ECOHYDROLOGICAL STUDY OF WATERSHEDS WITHIN THE MILITARY INSTALLATION IN FORT BENNING, GEORGIA

By

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A DISSERTATION PRESENTED TO THE GRADUATE SCHOOL
OF THE UNIVERSITY OF FLORIDA IN PARTIAL FULFILLMENT
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This work is dedicated to my parents, Tara and Narmada Bhat, and uncle Devendra Bhat.

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Chair: Kirk Hatfield
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Relationships among watershed physical characteristics and water quality parameters were explored for seven watersheds in Fort Benning, Georgia, using statistical analyses to identify chemical indicators of ecological changes. Correlations were identified among the indicators and watershed physical characteristics. Regression results suggested that pH, chloride, total phosphorus, total Kjeldahl nitrogen, total organic carbon, and total suspended solids are useful indicators of watershed physical characteristics that are susceptible to perturbations.

The magnitude, frequency, duration, timing, and rate of change of hydrologic conditions regulate ecological processes in aquatic ecosystems. Analysis of 26 non-redundant hydrologic indices showed undisturbed watershed produced higher magnitude and more frequent low- and high-flows as compared to the disturbed watershed. Eighteen storm-based indices, grouped into four flow components, were proposed to statistically characterize hydrologic variation among different watersheds. Results

showed that these storm-based indices might be used as surrogates to the indices derived from long-term data. Statistical analysis showed that the watershed physical characteristics such as military training land, road density, and the number of roads crossing streams predicted hydrologic indices such as storm-based baseflow index, bankfull discharge, response lag, and time of rise well.

Riparian vegetation has an important role in altering water quality in the forested watersheds. The leaching of organic or mineral products on the forest floor provides potential additional effects on water quality. In these areas, nutrients are released into the fresh water systems due to the leaching and decomposition of vegetation litter. The release of nutrients from plant litters prior to decomposition may be an important aspect of characterizing stream water quality. To explore nitrogen leaching in a riparian area during litterfall, Riparian Ecosystem Management Model developed by USDA-ARS has been applied. The simulated total Kjeldahl nitrogen masses in the study watershed were close to the observed values. The model effectively captured the trends of litter mass accumulation in the riparian area and subsequent high concentrations and masses during those periods.

CHAPTER 1
GENERAL INTRODUCTION

Ecohydrology is the interdisciplinary research field in which hydrology and ecology come together. It may be defined as the study of the functional relations between hydrology and biota at the watershed scale (Zalewski, 2000). Resource managers and land use planners seek ecohydrological knowledge enabling them to design effective land use plans and water management strategies.

Water, soil, and plant cover are fundamental components that determine the productivity of the land. The successional stages in evolution of ecosystems depend on climatic and hydrologic conditions and on nutrient availability. The unique composition of plants and animals determines the ability to retain water and nutrients within the system (Zalewski et al., 1997). The amount of water and its quality in the aquatic environment are guided mostly by climate. Water and nutrient cycling in terrestrial systems are tightly linked with each other and with human activity. The growth and activity of human populations have increased the input of nutrients to terrestrial and aquatic ecosystems (Schindler and Bayley, 1993; Vitousek et al., 1997). Various forms of environmental perturbations affect quality and quantity of inland waters by increasing runoff, erosion, sedimentation, and pollution.

Modifications of land use affect the water quality, residence time, surface runoff, soil moisture, evaporation, and ground water. For example, an increase in urbanization and non-forest land increased nitrogen concentration in the streams (Osborne and Wiley, 1988; Sponseller et al., 2001). Land-use practices can potentially affect and modify

channels along the flow paths in a landscape and may enhance or disrupt runoff (Nakamura et al., 2000). Forest cutting can increase the frequency and volume of debris slides due to dead and decayed tree roots that contribute to soil strength on marginally stable sites (Sidle et al., 1985).

Water quality is highly variable, from place to place and from time to time, even within a particular ecosystem. It is dependent on many factors, both natural and as a consequence of human activities. In both cases, the quality of water is further affected by the soils and vegetation over and through which it passes. Nutrients are released into the freshwater systems due to the decomposition of the vegetation litter and modify the water quality. Riparian vegetation has an important role in altering water quality as it functions as a source or a sink of various organic and mineral components (Cirmo and McDonnell, 1997).

Agricultural practices and urbanization are widely recognized human impacts on water resources. A specific type of human impact is caused at military installations. Military training within a watershed can affect the drainage patterns, vegetation, and soils. The impact is due to troop maneuvers and large tracked and wheeled vehicles that traverse thousands of hectares in a single training exercise (Quist et al., 2003). The impacts of such activities range from minor soil compaction and lodging of standing vegetation to severe compaction and complete loss of vegetation cover in areas where training is concentrated (Wilson, 1988; Milchunas et al., 1999). The resultant impacts are evident both in the stream hydrology and the stream ecosystems. Potential impacts on terrestrial and aquatic ecosystems include disruption in soil density and water content (Helvey and Kochenderfer, 1990), addition of sediment, nutrients, and contaminants in

aquatic ecosystems (Gjessing et al., 1984), and impairment of natural habitat development,and woody debris dynamics in forested floodplain streams (Piegay and Landon, 1997). Military roads can significantly alter hillslope hydrology by redistributing soil and rock materials on slopes and increasing the rate of debris slide initiation. Roads also directly change the hydrology by intercepting shallow groundwater flow paths, diverting the water along the roadway and routing it to stream at road crossings. Road crossings can intercept groundwater drainage networks and collect groundwater from upslope areas, diverting it into drainage ditches as surface water (Wemple et al., 1996). Road crossings commonly found in military training areas may act as barriers to the movement of fish and other aquatic habitats (Furniss et al., 1991). Given the nature of military land use, management or military testing and training, military land managers face the conflicting demands of balancing the primary military mission with legal requirements to protect land and water quality (Milchunas et al., 1999).

The military impacts can result in significant disruptions to the water and nutrient cycles in the freshwater ecosystems and water resources. A few studies in the past have addressed the effects of military training on terrestrial and aquatic ecosystems. For example, Wilson (1988) found that the tank traffic resulted in a significant loss of native species, increased abundance of introduced species, and increased bare soil at a training site located in Manitoba. Milchunas et al. (1999) examined the effects of military vehicles on plant communities and soil characteristics in Pinon Canyon Maneuver Site, Colorado. Whitecotton et al. (2000) examined the impact of foot traffic from military training on soil bulk density, infiltration rate, and aboveground biomass. Recently, Quist

et al. (2003) conducted a study in the Fort Riley Military Reservation, Kansas, to study the effects of military use on terrestrial and aquatic communities.

Understanding human impacts in many landscapes needs the identification of critical landscape elements and analysis of landscape pattern change (O'Neill et al., 1997). Generalized response to military training includes the reduction in native and perennial grasses, abundance of introduced species, and an increase in bare soil. Clearly, these responses will alter biogeochemical and hydrological processes, which regulate nutrient and water dynamics. However, impact of landscape changes to water quality and stream is not well understood (Wand et al., 2001).

In order to maintain and improve water quality, there is an increasing need to understand the relationships among watershed land use and stream ecosystems (Wang et al., 2001). In parallel, perturbation induced effects on water quantity are critical to stream ecology.

One of the many approaches to study the functional interrelations between the hydrology and the stream biota is streamflow characterization and classification. This approach develops hydrologic indices that account for characteristics of streamflow variability that are biologically relevant (Olden and Poff, 2003). Characterization of streams through development of ecologically relevant hydrologic indices is based on long-term streamflow data. However, past studies overlooked the importance of storm-flow data for the development of such hydrologic indices. Flow characteristics are important where changes in land use are anticipated and where alterations to the flow regimes need to be assessed.

The leaching of organic or mineral products from the forest floor provides potential additional effects on water quality. Fluxes of nitrogen through the riparian zone are intrinsically linked to water movement, both over and through the soil, and are also strongly influenced by biological processes occurring in that zone. Nitrogen and organic carbon dynamics in riparian zones are closely interrelated. While many of the factors that can potentially influence nitrogen and carbon fluxes through riparian zones are broadly known, there is presently incomplete quantitative information on the relative importance of the flushing of nitrogen from freshly fallen leaves during precipitation events.

Objectives

The United State's Department of Defense (DOD) policy has established ecosystem management as its approach to manage the military lands by maintaining and improving the sustainability and biological diversity of terrestrial and freshwater ecosystems while supporting human needs, including the DOD missions. In order to identify critical deficiencies and research opportunities on ecosystem management problems on defense installations, the Strategic Environmental Research and Development Program (SERDP) of DOD initiated the SERDP Ecosystem Management Program (SEMP) in December 1997. The objectives of SEMP were to (1) establish long-term research sites on DOD lands for military-relevant ecosystem research, (2) conduct ecosystem research and monitoring activities relevant to DOD requirements and opportunities, and (3) facilitate the integration of results and findings of research into DOD ecosystem management practices.

The goal of this research is to illustrate how an ecohydrological approach could be used to advance our ability to predict the effects of anthropogenic perturbations on water-

vegetation-nutrient interactions in the military installation at Fort Benning, Georgia. The issues addressed in this research encompass many of the main scientific challenges in the military installations' ecohydrology that include the effects of military related perturbations on stream water quality and quantity, and the value of studying nutrient dynamics in the riparian corridors of such regions. The specific objectives of this research include (1) identification and examination of the statistical relationships among water quality parameters and the watershed physical characteristics in low-nutrient watersheds, (2) identification and development of hydrologic indices that characterize the impact of military land management on watersheds, and (3) investigation of the effects of nitrogen leaching from freshly fallen leaves on nutrient dynamics in a riparian area.

Dissertation Organization

Each chapter of this dissertation, except Chapters 1 and 5, is written as a self-contained individual paper focusing on a topic that has not been addressed before. Contributions are in the areas of water quality and land use within a military installation (Chapter 2), development of storm-based hydrologic indices (Chapter 3), and watershed scale nutrient leaching from a riparian area (Chapter 4). Chapter 5 summarizes and concludes the research work

Chapter 2 is an original contribution in the effects of military activities related land use on surface water quality. The major results from Chapter 2 are that the concentrations of total organic carbon, total Kjeldahl nitrogen, total suspended solids, pH, and total phosphorus in the stream show the greatest susceptibility to direct effects of military activities. This chapter also identifies significant statistical relationships among the water quality parameters and the military land uses. These relationships provide the

guidance for maintaining the surface water quality within the Fort Benning military installation.

Chapter 3 is an original contribution in the ecohydrology that presents both annual-based and storm-based methods for determining hydrologic indices that are of ecological importance. Detailed descriptions of the methods for determining these indices and their significance in aquatic ecosystems are described in this chapter. To statistically characterize the hydrologic variation among different watersheds, 32 annual-based hydrologic indices are analyzed. Eight out of 32 annual-based indices are recommended to use for management practices within the Fort Benning military installation. As a new approach to characterize the streamflow variability, 18 storm-based indices are proposed. These indices are grouped into magnitude, frequency, duration, and rate of change of flow. Storm-based indices are compared with annual-based indices. The storm-based methodology provides guidance for measurements of appropriate indices within the military installation.

Chapter 4 is an original contribution to the water quality function of the riparian area. Nitrogen leaching from freshly fallen leaves in a riparian area during the precipitation events is quantified. The observed nitrogen masses in the stream during the precipitation events are compared with the model simulated values. Riparian Ecosystem Management Model (REMM) is used to quantify the nitrogen in the riparian area. In this chapter, details of the nitrogen leaching from the freshly fallen leaves are explored. The results showed that the model effectively captured the trends of litter mass accumulation in the riparian area and subsequent high concentrations during those periods. Analysis

showed that the simulated total Kjeldahl nitrogen masses during the precipitation events were close to the observed masses.

Chapter 5 summarizes and concludes the research work. This chapter contains recommendations to improve management of the water resources within the Fort Benning military installation. Future research needs are also outlined in this chapter.

CHAPTER 2
ECOLOGICAL INDICATORS IN FORESTED WATERSHEDS IN FORT BENNING, GEORGIA: RELATIONSHIP BETWEEN LAND USE AND STREAM WATER QUALITY

Introduction

Ecological monitoring is essential to protect ecological health and integrity. As human activity alters land cover, degradation of water resources begins in the upland areas of a watershed. The first step toward effective ecological monitoring and assessment is to realize that the ultimate goal is to measure and evaluate the consequences of human actions on ecological systems. Human activities that alter land use eventually affect biogeochemical processes that influence water quality and alter ecological processes.

The National Research Council of the United States recently conducted a critical evaluation of indicators used to monitor ecological changes from either natural or anthropogenic causes. During recent decades, efforts have been increasing to develop reliable and comprehensive environmental indicators because of growing environmental concerns (National Research Council, 2000). Indicators rapidly and effectively communicate system status. Ecological indicators help to elucidate both the effects of human activities and natural processes. They can also help to assess future implications of these factors on ecosystem integrity. Once indicators identify areas or elements of the environment that are under stress, successful management of problems can be measured relative to both interim targets and long-term goals.

Indicators that relate key ecological responses to human perturbations provide useful tools to better understand ecological effects and their monitoring and management. A suite of indicators ranging from microbiologic to landscape metrics is necessary to capture the full spatial, temporal, and ecological complexity of impacts (Dale et al., 2002). Evaluation of representative indices across major physical gradients (e.g., soils, geology, land use, water quality and quantity) can signal early environment change and help diagnose the cause of an environmental problem.

