21 Comparison of NC SAM ranks from replicate samples from winter 2013 (x-axis) and summer 2013 (y-axis). The size and color of the marker indicates the number of replicate pairs with the associated combination of ranks.

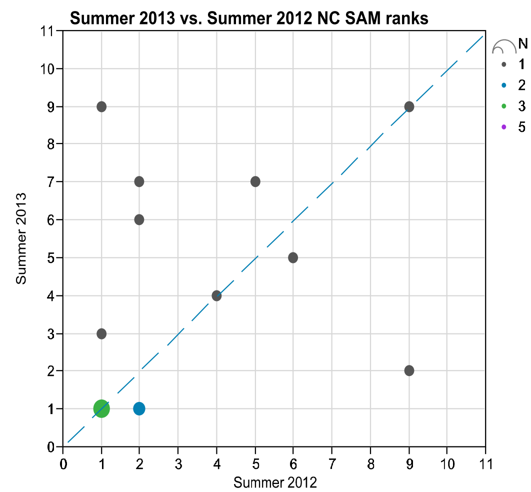


Figure C-22 Comparison of NC SAM ranks from replicate samples. Sites were sampled in the summer 2012 (x-axis) and summer 2013 (y-axis).

Results from the duplicate assessments stress the importance of training and experience with the method before it can be consistently applied. Even with experience there can be instances where there is great variability between sites (for example, two sites received rankings of 2 and 6 during the repeated assessments), though the variability appears minimized through experience. The variability was likely due primarily to the observational nature of the method. The inclusion of quantitative measurements would likely reduce the variation, though that would likely add time and cost to implementing the method, which would negate the initial intent of the NC SAM development team for a rapid and efficient assessment method.

The local stream condition measures discussed thus far (NC SAM, bioclassifications, field habitat assessments, water quality monitoring data, and hydrograph characteristics) were compared to watershed conditions, including land cover, drainage area, and width function summary statistics to identify large scale spatial factors that influence reach-level observations. Results of correlations are shown in Table C-4.

With the exception of T_{Qmean} and the field habitat assessment, higher values were associated with degraded instream conditions. Almost all of the instream indicators showed a significant correlation with development and total impervious area (TIA), with higher percentages in the watershed associated with degraded instream conditions. The exception of the runoff ratio was surprising, as urban development is often associated with increases in runoff and decreases in infiltration. Agricultural land use showed very few correlations, with the exception of runoff ratio and turbidity, and in these cases increasing agriculture was associated with decreasing turbidity and runoff ratio. This was unanticipated, as agricultural land use is a commonly cited cause for stream degradation (Allan, Erickson, Fay 1997; Lenat and Crawford 1994; Poff, Bledsoe, Cuhaciyan 2006). However, more recent research has suggested that the effect of agricultural land use is quite difficult to differentiate from urban land use, and any effect seen due to agriculture is more likely due to the absence of urban land use (Burcher, Valett, Benfield 2007; King et al. 2005).

Table C-4 Results of Spearman correlations between local instream and watershed conditions. The Spearman ρ values are shown and also represented by the green or orange bar within each cell (longer bars indicate higher ρ). Significant correlations (p<0.05) are underlined and bold.

Stream function	Variable	Development (%)	TIA (%)	Agriculture (%)	Forest (%)	Drainage area (km^2)	Watershed slope (%)
Overall	NC SAM overall rank	**0.605**	**0.624**	-0.163	**-0.435**	-0.013	**-0.234**
	Bioclassification	**0.765**	**0.782**	-0.055	**-0.751**	**-0.352**	**-0.533**
Habitat	NC SAM habitat	**0.287**	**0.287**	-0.118	-0.147	-0.020	-0.018
	Field habitat assessment	**-0.674**	**-0.675**	0.056	**0.556**	0.008	**0.292**
Hydrology	NC SAM hydrology	**0.555**	**0.580**	-0.165	**-0.471**	0.029	**-0.261**
	Runoff ratio	-0.065	-0.072	**-0.708**	**0.303**	-0.008	**0.545**
	RBI	**0.580**	**0.562**	-0.073	**-0.542**	**-0.719**	**-0.441**
	T_{Qmean}	**-0.596**	**-0.613**	-0.049	**0.664**	**0.607**	**0.639**
Water quality	NC SAM water quality	**0.696**	**0.724**	-0.179	**-0.457**	-0.030	**-0.267**
	$log_{10}(SC)$	**0.738**	**0.759**	0.089	**-0.864**	**-0.305**	**-0.768**
	Temperature	**0.565**	**0.592**	0.054	**-0.740**	0.030	**-0.649**
	Turbidity	**0.480**	**0.478**	**0.315**	**-0.497**	0.083	**-0.400**
	$log_{10}(FC)$	**0.442**	**0.489**	-0.042	**-0.504**	**-0.439**	**-0.376**
	TN	**0.543**	**0.583**	0.158	**-0.758**	-0.050	**-0.622**
	TP	**0.371**	**0.430**	0.187	**-0.632**	0.021	**-0.676**

Forest cover and watershed slope were significantly correlated with all local instream responses, with the exception of NC SAM habitat rating. Increases in both watershed conditions were associated with better instream conditions, with the exception of the runoff ratio. The similarities of responses suggest that they co-vary or are spatially correlated, but the increase in runoff ratio suggests that watershed slope may be the primary driver, as higher slopes tend to reduce infiltration. Conversely, forest cover has been associated with decreasing runoff ratio through interception, increasing surface roughness, and increases in evapotranspiration (Guswa 2012).

Drainage area was correlated with relatively few instream indicators. The most notable was stream flashiness (RBI and T_{Qmean}) and this effect has been well documented (Leopold 1964), so further analyses focused on flashiness should ensure that the effect of drainage area is controlled during selection of factors for comparison.

NC SAM ratings and other local instream indicators did respond similarly to the watershed conditions examined, with the exception of the habitat rating. This suggests that NC SAM at the statewide scale does have promise as a rapid assessment method for streams similar to those used in this study, with the caveat that there were site-specific differences noted. As such, NC SAM would likely be appropriately used as a screening tool, or as a complement to other data collections or watershed information.