Understanding human impacts in many landscapes needs the identification of critical landscape elements and analysis of landscape pattern change (O'Neill et al., 1997). Attention has refocused on relationships among watershed characteristics and stream water quality (Johnson et al., 1997). In order to maintain and improve water quality, there is an increasing need to understand the relationships among watershed land use and stream ecosystems (Wang et al., 2001).

Land use provides information about ecosystem function and characterizes the extent and diversity of ecosystem types. National Research Council (2000) has recommended land use as one of the most effective indicators for ecological assessment. Hydrologists and aquatic ecologists have long known that the pathway by which water reaches to a stream or lake has a major effect on water quality. Early studies on the physical (Harrel and Dorris, 1968) and chemical (Hynes, 1960) characteristics of watersheds focused on the influence of geomorphic characteristics such as drainage area, gradient, and stream order on turbidity, dissolved oxygen concentration, and temperature. Many recent studies examine the influence of terrestrial ecosystems on stream or wetland water quality (Richards and Host, 1994; Richards et al., 1997). Many other studies have

found relationships between land use and concentrations of nutrients in streams (e.g., Hunsaker and Levine, 1995; Johnes et al., 1996; Bolstad and Swank, 1997). Watershed properties constrain in-stream physicochemical and biotic features. Richards et al. (1996) showed that ecosystems could be influenced by land use at regional or broad geographic scales. Osborne and Wiley (1988) found that the distance of urban land cover from the stream effectively predicts stream nitrogen and phosphorus concentrations.

Within a military installation context, land managers are challenged to use the land for military training purposes in a manner that is both ecologically sound and meets military mission requirements (Garten Jr et al., 2003). Lands can suffer a slow degradation if over-utilized by long-term human activities. The heavy vehicles used in mechanized military training cause disturbance of soil structure and can change the physical properties of the soil (Iverson et al., 1981). In rangelands, tracked vehicle traffic affects the hydrological characteristics (Thurow et al., 1993). Trampled vegetation, vehicle tracks through undisturbed area, and erosion caused by the overuse of trails are some examples of the visible degradation to a landscape caused by military training exercises. Few studies have developed predictive relationships among watershed physical characteristics and surface water chemistry specific to military land use and low-nutrient systems.

In the coastal plain of the Apalachicola-Chattahoochee-Flint (ACF) river basin, cropland and silvicultural land in upland areas is separated from streams by relatively undisturbed riparian flood plain and wetland habitats (Frick et al., 1998). This is in contrast to many intensively farmed areas of the United States where wetlands have been drained, channelized or filled, and little or no riparian buffers remain between cropland

and streams. Frick et al. (1998) reported that the lower nutrient concentrations in streams within the ACF river basin could partially be attributed to wetland buffer areas, and minimal use of pesticides as compared to other areas of the United States. Other studies in the southeastern coastal plain watersheds (e.g., Lowrance 1984; Lowrance et al., 1992; Perry et al., 1999; Fisher et al., 2000) focused primarily on the agricultural impacts, and urbanization on stream water quality. The contribution of areas affected by military training to nutrient discharges, specifically in Fort Benning watersheds, is yet to be quantified.

This paper identifies and examines the statistical relationships among water quality parameters and the watershed physical characteristics in seven low-nutrient watersheds located in the Fort Benning military installation, Georgia. It is hypothesized that surface water quality parameters can be used as indicators of ecological changes in watersheds.

Study Area

The Fort Benning Army Installation occupies approximately 73,503 ha in Chattahoochee, Muscogee, and Marion Counties of Georgia and Russell County of Alabama (Figure 2-1). The climate at Fort Benning is humid and mild. Rainfall in this region occurs regularly throughout the year. July and August are the warmest months with average daily maximum and minimum temperatures of 37 and 15°C. An average daily maximum and minimum temperature of 15.5 and -1°C are reported in the coldest months, January and February. Annual precipitation averages 1050 mm with October being the driest month (Dale et al., 2002). Most of the precipitation occurs in the spring and summer as a result of thunderstorms. Heavy rains are typical during the summer but can occur in any month. Snow accounts for less than 1% of the annual precipitation.

Fort Benning is located within the southern Appalachian Piedmont and Coastal Plains. The northern boundary of the installation lies along a transition zone between the Piedmont and Upper Coastal Plain. The soils in the area are dominated by loamy sand with some sandy loam. Following establishment of the installation in 1918, with subsequent additions in 1941, we see that heavy training impacts only selected, mostly upland, portions of the installation. Many areas are maintained as safety buffers, and have little military use. Timber management includes harvesting and thinning. The loblolly and longleaf pine forests are subjected to regular low-level fires for management purposes (Dale et al., 2002).

Methods of Study

Description of Watersheds

The study watersheds, Bonham-1 and Bonham-2, Bonham, Little Pine Knot, Sally, Oswichee, and Randall (named for the creek which drains the watershed), within Fort Benning represent a range of region's soils, topography, land use, and vegetation communities (Figure 2-1). These watersheds have a heterogeneous land cover predominantly consisting of either forested or open areas. Forested areas are broadly characterized as mixed pine and hardwoods or pine that are mostly 30-50 years old with the soils in A-horizon range approximately 1-10 cm in depth (Garten Jr et al., 2003). Open areas are either military, brush, or managed wildlife openings. Other cover includes upland and bottomland hardwood forests. The military openings are clear-cut parcels of land dominated by grass and bare soil that are used as military training grounds. The brush openings consist of tall grass and immature hawthorn. The wildlife openings are natural openings in the forests that are vegetated primarily by grass. Land

impacts due to heavy military activities (e.g., infantry, artillery, wheeled, and tracked vehicle training) occur only in selected portions in these watersheds.

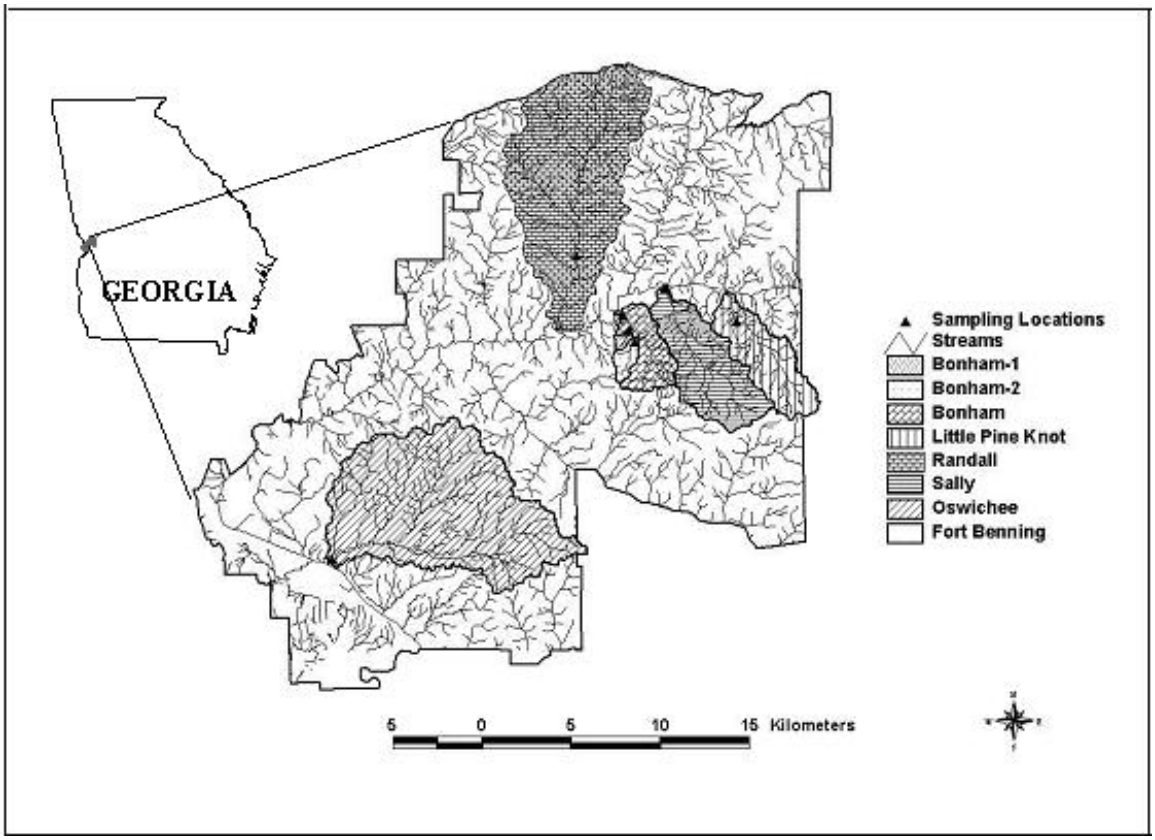


Figure 2-1. Study watersheds, Bonham-1, Bonham-2, Bonham, Little Pine Knot, Randall, Sally, and Oswichee, in the Fort Benning military installation. Also shown are stream network and sampling locations.

Characterization of Disturbance Categories

A disturbance index (DIN) was defined to characterize the watersheds at Fort Benning into two disturbance categories: low-impact and high-impact. A DIN is the sum of the area of bare ground on slopes greater than 3 degrees and on roads, as a proportion of the total watershed area (Maloney et al., in press). The road areas were estimated by multiplying their length by the measured average width of 20 m. Percentage of bare ground was determined by using TM imagery and slope was derived from digital

elevation maps. The TM imagery and digital elevation maps were obtained from Strategic Environmental Research and Development Plan (SERDP)'s Ecosystem Management Project (SEMP) database. Watersheds having a disturbance index from 0 to 11% are designated as low-impacted watersheds. High-impacted watersheds have disturbance indices greater than or equal to 11%.

Collection and Analysis of Stream Samples

Surface water quality data were collected at seven streams biweekly from October 2001 to November 2002, and monthly thereafter to September 2003. Water samples were collected in high-density polyethylene bottles. Bottles were soaked in de-ionized water and rinsed with sample water prior to collection. The filtration was conducted at the sampling sites using 0.45 μm pore size polyethersulfone membranes. Filtered sample was used to determine chloride (Cl) concentration, whereas raw sample was used for total suspended solids (TSS) determination. Unfiltered samples for analyzing total Kjeldahl nitrogen (TKN), total phosphorus (TP), and total organic carbon (TOC) were acidified using double distilled sulfuric acid. The stream water pH, conductivity, and temperature were measured at the time of sampling. All samples were kept cool in an icebox, transported to the Soil and Water Science Department laboratory, University of Florida, and refrigerated until analyzed. All samples were analyzed using standard methods (American Public Health Association, 1992).

Statistical Analyses

The Pearson's correlation coefficients were calculated to examine the strength and significance of the relationships between a watershed physical characteristic and a water quality parameter. Two-sample t-tests were performed at 5% level of significance to test whether mean values of watershed physical characteristics and water quality parameters

differ between low- and high-disturbance watersheds. Characteristics showing significant correlations with a water quality parameter were considered for stepwise multiple linear regression models. Only variables having less than or equal to 0.05 significance level were retained in the regression models.

Results

Watershed Physical Characteristics

The watersheds' physical characteristics are summarized in Table 2-1. Most of the watersheds are highly vegetated (70% or more) except Oswichee (38%) with the majority characterized by pine and mixed pine and hardwoods. Deciduous forest typically covers only a small percentage of these watersheds. However, Bonham-1 consists of 27% of deciduous forest. The study watersheds range from less than 1 to 84 km^2. The topographic characteristics of study watersheds are typical of forested watersheds of southeastern coastal plain (Lowrance, 1992; Perry et al., 1999). Average elevations vary from 104 to 148 m above mean sea level. Maximum slopes vary from 4 to 6 degrees. Sandy soils are common in most of the study watersheds. However, loamy soils cover most of the Sally and Oswichee watersheds. Bottomlands comprise 6 to 20% of the watershed. The military training extent (0 to 6%) is relatively small. Total bare lands in these watersheds comprise 9 to 21% of the watershed area, of which 1 to 8% of the total area is unpaved roads and trails. This extent and variability of military training and bare land are typical of the entire Fort Benning installation.

Some watershed characteristics are strongly correlated at significance level of 0.05 or lower (Table 2-2). Significant positive correlations exist for pine with the bottomland wetlands, deciduous vegetation with stream density, number of roads crossing streams with road length and percent of loam, and disturbance index with percent bare land.

Negative correlations were found for mixed vegetation with percentage of loamy soil and number of roads crossing streams (NRC), military areas with normalized difference vegetative index (NDVI) and stream density, and DIN, and road density with NRC.

Table 2-1. Physical characteristics of study watersheds in Fort Benning, Georgia. Acronyms BON-1, BON-2, BON, LPK, OSW, Ran, and SAL represent Bonham-1, Bonham-2, Bonham, Little Pine Knot, Randal, and Sally, respectively.

	Watersheds						
Physical Characteristics	BON-1	BON-2	BON	LPK	OSW	RAN	SAL
Topography							
Area, km^2	0.76	2.21	12.73	18.01	83.39	74.38	25.31
Average Elevation, m	121.8	133.5	125.5	146.3	104.2	136.8	136.8
Average Slope, degree	5.46	4.89	5.04	5.32	4.48	4.57	5.42
Vegetation							
Pine, %	28	30	40	41	26	58	48
Deciduous, %	27	6	8	2	0	3	12
Mixed, %	39	50	22	34	5	9	15
Wetland, %	6	8	9	17	7	20	10
Military Land, %	0	6	5	2	5	3	2
NDVI	0.36	0.30	0.32	0.34	0.35	0.34	0.36
Soil							
Sandy loam, %	78	69	69	72	24	68	49
Loamy sand, %	9	9	31	28	73	32	51
Road							
Road Length, km	3.6	11.4	51.6	56.6	196.6	415.1	97.6
Road Density, km/km^2	4.8	5.1	4.1	3.1	2.4	3.1	3.8
Stream							
Stream Length, km	2.6	3.9	29.1	43.3	170.6	323.5	65.2
Stream Density, km/km^2	3.4	1.7	2.3	2.4	2.1	2.4	2.6
Stream Order	2	2	4	4	5	6	4
Other							
No. of Roads Crossing Streams	1	2	13	11	55	43	21
Bare Land, %	1	11	11	4	4	4	7
Disturbance Index, %	11	21	19	11	9	10	15

Table 2-2. Pearson correlation coefficients between watershed characteristics. Characteristics are acronymed as follows: Pine forest (PIN), Deciduous forest (DCD), Mixed forest (MXD), Wetland (WET), Military land (MIL), Sandy Soil (SND), Loamy Soil (LOM), Road Length (RDL), Road Density (RDN), Stream Density (STD), Normalized Difference Vegetative Index (NDVI), No. of Roads Crossing Streams (NRC), % Bare Land (PBL), and Disturbance Index (DIN). *, **, and *** indicates significance at or below 0.05, 0.01, and 0.001 probability levels, respectively.

	PIN	DCD	MXD	WET	MIL	SND	LOM	RDL	RDN	STD	NDVI	NRC	PBL
DCD	-0.25												
MXD	-0.43	0.40											
WET	0.96***	-0.33	-0.42										
MIL	-0.20	-0.65	0.00	-0.18									
SND	0.21	0.44	0.67	0.17	-0.30								
LOM	0.06	-0.50	-0.85*	0.14	0.15	-0.93***							
RDL	0.63	-0.47	-0.74	0.50	0.05	-0.28	0.41						
RDN	-0.26	0.65	0.81*	-0.34	-0.01	0.63	-0.81*	-0.64					
STD	0.01	0.84*	0.04	-0.04	-0.92***	0.34	-0.24	-0.15	0.17				
NDVI	0.10	0.41	-0.48	0.07	-0.80*	-0.30	0.41	0.22	-0.39	0.74			
NRC	0.23	-0.58	-0.90**	0.19	0.21	-0.76*	0.83*	0.81*	-0.86**	-0.27	0.33		
PBL	0.00	-0.28	0.25	0.09	0.72	0.04	-0.11	-0.34	0.40	-0.67	-0.79*	-0.30	
DIN	-0.10	0.05	0.52	-0.07	0.55	0.30	-0.42	-0.52	0.71	-0.44	-0.77*	-0.58*	0.93***

Water Quality Parameters

Water quality parameters in the study watersheds varied over the sampling period and among watersheds. Variability of water quality parameters among watersheds is observed (Figure 2-2). Mean pH in the study watersheds ranged from 4.2 to 7.0. Mean conductivity ranged from 16.4 to 44.5 μS/cm. Mean temperatures varied from 17.5 to 20.8^{o}C. Low concentrations of TP and TKN were observed in all the watersheds under study as compared to forested watersheds in the southeastern coastal plain watersheds (Lowrance et al., 1984), and across the United States (Meader and Goldstein, 2003; Fisher et al., 2000). Concentrations of TKN, TP, and Cl were often below the detection limit. Mean concentrations of TP varied widely, ranging from 0.003 to 0.020 mg/L and TKN varied from 0.20 to 0.35 mg/L; TOC from 1.35 to 3.33 mg/L; Cl from 1.46 to 4.13 mg/L; and TSS from 4.15 to 10.30 mg/L. As depicted in Figure 2-2, each stream exhibited distinct water quality signatures with the exception of temperature and TSS. Seasonal variations in the water quality parameters are responsible for much of this observed variability among watersheds.

Stream pH fluctuated more during June and July and was elevated during December through February. Conductivity values showed slight fluctuations from May to July. On multiple occasions, high conductivity values were observed in the Randall stream. Chloride, TKN, and TP showed distinct seasonal patterns. The concentrations of these parameters were low from June to September, and high from March to May and from October to December. In contrast, TOC peaked from August to October and again from March to July. Higher concentrations of TSS were observed from July to September in all the streams.

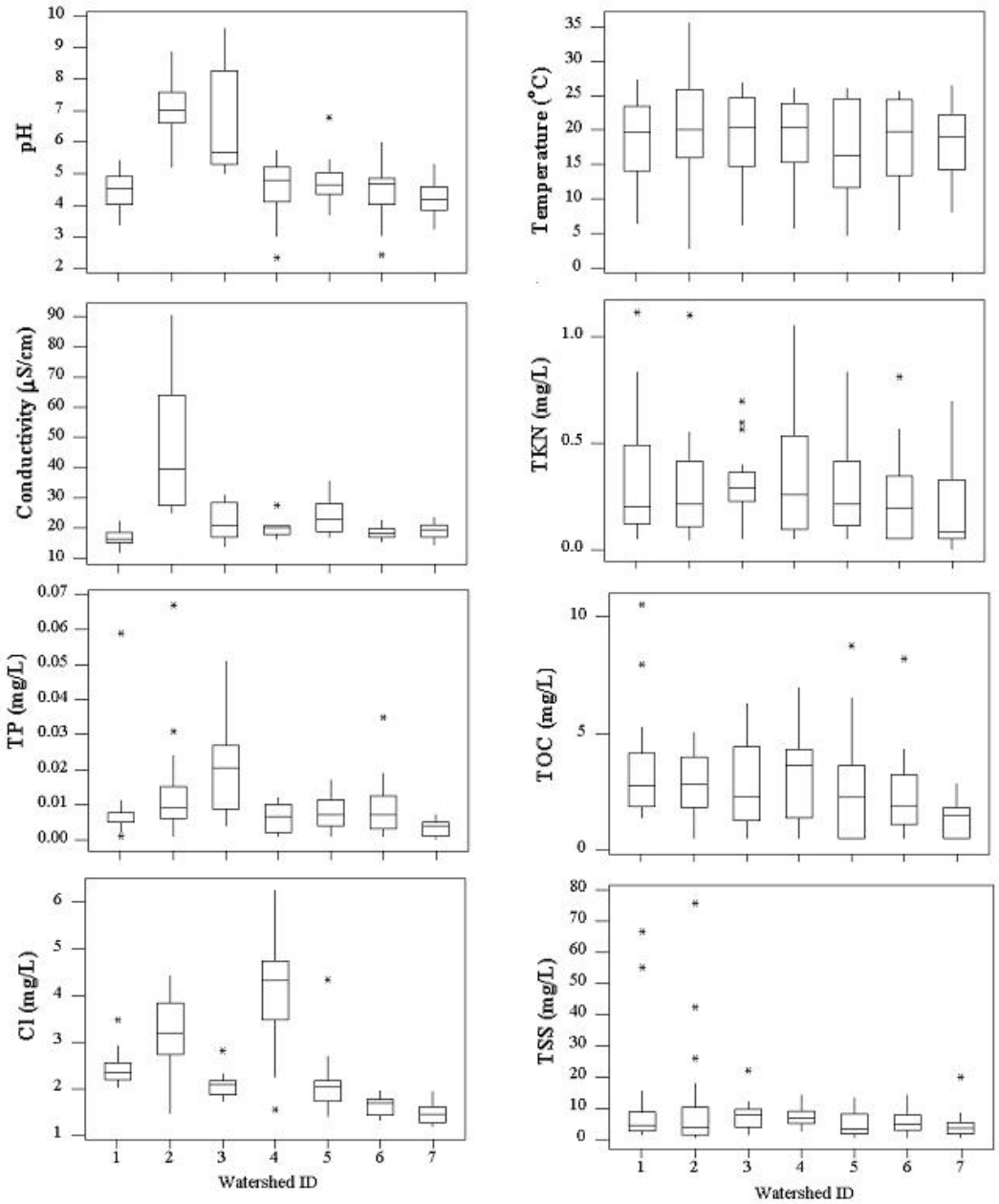


Figure 2-2. Box plots of water quality parameters. Each plot consists of outliers, most extreme data, 75th, 50th, and 25th percentile values. Watershed IDs represent- 1: Bonham-1, 2: Randall, 3: Oswichee, 4: Little Pine Knot, 5: Sally, 6: Bonham, and 7: Bonham-2.

Effects of Disturbance Categories

Table 2-3 presents the t-test results of the comparison of physical characteristics and water quality parameters based on the watershed disturbance level. The only physical characteristic having a significant difference ($\alpha = 0.04$) between these two groups was DIN. While the low-impact watersheds tended to have higher chemical

Table 2-3. t-test results for differences in mean values of watershed physical characteristics and water quality parameters. * indicates significance at or below 0.05 probability level. NS indicates non-significant difference at the 0.05 probability level.

	Low-Impacted		High-Impacted		
	Mean	SD	Mean	SD	
Watershed Characteristics					
Pine, %	38.3	14.8	39.3	9.0	NS
Deciduous, %	8.0	12.7	8.7	3.1	NS
Mixed, %	21.8	17.2	29	18.5	NS
Wetland, %	14.5	9.5	16	7.2	NS
Military Land, %	2.4	2.0	4.3	2.3	NS
Sand, %	60.5	24.7	62.3	11.5	NS
Loam, %	35.5	26.9	30.3	21.0	NS
NDVI	0.35	0.01	0.32	0.03	NS
Road Length, km	168	184	53.5	43.2	NS
Road Density, km/km^2	3.3	1.0	4.3	0.7	NS
Stream Density, km/km^2	2.6	0.6	2.2	0.4	NS
No. of Roads Crossing Streams	27.5	25.6	12	9.5	NS
Disturbance Index, %	10.2	0.9	18.3	3.1	*
Water Quality Parameters					
pH	5.6	1.3	4.5	0.3	NS
Temperature, ^{o}C	19.3	0.9	18	0.3	NS
Conductivity, µS/cm	25.9	13.3	20.4	2.5	NS
TKN, mg/L	0.3	0.03	0.2	0.05	NS
TP, mg/L	0.011	0.006	0.007	0.003	NS
TOC, mg/L	2.9	0.4	2.1	0.7	NS
Cl, mg/L	2.4	0.5	1.8	0.3	NS
TSS, mg/L	9.1	2.2	4.8	0.5	*

concentrations than the high-impact watersheds, only TSS showed significant difference ($\alpha = 0.03$). Even though the results showed no significant statistical differences at a confidence level of 95%, the t-test results of all water quality parameters, except conductivity and TP, showed significant differences at 80% confidence interval between high- and low-impacted watersheds. The relatively small sample size and natural variability among watersheds may have limited the ability to discern significance differences.

Relationship between Watershed Physical Characteristics and Water Quality Parameters

Correlation and regression analyses were performed to identify relationships among the watershed physical characteristics and the water quality parameters. Table 2-4 shows that each water quality parameter had a significant relationship with one or more watershed physical characteristics (Table 2-4). The correlation results show that decreasing mixed vegetation increased pH and TP. Sandy and loamy soils had opposite effects on TP. An increase in sandy soil decreased TP, whereas an increase in loamy soil increased TP. Increasing military land decreased TOC. Temperature, pH, conductivity, and Cl increased as the road length increased. The number of roads crossing streams had positive correlations on pH and TP. Percent bare land was negatively correlated with TOC and TSS. Disturbance index was negatively correlated with TKN and TOC.

Graphical relationships provide insight into nonlinear relationships that exist between indicators and response variables. Figures 2-3 to 2-6 show some of the most striking relationships between watershed characteristics directly and/or indirectly affected by management of military lands and their effect on water chemistry. As the % military land increases, the TKN, TOC, and TSS decrease in a linear fashion. However,

Table 2-4. Pearson correlation coefficients between watershed characteristics and water quality parameters. Characteristics are acronymed as follows: Pine forest (PIN), Deciduous forest (DCD), Mixed forest (MXD), Wetland (WET), Military land (MIL), Sandy Soil (SND), Loamy Soil (LOM), Road Length (RDL), Road Density (RDN), Stream Density (STD), Normalized Difference Vegetative Index (NDVI), No. of Roads Crossing Streams (NRC), % Bare Land (PBL), and Disturbance Index (DIN). *, **, and *** indicates significance at or below 0.05, 0.01, and 0.001 probability levels, respectively.

	pH	Temperature	Conductivity	TKN	TP	TOC	Cl	TSS
PIN	0.36	0.78	0.54	0	-0.09	0.15	0.56	-0.14
DCD	-0.47	-0.37	-0.40	-0.12	-0.33	0.39	0.13	0.42
MXD	-0.79*	-0.55	-0.38	-0.39	-0.83*	-0.24	-0.49	-0.11
WET	0.24	0.62	0.44	0.11	-0.11	0.18	0.37	-0.29
MIL	0.09	0	-0.10	-0.50	0.10	-0.87**	-0.47	-0.53
NDVI	0.32	0.11	0.10	0.60	0.46	0.80*	0.55	0.54
PBL	-0.45	-0.25	-0.51	-0.72	-0.42	-0.81*	-0.68	-0.87**
DIN	-0.65	-0.36	-0.60	-0.84*	-0.63	-0.76*	-0.65	-0.72
SND	-0.50	-0.02	0.14	-0.19	-0.78*	0.15	0.06	0.15
LOM	0.57	0.16	0.03	0.39	0.81*	0.07	0.07	-0.14
RDL	0.94***	0.95***	0.82*	0.19	0.57	0.09	0.79*	0.30
RDN	-0.75*	-0.42	-0.54	-0.72	-0.78*	-0.35	-0.32	-0.15
NRC	0.93***	0.59	0.51	0.36	0.90**	0.06	0.47	0.19
STD	-0.14	-0.11	-0.01	0.39	-0.04	0.82*	0.44	0.65

disturbance index may operate as a threshold indicator of pH and TSS where pH decreases in response to a relatively low level DIN while the TSS threshold for DIN impact is somewhat higher. No significant relationships are found between the water quality parameters and extent of pine and deciduous forest. Similarly, wetland showed no effect on these parameters.

Stepwise multiple regressions identified relationships between the water quality parameters and watershed physical characteristics that are susceptible to the disturbances (Table 2-5). A statistically significant regression model was found for every water quality parameter. Prediction of pH variability among watersheds is particularly well captured by pine forest and road length. The regression relationships indicate that all of

the water quality parameters depend on at least one aspect of military management. Several water quality parameters, Cl, TP, TOC, TSS, and TKN, depend only on management aspects of the military installation. For example, Cl depends on change in military land and road length. Total phosphorus is strongly related to the number of roads crossing streams. However, the influence of vegetation and soils characteristics is clearly important in pH, conductivity, and temperature. For example, conductivity appeared to be well captured by area covered by sandy soil and the number of roads

Table 2-5. Stepwise multiple regression models for water quality parameters. pH is unitless, temperature is measured in degrees centigrade, conductivity is measured in mS/cm, TP, TKN, TOC, Cl, and TSS are measured in mg/L. Pine forest (PIN), Military land (MIL), Sandy Soil (SND), Loamy Soil (LOM), Road Length (RDL), Road Density (RDN), No. of Roads Crossing Streams (NRC), Percent Bare Land (PBL), and Disturbance Index (DIN) are the independent variables retained in the regression analyses. *, **, and *** indicates significance at or below 0.05, 0.01, and 0.001 probability levels, respectively.

Water Quality Parameters	Independent Variables Retained and Regression Equations	R^2
	Pine, Road Length	
pH	5.50 – 0.0382 PIN + 0.00924 RDL	0.98***
	Military, Road Length	
Cl	2.19 + 0.145 MIL + 0.00344 RDL	0.90**
	No. of Roads Crossing Streams	
TP	0.00451 + 0.000236 NRS	0.81**
	Sandy Soil, No. of Roads Crossing Streams	
Conductivity	- 24.2 + 0.552 SND + 0.666 NRS	0.80*
	Loamy Soil, Road Density	
Temperature	26.1 – 0.051 LOM – 1.49 RDN	0.77*
	Military Land	
TOC	3.45 – 0.274 MIL	0.76**
	Percent of Bare Land	
TSS	11.2 – 0.648 PBL	0.76**
	Disturbance Index	
TKN	0.445 - 0.0109 DIN	0.70*

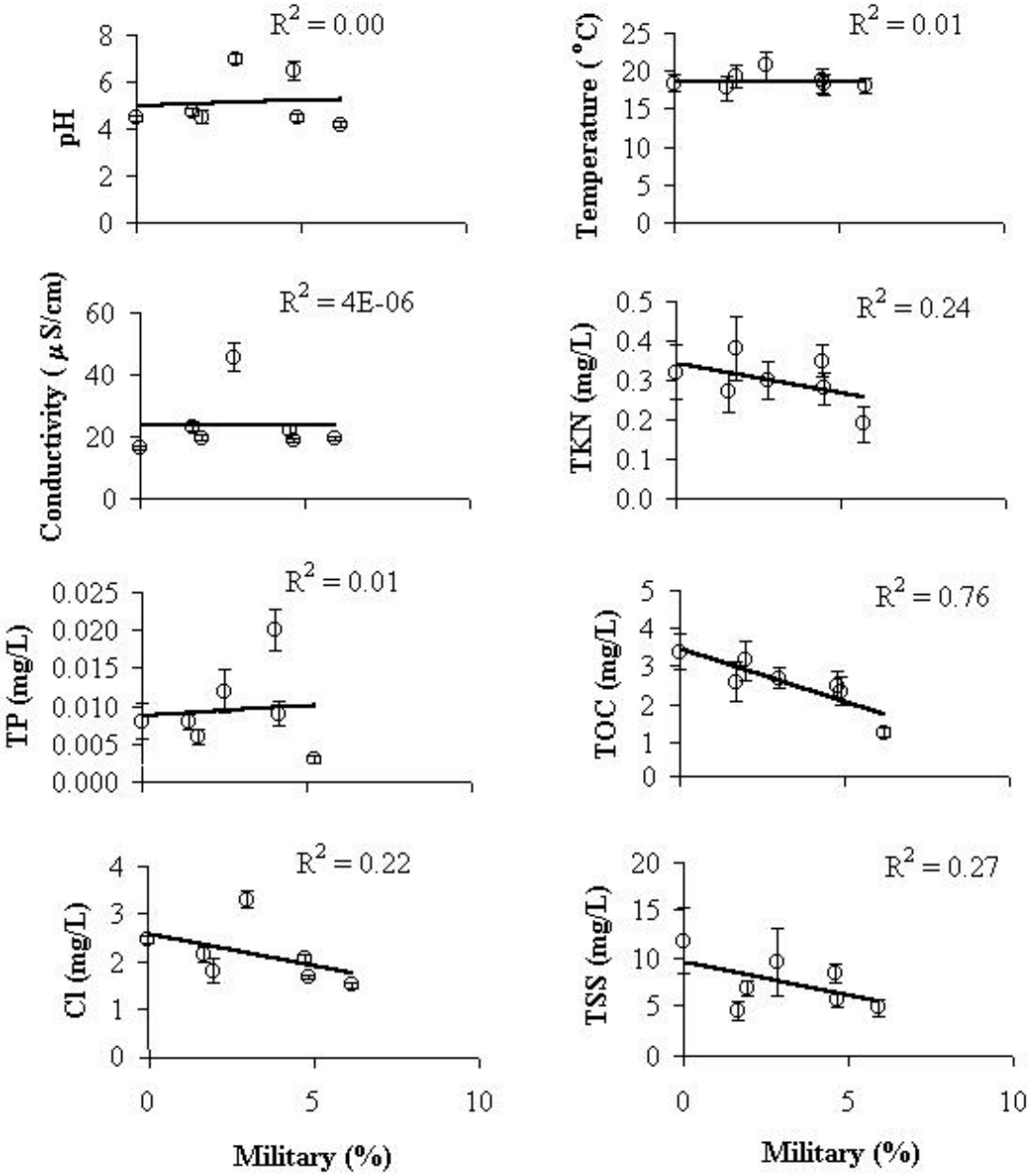


Figure 2-3. Relationships between military land and water quality parameter. Vertical bars represent standard errors.

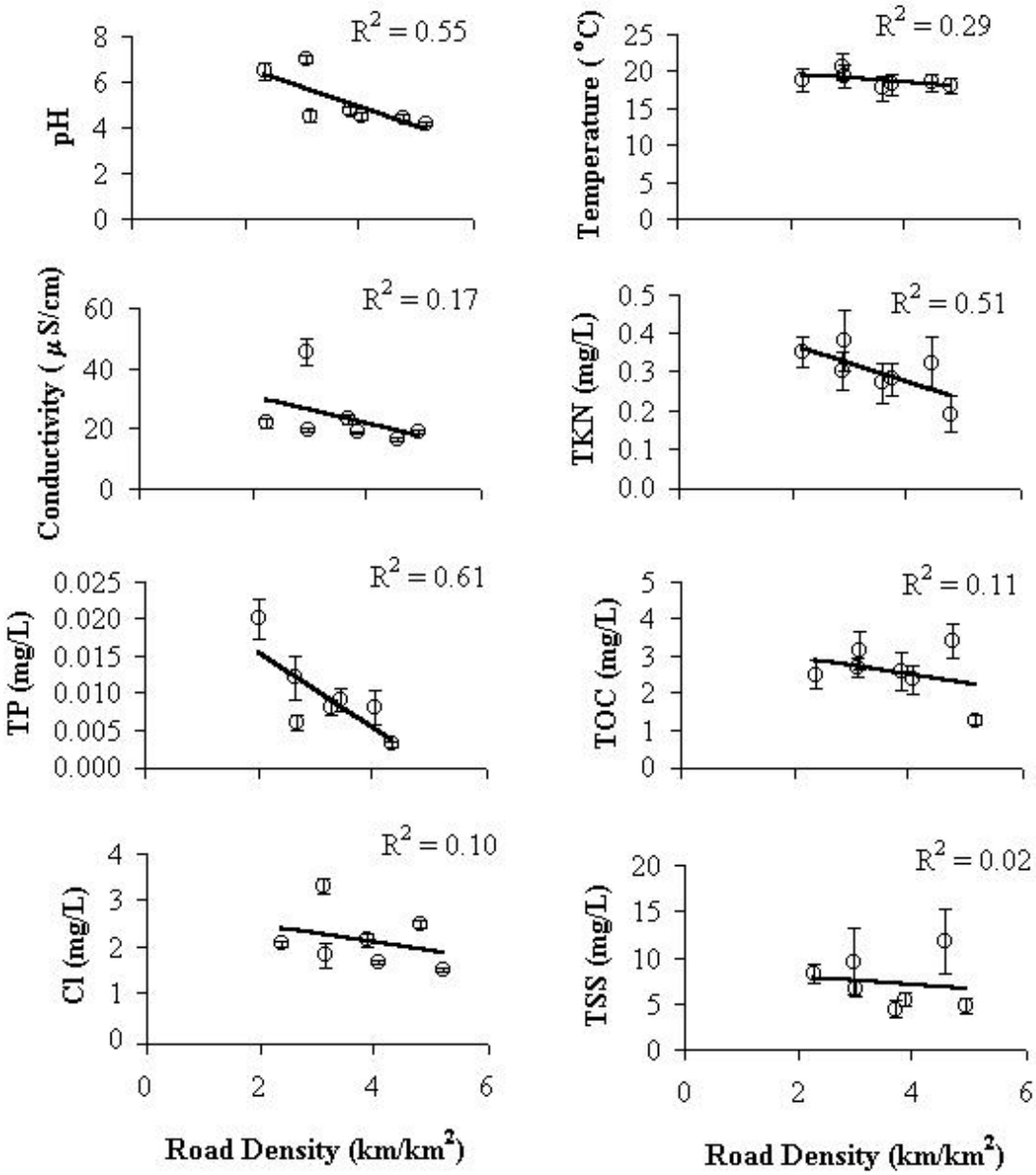


Figure 2-4. Relationships between road density and water quality parameter. Vertical bars represent standard errors.

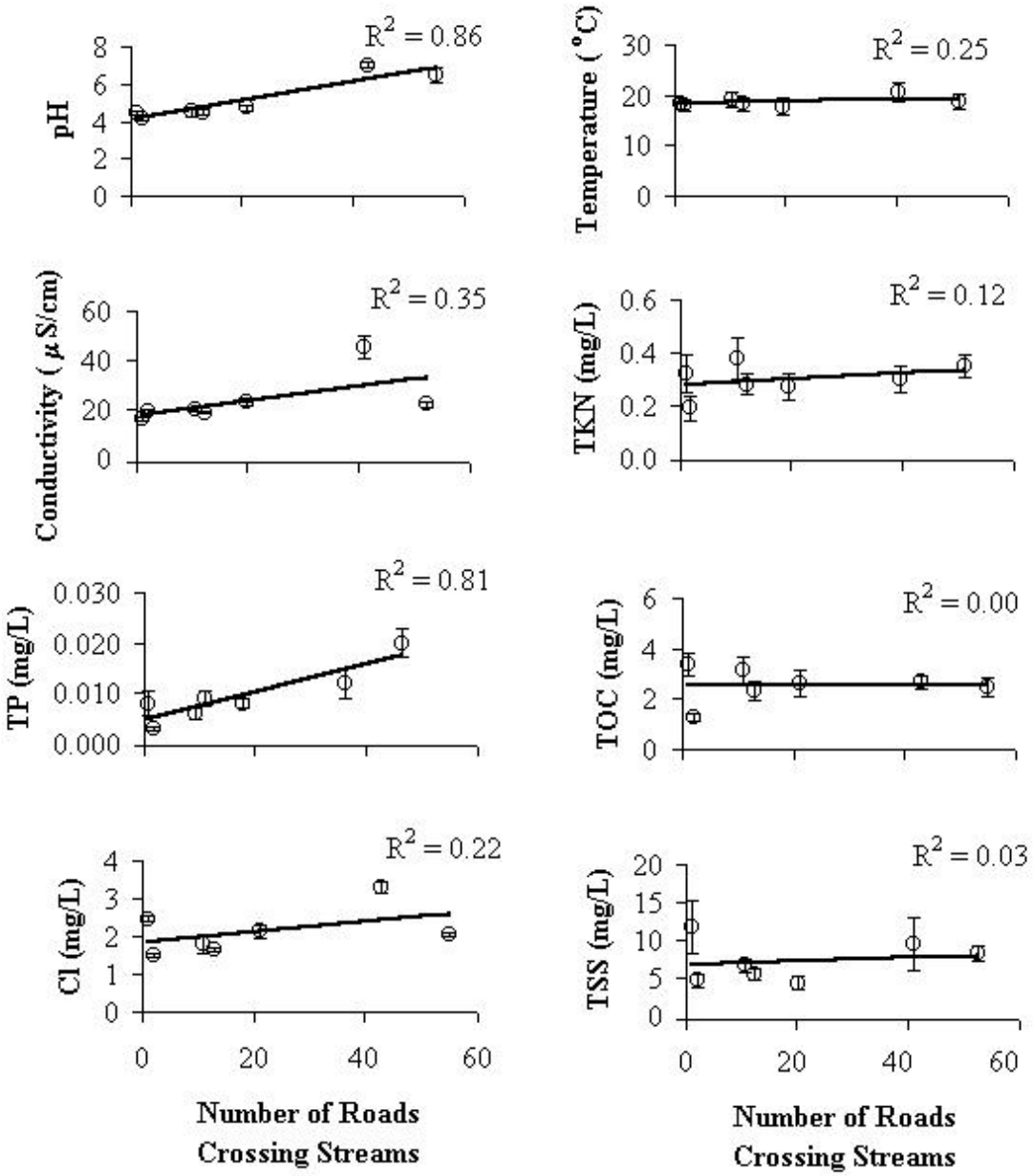


Figure 2-5. Relationships between number of roads crossing streams and water quality parameter. Vertical bars represent standard errors.

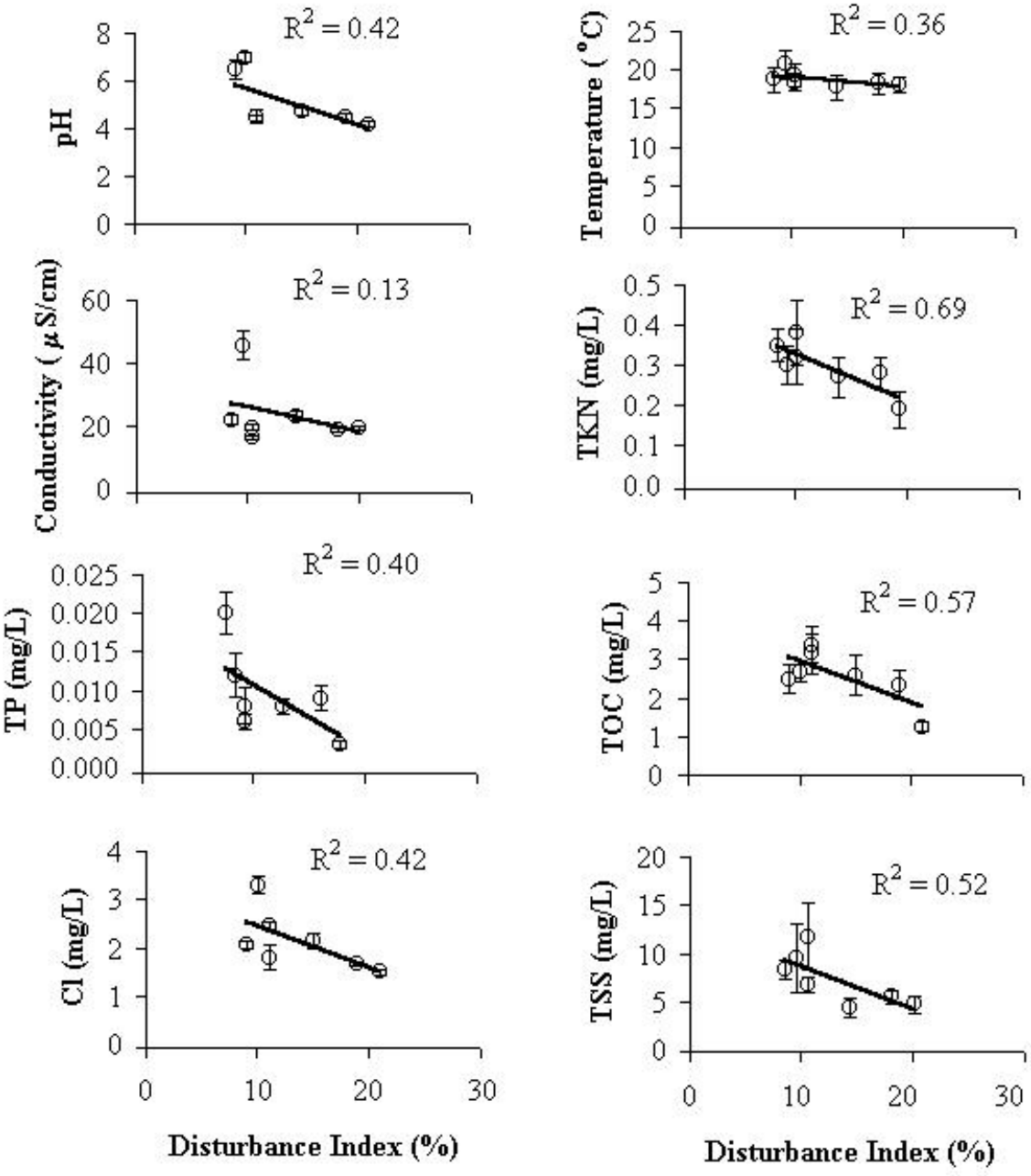


Figure 2-6. Relationships between disturbance index and water quality parameter. Vertical bars represent standard errors.

crossing streams, whereas temperature was captured by loamy sand soil and road density. Overall, the regression models show that it is possible to quantify the effects of watershed physical characteristics on the water quality parameters.

Discussion

Variations in stream water chemistry among the study watersheds reflect differences in biogeochemical reactions occurring in the watersheds. The results of this study indicate that even in low impact watersheds, physical characteristics may be used to explain variations in stream water chemistry and, by inference, the relative watershed disturbance levels. The observed variability in many of the chemical parameters studied in these watersheds can be attributed to physical characteristics of the watersheds or land management patterns within the watersheds as evidenced by the road network, forestry practices, and military training.

Diverse human activities interact to affect conditions in watersheds and water bodies. Sites of interest can be grouped and placed on a gradient according to activities and their effects. The results of this study suggest that the vegetation type, road length, number of roads crossing streams, and disturbance index are important predictors of water quality variability. Vegetation cover was related to stream pH and TP (mixed forest) and conductivity (pine forest). However, deciduous forest cover was not related to any of the water quality parameters suggesting a limited effect in organic matter and nutrient production and variability as observed among Fort Benning watersheds. However some other subtle landscape changes resulted in relatively larger impacts on water quality parameters. Low-impact watersheds tend to produce higher concentrations of nutrients in the streams. This can be attributed to the availability of more soil organic

matter and the rapid biogeochemical processes occurring in the low-impact watersheds as compared to the high-impact watersheds.

It is extremely difficult to capture all aspects of human influence in a single graph or statistical test. However, sometimes meaningful chemical patterns can be lost by excessive dependence on the outcome of menu-driven statistical tests (Karr and Chu, 1999). Figures 2-3 to 2-6 depict several different aspects of stream's chemical conditions against several measures of human influences, such as military land use, disturbance index, number of roads crossing streams, and road density. The distribution of circles in most of these figures illustrate that a chemical metric indicates little about a condition simply because it does not correlate strongly with a single surrogate of that condition. However, where the relationship between human influence and stream's chemical response is strong, statistics and graph agree.

The correlation tests identify linear relationships between a chemical response and watershed characteristics. Weak statistical correlations observed in these analyses may have missed important chemical patterns. For example, nonlinear patterns were observed for forest types, bare land (not shown here), and disturbance index (Figure 2-6). The plots in Figure 2-6 show a step-function for TSS and pH. The scatter of this dataset shows little or no statistical significance, but can be interpreted chemically. For TSS, those watersheds having a disturbance index of 11% or lower had a higher level than those with a greater disturbance index.

When a number of variables interact to influence water quality conditions, it may be difficult to explain observed variability in a single plot against one dimension of human influence (Figure 2-4). Chemical responses were plotted against the road

densities for various watersheds. The Pearson correlation coefficient for TP was significant capturing human influence on this chemical parameter. The response of TP is visibly distinguished from others. A similar discussion is true for military land with TOC (Figure 2-3); number of roads crossing streams with pH and TP (Figure 2-5); and disturbance index with TKN (Figure 2-6).

The relationships between water quality parameters and physical characteristics indicate that disturbances in low nutrient forested environments decrease some chemical signatures. Watersheds with more roads, e.g., Randall and Oswichee, have relatively high pH, conductivity, and Cl compared to the watersheds with fewer roads. Watersheds with a small portion of military land, e.g., Bonham-1, Sally, and Little Pine Knot, have relatively high TOC concentrations. In contrast, watersheds characterized by higher road densities, e.g., Bonham and Bonham-2, had low TP concentrations. Higher disturbance index, similar to the road density, showed lower TKN and TOC concentrations in the streams. Mixed vegetation, road length, percent of bare land, DIN, and number of roads crossing streams were able to capture most of the variability in water quality parameters.

In a watershed scale study conducted in Ontario, Canada, Sliva and Williams (2001) found a negative correlation of forested land cover with TSS and chloride. In contrast, Johnson et al. (1997) showed a positive relationship of forest with TSS in a study of landscape influence on water chemistry in the Saginaw Bay watershed of central Michigan. Johnson et al.'s (1997) results indicated that row crop agriculture had the highest effect on total nitrogen, nitrate, and total dissolved solids. They also observed that urban and forest areas were positively correlated good predictors of TSS, whereas row crop agriculture was positively correlated with total nitrogen. Basnyat et al. (1999)

reported a positive association of TSS with agricultural practices in the Fish River watershed, Alabama. The Fort Benning installation is characterized by relatively low variability in forest cover and suggests, in contrast to other studies, that neither TSS nor Cl may be related to forest cover under existing land management practices. Instead, Fort Benning's road extent and percent bare land are better predictors of TSS and Cl.

Most studies identified urban land use as a dominant factor causing elevated total nitrogen and nitrate concentrations in the streams (Hill, 1981; Osborne and Wiley, 1988). Sponseller et al. (2001) found positive correlation of total inorganic nitrogen with percentage of non-forested land in southwestern Virginia watersheds. A negative correlation of TKN to DIN, in this study, is consistent with studies (e.g., Sponseller et al., 2001; Hunsaker and Levine, 1995; Johnson et al., 1997) that have shown stream nitrogen concentrations to be good predictors of non-forest area at the watershed scale.

In a study of 101 watersheds in New Zealand, Close and Davis-Colley (1990) found that between 60 and 80% of the variance in conductivity, total nitrogen, and nitrate was accounted for by landscape factors including geology and land use. However, in that same study, landscape factors accounted for only 50% of the variance in ammonia and phosphorus species. Their results parallel those found at Fort Benning in that no strong relationships were observed between land use and TP. The strong negative correlation of loamy sand soil with TP in Fort Benning is consistent with Hill's (1981) study, conducted in a sandy loam region similar to portions of Fort Benning watersheds that reported negative correlations between phosphorous concentrations and abandoned farmland and forest.

Most variations in stream water chemistry are driven by climatic and biotic factors and are therefore largely governed by the processes that are taking place in the terrestrial part of the watershed such as natural or human induced vegetation cover changes (Semkin et al., 1994). Our results show interactions among landscape factors and water quality indicators. Results also indicate that it is possible to observe the response of these water quality parameters to physical attributes of watersheds. The importance of water quality parameters in the present study appeared to be attributable to the perturbations related to military training and associated parameters within a watershed as these parameters clearly captured the changes in physical parameters that are more sensitive to such kind of influences.

Conclusion and Recommendations

Sometimes a single variable can capture and integrate multiple sources of influence. More often, a small number of ecological attributes provide reliable signals about ecological condition. Water chemistry prediction using watershed physical characteristics in this study showed mixed results compared to the other investigations. However, most of the watershed physical characteristics used in our analysis did explain the variability in water quality parameters. This study documented strong relationships between certain watershed physical characteristics that are more susceptible to human induced perturbations, specifically military related disturbances, and water quality parameters in military installation at Fort Benning. Watersheds with more roads crossing streams tended to produce more TP. Total Kjeldahl nitrogen and TOC variations were well captured by DIN and extent of military land, respectively. Total suspended solids' variability, on the other hand, was captured by the percent of bare land within a watershed. Road length captured most of the variability in pH and Cl. Conductivity and

temperature values were dependent on soil types and road characteristics. The variations in stream water chemistry are largely attributable to disturbance levels and the types of biogeochemical reactions occurring in the watersheds. Regression results suggest that TOC, TKN, and TSS were useful indicators of watershed physical characteristics as they are more susceptible to direct effects of military activities. Although pH, conductivity, and TP showed good correlations with the road length, these parameters indicated strong but indirect influence of military training activities on watersheds.

Foreseeing a single indicator of water quality that would be sensitive to all kinds of perturbations in the watershed is extremely difficult. Ability to detect perturbations can be related to spatial and temporal scales. It is necessary to recognize the effects of natural disturbances on ecosystem structure and functioning. It is suggested that priorities for determining ecological indicators specific to water quality should include (1) development of framework to determine proper reference states within watershed against which to detect loss of ecosystem health, (2) broadening our knowledge of ecosystem sensitivity to perturbations of varying intensity, spatio-temporal distribution, and type, and (3) development of suites of indicators necessary to detect the broadest spectrum of perturbations in watersheds.

CHAPTER 3
HYDROLOGIC INDICES OF WATERSHED SCALE MILITARY IMPACTS IN FORT BENNING, GA

Introduction

A goal of stream flow characterization and classification is to develop hydrologic indices that account for characteristics of streamflow variability that are biologically relevant (Olden and Poff, 2003). Broadly, indices are attributes that respond in a known way to a disturbance i.e., they relate key ecological responses to human activities. Index identification is based on the goals and objectives set for a particular ecosystem or region. A good index should be sensitive to stressors, biologically and socially relevant, broadly applicable to many stressors and sites, diagnostic of the particular stressor causing the problem, measurable, interpretable, and not redundant with other measured indices (Cairns et al., 1993).

Ecologically relevant hydrologic indices developed in the past not only characterize particular regions, but also quantify flow characteristics that are sensitive to various forms of human perturbations. For example, early studies on hydrological indicators focused on variation of mean daily flow to study the pattern of fish in Illinois and Missouri (Horwitz, 1978). In Great Britain, Moss et al. (1987) used average flow conditions to predict macro-invertibrate fauna of unpolluted streams and Townsend et al. (1987) examined persistence of community structure for benthic invertebrates. In arid regions of southwest United States, Minckley and Meffe (1987) studied effects of short-term flood frequency in stream fish communities. Poff and Ward (1989) used long-term

discharge records (17-81 years) of 78 streams from across the continental United States to develop a general quantitative characterization of streamflow variability. Similarly, Jowett and Duncan (1990) studied skewness in flows and peak discharges in relation to in-stream habitat and biota in New Zealand.

More recent investigations have begun to focus on examining suites of hydrologic indices that are ecologically relevant to quantify hydrologic regimes. These studies report numerous such indices. For example, Poff and Allan (1995) studied stream fish assemblage for 34 sites in Wisconsin and Minnesota in conjunction with long-term stream flow variability and predictability as well as frequency and predictability of high and low flow extremes. In the process of deriving ecologically relevant hydrologic indices, Clausen and Biggs (1997) identified thirty-four hydrological variables from daily flow records at eighty-three New Zealand sites. The authors related these variables to benthic biota including periphyton and invertibrate species richness and diversity. Wood et al. (2000) reported the importance of hydrological conditions in explaining the ecological role when the authors studied the changes in macro-invertibrate community in response to flow variations in the Little Stour River in the United Kingdom. Pettit et al. (2001) described a method for assessing seasonality and variability of natural flows and their influence on riparian vegetation in two contrasting river systems in western Australia.

To isolate core flow variables for ecological studies, it is important to know not just the ecological relevance of the variables, but also the interrelationships among the variables in order to avoid redundancy in the analyses (e.g., Clausen and Biggs, 2000; Olden and Poff, 2003). Hydrologic indices have been criticized for being overly

simplified and lacking adequate biological relevance. Stream ecologists are now facing difficulty in choosing appropriate and relevant ones from the available suit of indices. For example, the Indicators of Hydrologic Alteration (Richter et al., 1996) approach is commonly used for characterizing human modification of flow regimes, yet it contains 64 statistics (32 measures of central tendency and 32 measures of dispersion), many of which are inter-correlated (Olden and Poff, 2003).

To date, characterization of streams or regions through determination and development of ecologically relevant hydrologic indices are based on long-term stream flow data. However, given the multitude of methods to characterize stream flow, past studies overlooked the value of storm flow data for the development of such ecologically relevant hydrologic indices. Flow characteristics are especially important where changes in land use are anticipated and where alterations to the flow regime need to be assessed.

The primary objective of the study is to identify hydrologic indices that characterize the impact of military land management on watersheds in Fort Benning, Georgia. Towards this end, this study investigates both storm and annual hydrographs. it is hypothesized that, in addition to annual-based indices, storm-based hydrologic indices are indicative of alteration in stream ecology. Here, a suite of event based hydrologic indices is proposed. Storm-based and the annual-based indices are calculated and used to compare and contrast impacted watersheds with a reference watershed. Additionally, specific military land management practices are used to predict storm-based indices.

Flow Regimes and Hydrologic Indices

To assess hydrologic alterations within an ecosystem, Richter et al. (1996) developed a method to compute representative, multi-parameter suite of hydrologic characteristics that are of ecological relevance, commonly known as Indicators of

Hydrologic Alteration (IHA). Olden and Poff (2003) comprehensively reviewed currently available hydrologic indices for characterizing streamflow regimes and recommended non-redundant indices for various stream types that may differ in major aspects of ecological organization. Poff (1996) provides a comprehensive catalog of the stream types for small to mid-size relatively undisturbed streams, classified according to variation in ecologically relevant hydrological characteristics, in continental United States. The assessment of IHA as well as other studies (e.g., Poff and Allan, 1995; Clausen and Biggs, 1997; Wood et al., 2000; Pettit et al., 2001; Olden and Poff, 2003) to identify hydrologic indices is based on long-term flow data.

An alternative approach to identify ecologically relevant hydrologic indices is to conduct an assessment based on the storm hydrograph. This approach is useful when long-term data for a particular stream or region are not available, when significant data gaps exist, or coincident records are not available. Storm hydrographs are traditionally described by characteristics including peak flow, total volume of direct runoff, and duration as shown in Figure 3-1. The time characteristics of the hydrograph and its relationship to the precipitation event are presented in Table 3-1. Towards the goal of characterizing the spatial variations of hydrologic conditions using storm-based indices that are ecologically relevant as well as sensitive to human influences, the ecological function of hydrologic characteristics that are relevant to storm-based hydrologic indices are considered.

Examination of the storm hydrographs reveals numerous potential indices. A set of 18 storm-based ecologically relevant hydrologic indices that characterize variation in water condition in individual watershed is proposed. Included proposed indices are

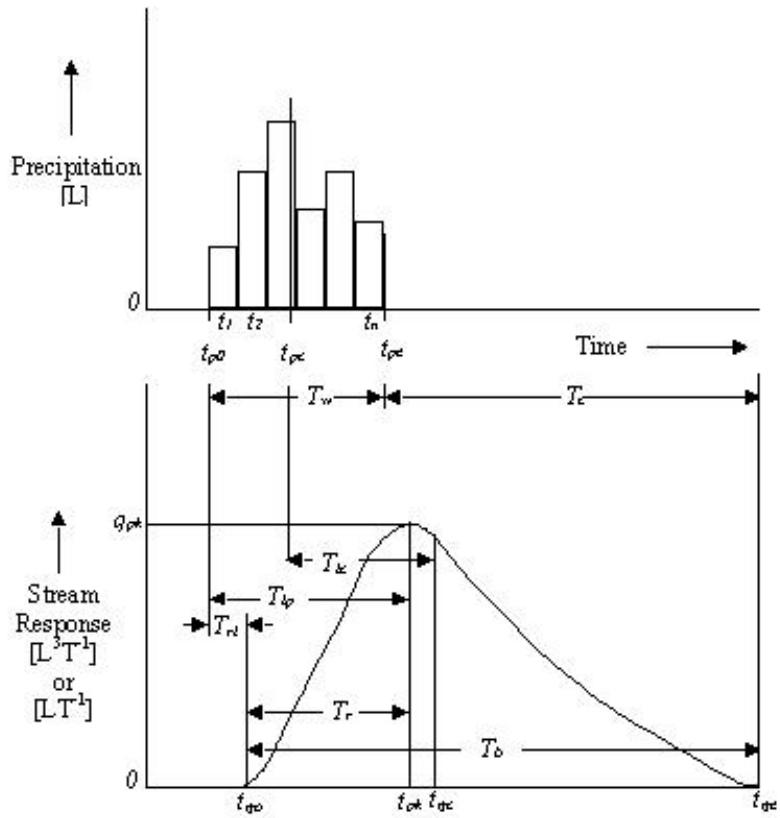


Figure 3-1. Terms used to describe hyetograph and response hydrograph. Refer to Table 3-1 for the definitions of the terms.

Table 3-1. Definitions of terms used to describe hyetograph and response hydrograph.

Time instants	Time durations
t_{p0} = beginning of precipitation	$T_w = t_{pe} - t_{p0}$ = duration of water input
t_{pc} = centroid of precipitation	$T_{rl} = t_{q0} - t_{p0}$ = response lag
t_{pe} = end of precipitation	$T_r = t_{pk} - t_{q0}$ = time of rise
t_{q0} = beginning of hydrograph rise	$T_{lp} = t_{pk} - t_{p0}$ = lag-to-peak
t_{pk} = time of peak discharge	$T_{lc} = t_{qc} - t_{pc}$ = centroid lag
t_{qc} = centroid of response hydrograph	$T_b = t_{qe} - t_{q0}$ = time base
t_{qe} = end of response hydrograph	$T_c = t_{qe} - t_{pe}$ = time of concentration

response factor (ratio of direct runoff depth to precipitation depth), baseflow index (ratio of baseflow volume to total volume during an event), peak discharge, dimensionless numbers related to hydrograph response lag (T_{rl}), time of rise (T_r), and time base (T_b), bankfull discharge and the slopes of rising and falling limb of the hydrographs. A flow

with 1.67-year return interval is often recognized as bankfull discharge (Poff, 1996). The eighteen ecologically relevant hydrologic indices were divided into four components of hydrologic regimes, magnitude, frequency, duration, and rate of change, to statistically characterize hydrologic variation.

Effects of Flow Magnitude on Stream Ecology

This group includes 6 parameters (mean and coefficient of variation) related to response factor, baseflow index, and the peak discharge (q_{pk}). The magnitude of the water condition at any given time is a measure of the availability or suitability of habitat and defines such habitat attributes as wetted area or habitat volume, or the position of a water table relative to wetland or riparian plant rooting zones (Richter et al., 1996). High flows maintain ecosystem productivity and diversity. For example, high flows remove and transport fine sediments that would otherwise fill the interstitial spaces in productive gravel habitats (Beschta and Jackson, 1979). Floods import woody debris into the channel (Keller and Swanson, 1979), where it creates new, high-quality habitat (Moore and Gregory, 1988; Wallace and Benke, 1984). Floodplains and wetlands provide important nursery grounds for fish and export organic matter and organisms back into the main channel (Junk et al., 1989; Welcomme, 1992). The scouring of floodplain soils revives habitat for plant species that germinate only on barren, wetted surfaces that are free of competition (Scott et al., 1996) or that require access to shallow water tables (Stromberg et al., 1997).

Effects of Flow Duration on Stream Ecology

The 6 parameters in this group measure the duration of all the events considered in the analysis. The parameters included in this group are the mean, and coefficient of variation of dimensionless indices T_{rl}/T_{lc}, T_r/T_{lc}, and T_b/T_{lc}, where T_{rl}, T_r, T_b, and T_{lc}

corresponding to the response lag, time of rise, time base, and centroid lag of a storm hydrograph, respectively. The duration is the period of time associated with a specific storm condition that determines the differences in tolerance to prolonged flooding in riparian plants (Chapman et al., 1982). Changes in duration of inundation can alter the abundance of plant cover types (Auble et al., 1994). For example, increased duration of inundation has contributed to the conversion of grassland to forest along a regulated Australian River (Bren, 1992). For aquatic invertebrates and fishes prolonged flows of particular levels can also be damaging (Closs and Lake, 1996). Whether a particular life-cycle phase can be completed or the degree to which stressful effects such as inundation of a flood plain can accumulate may be assessed from the duration of time over which a specific water condition exists (Richter et al., 1996).

Effects of Flow Frequency on Stream Ecology

The two parameters in this group measure the number of occurrences of the magnitude of the stream flow condition with respect to bankfull discharge. These numbers of occurrences of bankfull discharges are reported as percentages of the total events under analysis. Measures of exceedance of bankfull conditions have greater ecological importance. These flows regulate numerous ecological processes within riparian as well as flood plain areas. Frequent, moderately high flows effectively transport sediment through the channel (Leopold et al., 1964). Movement of sediment and organic resources such as detritus and attached algae revive the biological community and allow many species with fast life cycles and good colonizing ability to re-establish (Fisher, 1983). Consequently, the composition and relative abundance of species that are present in a stream often reflect the frequency and intensity of high flows (Meffe and Minckley, 1987; Schlosser, 1985).

Effects of Flow Rate of Change on Stream Ecology

This group includes 4 parameters, mean rate of change in peak discharge in rising limb and falling limb of the storm hydrograph and their variabilities. Flow conditions' rate of change, or flashiness, refers to the rate at which flow changes from one magnitude to another, can influence species persistence and coexistence. At the extremes, flashy streams have rapid rates of change, whereas stable streams have slow rates of change (Poff et al., 1997). The rate of change in water conditions may be tied to the stranding of certain organisms along the water's edge or in ponded depressions, or the ability of plant roots to maintain contact with phreatic water supplies (Richter et al., 1996). Non-native fishes generally lack the behavioral adaptations to avoid being displaced downstream by sudden floods (Minckley and Deacon, 1991). Meffe (1984) documented that a native fish, the Gila topminnow (*Poeciliopsis occidentalis*), was locally extirpated by the introduced predatory mosquitofish (*Gambusia affinis*) in locations where natural flash floods were regulated by upstream dams, but the native species persisted in naturally flashy streams.

Data Collection

Watershed Characteristics

The Fort Benning study watersheds (Figure 3-2), Bonham-1 and Bonham-2, Bonham, Little Pine Knot, and Sally Branch (named for the creek/stream that drains the watershed), represent a range of the region's soils, topography, land use, and vegetation communities (Table 3-2). In addition to standard watershed characteristics, the military training land area and a dimensionless military disturbance index (DIN) were calculated for each watershed. The military land use include fire ranges, maneuver training areas for light, amphibious, and heavy training, air field, artillery and mortar firing points,

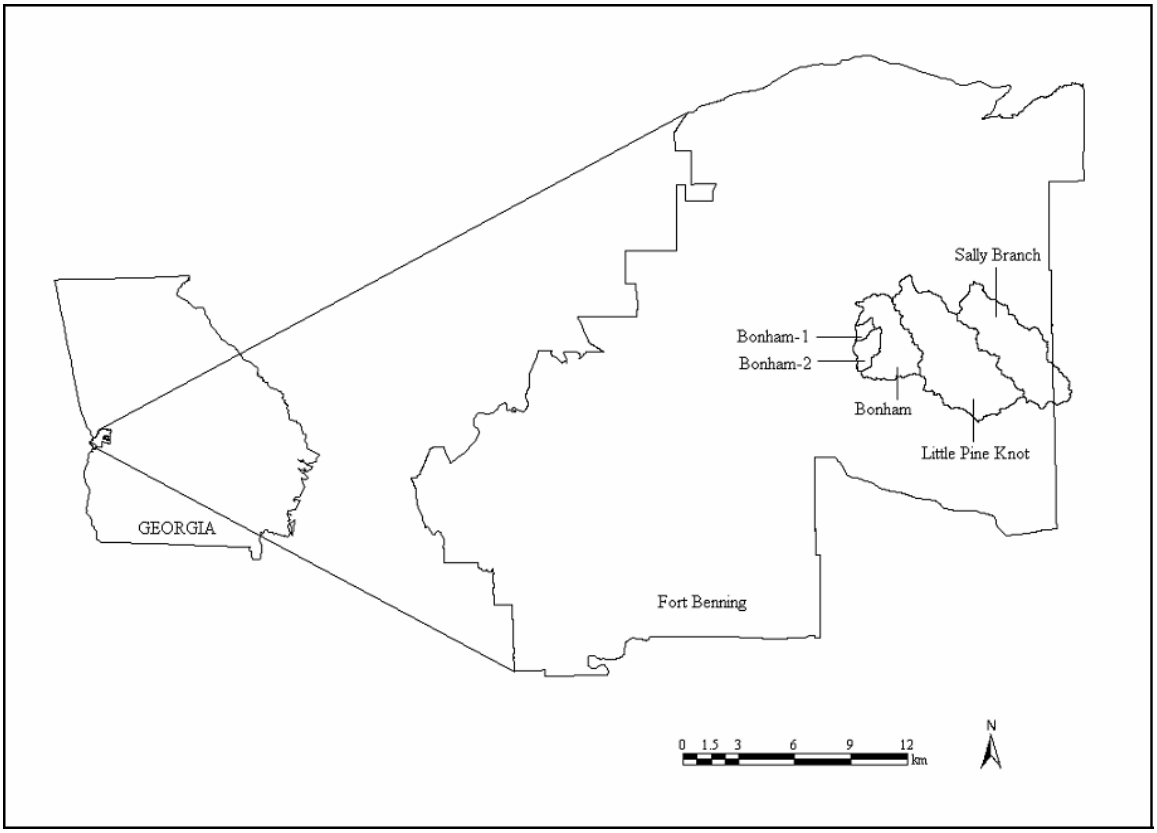


Figure 3-2. Study watersheds, Bonham-1, Bonham-2, Bonham, Little Pine Knot, Randall, Sally Branch, and Oswichee, in the Fort Benning military installation.

airborne drop zones, specialized non-live-fire training assets, and duded impact areas. A DIN is the sum of area of bare ground on slopes greater than 3 degrees and on roads as a proportion of the total watershed area (Maloney et al., in press). Additional details regarding the study area are found in Bhat et al., Ecological indicators in forested watersheds in Fort Benning, GA: Relationship between land use and stream water quality, submitted to Ecological Indicators; hereinafter referred to as Bhat et al., submitted manuscript).

According to variations in ecologically relevant hydrological characteristics that are based on flow variability and predictability, and low- and high-flow extremes, the

Table 3-2. Physical characteristics of study watersheds. Acronyms BON-1, BON-2, BON, LPK, and SAL represent the streams (or watersheds) Bonham-1, Bonham-2, Bonham, Little Pine Knot, and Sally Branch, respectively.

Physical Characteristics	BON-1	BON-2	BON	LPK	SAL
Topography					
Area, km^2	0.76	2.21	12.73	18.01	25.31
Average Elevation, m	121.8	133.5	125.5	146.3	136.8
Average Slope, degree	5.46	4.89	5.04	5.32	5.42
Vegetation					
Pine, %	28	30	40	41	48
Deciduous, %	27	6	8	2	12
Mixed, %	39	50	22	34	15
Wetland, %	6	8	9	17	10
Military Training Land, %	0	6	5	2	2
NDVI	0.36	0.30	0.32	0.34	0.36
Soil					
Sandy loam, %	78	69	69	72	49
Loamy sand, %	9	9	31	28	51
Road					
Road Length, km	3.6	11.4	51.6	56.6	97.6
Road Density, km/km^2	4.8	5.1	4.1	3.1	3.8
Stream					
Stream Length, km	2.6	3.9	29.1	43.3	65.2
Stream Density, km/km^2	3.4	1.7	2.3	2.4	2.6
Stream Order	2	2	4	4	4
Other					
No. of Roads Crossing Streams	1	2	13	11	21
Bare Land, %	1	11	11	4	7
Disturbance Index, %	11	21	19	11	15

streams in the study watersheds are classified as ‘perennial runoff’ (Poff, 1996). As the present study focuses on the impacts of military training within an ecosystem as well as the effects of these impacts on the stream biota, a reference watershed was identified to contrast with watersheds impacted by military training activities. Approximately 94% of the area of Bonham-1 watershed is covered by forest. Mechanized military activities are

not conducted in this watershed as compared to 2-6% of the total area of other watersheds used for the same purpose. Also, the watershed has a small percentage of bare land, limited roads, and only one road crossing the stream. Hence, Bonham-1 was used as a reference watershed for this study.

Precipitation and Streamflow Data

Streamflow and precipitation were measured from January 2000 to December 2003. Precipitation was measured by twelve tipping bucket rain gauges distributed throughout the study area. Watershed precipitation was determined by areal weighting using the Thiessen polygon method. Daily discharge values for Bonham-1 and Bonham-2 were calculated from ten-minute continuous stage records using rating curves. Stream stage and velocity were measured half-hourly for Bonham, Little Pine Knot, and Sally Branch. These data were used to calculate daily discharges using the area-velocity method.

Methods

Annual-Based Hydrologic Indices

The annual-based approach uses multi-year streamflow records to define a series of ecologically relevant hydrologic indices. These indices may be used to characterize intra-annual variation in water conditions, analyze temporal variations, and compare impacts of alteration among watersheds. For this assessment, nonredundant yet biologically significant hydrologic indices are adapted as per the recommendations of Olden and Poff (2003). Table 3-3 lists these indices for the perennial runoff type of

Table3-3. Summary of the recommended hydrologic indices for 'perennial runoff' type of streams (after Olden and Poff, 2003) used in the Annual-Based Hydrologic Indices. BON-1, and BON-2 represent Bonham-1 and Bonham-2, respectively. BON-1 is reference (REF) and BON-2 is (IMP) watershed. To maintain the consistency to the past studies, the symbols presenting the hydrologic indices in this table are kept as the same as those used in Olden and Poff (2003). Data types A, M, and D stand for annual, monthly, and daily discharge data, respectively. For definitions and method of calculation of the indices, refer to Appendix A.

Flow components and name of the hydrologic index	Symbol	Units	BON-1 (REF)	BON-2 (IMP)	Data type
Magnitude of flow events					
Average flow conditions					
Variability in December flow	M_A26	unitless	0.70	0.47	M
Mean annual runoff	M_A41	$m^3/s/km^2$	0.00397	0.00504	A
Spreads in daily flows	M_A10	$1/m^3/s$	81.4	43.5	D
Low flow conditions					
Baseflow index	M_L17	unitless	0.12	0.57	A
Mean of annual minimum flows	M_L14	unitless	0.17	0.58	A
Median of annual minimum flows	M_L16	unitless	0.17	0.79	A
High flow conditions					
High flow volume	M_H23	days/no. of years	9.2	0	A
Mean maximum flow in May	M_H8	m^3/s	0.016	0.017	M
Median of annual maximum flows	M_H14	unitless	63.4	4.9	A
Frequency of flow events					
Low flow conditions					
Frequency of low flow spells	F_L3	$year^{-1}$	0.75	0	A
Variability in low flood pulse count	F_L2	unitless	0.31	0.26	A
High flow conditions					
High flood pulse count	F_H4	$year^{-1}$	6	0	A
Three times median flow frequency	F_H6	$year^{-1}$	10.25	2.50	A
Seven times median flow frequency	F_H7	$year^{-1}$	6	0	A

Table 3-3. Summary of the recommended hydrologic indices for 'perennial runoff' type of streams (after Olden and Poff, 2003) used in the Annual-Based Hydrologic Indices (Continued).

Flow components and name of the hydrologic index	Symbol	Units	BON-1 (REF)	BON-2 (IMP)	Data type
Duration of flow events					
Low flow conditions					
Variability in annual minima of 90-day means of daily discharge	D_L10	unitless	0.78	0.32	D/M/A
Variability in low flow pulse duration	D_L17	unitless	0.09	0.07	A
Variability in annual minima of 1-day means of daily discharge	D_L6	unitless	0.60	0.28	D/M/A
High flow conditions					
Means of 30-day maxima of daily discharge	D_H13	unitless	5.59	1.84	D/M/A
Variability in high flow pulse duration	D_H16	unitless	0.006	0.005	A
Flood free days	D_H24	days	56.0	61.5	A
Timing of flow events					
Average flow conditions					
Constancy	T_A1	unitless	0.28	0.44	D
Seasonal predictability of flooding	T_A3	unitless	0.70	0.84	M
High flow conditions					
Seasonal predictability of non-flooding	T_H3	unitless	0.40	0.41	M
Rate of change in flow events					
Average flow conditions					
Variability in reversals (Positive)	R_A9	unitless	0.04	0.07	D
Variability in reversals (Negative)	R_A9	unitless	0.06	0.12	
Change of flow (Decreasing)	R_A7	m^3/s	0.1136	0.0324	D
Change of flow (Increasing)	R_A6	m^3/s	0.1119	0.0496	D

streams found in the study region. The definitions and the methods used to calculate these indices are listed in Appendix A. In this study, the annual-based indices are used to compare and contrast the Bonham-1 and Bonham-2 watersheds.

Storm-Based Hydrologic Indices

The storm-based approach uses the storm hydrographs to determine the indices. Hydrograph separation was used to identify distinct storm events. 44-100 storm events from 2001 to 2003 were used to calculate the response factor, baseflow index, dimensionless indices (T_{rl}/T_{lc}, T_r/T_{lc}, and T_b/T_{lc}), watershed area (A) scaled peak discharge (q_{pk}/A), bankfull discharges, and the rate of change of peak discharges in rising and falling limbs for five watersheds, where Bonham-1 is the reference watershed (Table 3-4). Appendix B lists the indices, definitions, and the methods of calculation.

Statistical Analyses

The ANOVA and the Tukey's multiple comparison tests were performed to determine the differences between reference watershed and the impacted watersheds' mean values of indices. The percent of forest extent, military training land, road density, the number of roads crossing streams, bare land fraction, and disturbance index were considered to assess the effects of disturbance on hydrology and the potential impact on stream ecology. Pearson's correlation coefficients were calculated to examine the strength and significance of the relationships between a watershed physical characteristic and storm-based hydrologic index. Stepwise multiple linear regressions were performed to identify relationships between an index and management related watershed characteristics. Only variables having p-values less than or equal to 0.05 were retained in the regression models.

Results

Watershed Disturbance Characteristics

The watersheds' physical characteristics are summarized in Table 3-2. Most of the watersheds are highly vegetated (70% or more) with the majority characterized by pine

Table 3-4. Summary of hydrologic indices used in the Storm-Based Hydrologic Indices. BON-1, BON-2, BON, LPK, and SAL represent the streams (or watersheds) Bonham-1, Bonham-2, Bonham, Little Pine Knot, and Sally Branch, respectively. BON-1 is reference (REF) and other watersheds are impacted (IMP). * represents the mean values of the indices of the impacted watersheds that are different at significance level of 0.05 from the mean values of reference watershed as confirmed by Tukey's multiple comparison test. For definitions and method of calculation of the indices, refer to Appendix A.

Flow components and name			REF	IMP			
of the hydrologic index	Symbol	Units	BON-1	BON-2	BON	LPK	SAL
Magnitude							
Mean response factor	M_MRF	unitless	0.030	0.010	0.060*	0.001*	0.020
Variability in response factor	M_VRF	unitless	1.09	1.45	1.18	0.89	1.30
Mean baseflow index	M_MBF	unitless	0.73	0.86*	0.95*	0.90*	0.95*
Variability in baseflow index	M_VBF	unitless	0.19	0.11	0.04	0.07	0.04
Mean peak discharge	M_MPD	$m^3/s/km^2$	0.077	0.025*	0.030*	0.001*	0.014*
Variability in peak discharge	M_VPD	unitless	2.0	2.3	1.2	1.0	1.4
Frequency							
Bankfull discharge	F_1FD	%	90	47	2	29	6
Two times bankfull discharge	F_2FD	%	82	32	0	10	0
Duration							
Duration of time base	D_MTB	unitless	1.9	1.7	1.2*	1.3*	1.2*
Variability in time base	D_VTB	unitless	0.4	0.6	0.6	0.5	0.5
Duration of response lag	D_MRL	unitless	0.5	0.6	1.1*	0.8	1.3*
Variability in response lag	D_VRL	unitless	0.8	1.4	0.6	0.5	0.4
Duration of time of rise	D_MTR	unitless	0.5	0.5	0.5	0.5	0.5
Variability in time of rise	D_VTR	unitless	0.5	0.8	1.0	0.9	0.7
Rate of change							
Mean slope of rising limb	R_PPD	$m^3/s/hr$	0.20	0.06*	0.10	0.01*	0.04*
Variability in rising slopes	$R_{PV}PD$	unitless	2.2	2.4	1.4	1.4	1.6
Slope of falling limb	R_NPD	$m^3/s/hr$	0.050	0.020	0.070	0.001*	0.020
Variability in falling slopes	$R_{NV}PD$	unitless	2.4	2.0	1.5	1.2	1.2

and mixed pine and hardwoods. Deciduous forest typically covers only a small percentage of these watersheds. However, Bonham-1 consists of 27% of deciduous forest. The study watersheds range from less than 1 to 25 km^2. Average elevations and maximum slopes are relatively constant. Sandy and loamy soils are common in most of the study watersheds. The military training land (0 to 6%) is relatively small, but varies among watersheds. Total bare lands in these watersheds comprise 11 to 21% of the watershed area, of which 1 to 8% of the total area is unpaved roads and trails. This extent and variability of military training and bare land are typical of the entire Fort Benning installation. While road density is relatively comparable among watersheds, the number of roads crossing streams varies from 1 to 21. Military training land, bare land, and disturbance index are positively correlated at significance level of 0.05 or lower.

Annual-Based Hydrologic Indices

Table 3-3 summarizes annual-based analysis results for Bonham-1 (reference) and Bonham-2 (impacted) watersheds. The average flow conditions revealed that the mean annual flow (M_A41) is higher in the impacted watershed. However, the reference watershed has higher flow variability for both annual (M_A10) and December month (M_A26) periods. The average flow event timing is more constant (T_A1) and predictable (T_A3) in the impacted watershed.

The impacted watershed maintained a higher magnitude of minimum flows as depicted by the baseflow index (M_L17). This result is consistent across other low flow indices (M_L14 and M_L16) and the findings that the impacted watershed had no low flow spells (F_L3) and lower variability of low flow pulse counts (F_L2). Higher coefficient of variation in annual minima of 90-day (D_L10) and 1-day (D_L6) means of daily discharge for the reference watershed and similar variability in low flow pulse duration (D_L17)

were found for both watersheds. While low flow values vary more for the reference watershed, once the flow goes low, it stays low for same duration in both watersheds.

The reference watershed produces higher magnitude flow and thus maintains higher median flow (M_H14) during events. However, in the month of May, mean of the maximum monthly flows (M_H8) were similar for both the watersheds. During high flow conditions, the reference watershed crosses a threshold of seven times the median annual daily flow volume (M_H23). In the impacted watershed, these floods never occurred. The high flood pulse counts (F_H4) and the frequency of floods (F_H6 and F_H7) in the reference watershed are more than the impacted watershed. However, on a few occasions, the flow crossed a lower threshold of 3 times median frequency of flood (F_H6) in the impacted watershed. The 30-day floods (D_H13) went higher and stayed high (D_H16) in the reference watershed. Periods between floods (D_H24) are similar for both watersheds, that is, they have approximately two months of flood free days a year and of comparable predictability (T_H3).

Storm-Based Hydrologic Indices

The results of storm-based hydrologic assessment are summarized in Table 3-4. Analysis of variance tests indicated that the mean values of storm-based indices except the time of rise differ at the significance level of 0.05. For a number of indices, the reference watershed exhibits distinct behavior as compared to the impacted watersheds. Tukey's multiple comparison tests indicate that the indices other than time of rise and the rate of change in falling limb in the reference watershed were significantly different from the impacted watersheds. The reference watershed is characterized by a relatively low baseflow index (M_MBF) with significantly higher (M_MPD) and more variable (M_VPD) peak discharge. During events, 90% of the total events produced greater than bankfull

discharge (F_1FD) in the reference watershed indicating a highly connected system as compared to 2-47% in impacted watersheds.

Storm flows consistently lasted longer and responded faster to rain events in the reference watershed (D_MTB and D_VTB). Once the stream responded, the time of rise (M_MTR) was similar in all the watersheds. The reference watershed's combination of fast response and high peak discharge results in a rapidly increasing rising limb (R_PPD) as compared to impacted watersheds.

Relationship between Military Land Management and Storm-Based Hydrologic Indices

Correlation and regression analyses were performed to determine the relationship among the watershed physical characteristics and the storm-based hydrologic indices. Table 3-5 indicates that 7 key storm-based hydrologic indices are significantly related to military land management. Increased military training land, bare land, and the disturbance index will increase the time of rise as well as the variability in the time base. Increasing the road density increases the variability in the time base and the rate of change of rising limb. Increasing the number of roads crossing streams increases the storm response lag, but decreases the time base. Results also show that an increase in the number of roads crossing streams decreases the variability in the rate of change of falling limb. No effects on hydrologic indices were identified for forestry management practices.

Stepwise multiple correlations characterized the response of storm-based indices to military impacts (Table 3-6). The greatest impact of land management is found with statistically significant predictive models for indices of time base, response lag, and time

Table 3-5. Pearson correlation coefficients between watershed physical characteristics and event based hydrologic indices. Characteristics are acronymed as Forest (FOR), Military Training Land (MIL), Road Density (RDN), No. of Roads Crossing Streams (NRC), % Bare Land (PBL), and Disturbance Index (DIN). * and ** indicates significance at or below 0.05, and 0.01 probability levels, respectively.

Hydrologic Indices	FOR	MIL	RDN	NRC	PBL	DIN
M_MRF	0.25	0.15	0.15	0.13	0.27	0.31
M_VRF	0.40	0.59	0.71	-0.05	0.67	0.81
M_MBF	-0.62	0.44	-0.56	0.83	0.68	0.41
M_VBF	0.63	-0.39	0.61	-0.85	-0.63	-0.35
M_MPD	0.83	-0.37	0.64	-0.62	-0.47	-0.19
M_VPD	0.66	0.21	0.96**	-0.73	0.04	0.34
F_1FD	0.57	-0.40	0.57	-0.85	-0.65	-0.39
F_2FD	0.69	-0.43	0.62	-0.82	-0.64	-0.36
D_MTB	0.63	-0.17	0.77	-0.91*	-0.42	-0.12
D_VTB	-0.50	0.98**	0.14	0.08	0.96**	0.90*
D_MRL	-0.23	0.06	-0.54	0.97**	0.41	0.18
D_VRL	0.21	0.59	0.84	-0.78	0.33	0.57
D_MTR	-0.39	0.99**	0.32	-0.17	0.89*	0.90*
D_VTR	-0.86	0.67	-0.45	0.35	0.68	0.47
R_PPD	0.79	-0.34	0.59	-0.55	-0.42	-0.16
R_{PV}PD	0.63	0.13	0.91*	-0.78	-0.06	0.24
R_NPD	0.42	0.10	0.35	-0.13	0.14	0.26
R_{NV}PD	0.70	-0.07	0.85	-0.88*	-0.29	0.03

of rise. Military training land, road density, and the number of road crossing streams were the three management variables that impacted storm responses.

Table 3-6. Stepwise multiple regression models for event based hydrologic indices. Military Training Land (MIL), Road Density (RDN), and No. of Roads Crossing Streams (NRC), are the independent variables retained in the regression analyses. * and ** indicates significance at or below 0.05, and 0.01 probability levels, respectively.

Hydrologic indices	Independent variables retained and regression equations	R^2 (adj)
M_MBF	0.729 + 1.91 MIL + 0.00968 NRC	0.94*
M_VPD	- 1.21 + 0.666 RDN	0.88**
F_1FD	0.929 - 6.97 MIL - 0.039 NRC	0.94*
D_MTB	1.8 - 0.0352 NRC	0.77*
D_VTB	0.418 + 3.05 MIL	0.95**
D_MRL	0.483 + 0.0393 NRC	0.92**
D_MTR	0.454 + 0.837 MIL	0.97**
R_{NV}PD	2.20 – 0.0563 NRC	0.70*

Discussion

Results representing annual-based average flow conditions showed higher magnitudes of M_A26, M_A41, and M_A10 in the reference watershed as compared to the impacted one. In addition, these flows are less constant and less predictable over the years as compared to the impacted watershed. Aquatic communities can show distinct differences to changes in velocity and reduction in bed gradient, and associated fining of bed sediments (Clausen and Biggs, 2000). Periphyton and benthic invertebrates are particularly sensitive to different velocities and bed sediment size/stability (Minshall, 1984; Biggs, 1996). Clausen and Biggs (1997) found that invertebrate species richness and periphyton biomass changed based on flow. These changes are likely related to riparian vegetation on the amount of leaf litter input (Vannote et al. 1980; Davies-Colley and Quinn, 1998).

The higher magnitude of the annual-based low flow indices (M_L17, M_L14, and M_L16) in the impacted watershed is attributed to higher groundwater input as compared to the reference watershed. This increased input likely reflects the reduction of interception characteristic of the military land uses (Bryant et al., in press). The relative magnitude of low flows is likely to have important influences on biota through the intensity of habitat destruction associated with drying during low flows. In this study, the lower magnitudes of low flows as depicted by lower baseflow index (M_L17) and smaller low flow pulses (F_L3) suggest that the long periods of low flow condition are more likely in the reference watershed. Higher variability in the duration during low flow condition (D_L10, D_L17, and D_L6) in the reference watershed supports likelihood of such long periods of low flow conditions. Long periods of low flow conditions and higher variability in these conditions may provide selective pressure for specific life history characteristics such as invertibrate aestivation and egg diapause, and physiological tolerance to low dissolved oxygen (Williams and Hynes, 1977).

The higher magnitude of the annual-based high flow indices (M_H23 and M_H14) and peak discharges (M_MPD and M_VPD) in the reference watershed as compared to the impacted watershed suggests the likelihood of habitat regeneration associated with sediment transport and floodplain inundation during high flows. For high flow events, the degree of riverbed communities' disturbance is strongly related to degree of bed movement (e.g., Biggs et al., 1999). Dissolved inorganic nitrogen and phosphorus concentrations in rivers are strongly negatively correlated with specific yield and high flow magnitude among watersheds, and among years within watersheds (e.g., Biggs and Close, 1989; Close and Davies-Colley, 1990; Grimm and Fisher, 1992). This reflects the

degree of flushing of the nutrients mineralized through organic matter breakdown in the soil profile and leachate from the underlying substrata.

Floods or high flow conditions are also important in influencing community structure. The results based on the annual flow in this study clearly show that the reference watershed produced more frequent high flows (F_H4, F_H6, and F_H7). Similarly, the results from storm-based analysis show that the frequency of discharges equaling or exceeding bankfull (F_1FD and F_2FD) is higher in the reference watershed. Floods are widely viewed as reset mechanisms (Resh et al., 1988), and flood-related mortality to lotic organisms can result either directly from scouring, crushing, or downstream export of individuals (Minckley and Maffe, 1987) or indirectly from food resources loss (Hanson and Waters, 1974). As flood frequency increases, some invertebrates in the reference watershed actively migrate either into the substratum or to quieter backwaters to avoid sudden floods. Floods have been shown to regulate community structure by facilitating local coexistence between asymmetrically competitive algal species (Power and Stewart, 1987) and invertibrate species (Hemphill and Cooper, 1983) and between an exotic fish predator and its native, relatively flood-resistant prey (Meffe, 1984). The result in this study showed the consistency of storm-based F_1FD and F_2FD with annual-based F_H4, F_H6, and F_H7 indices suggesting F_1FD and F_2FD may be used as alternative indices.

As indicated by the higher annual-based mean duration of high flow condition (D_H13) and the duration of storms (D_MTB) in the reference as compared to the impacted watershed, it is apparent that water resides in the reference watershed for a longer period of time during high flow condition. However, the D_MRL is smaller for the reference

watershed suggesting a quick rise in hydrograph. This can be attributed to a better connectivity of riparian areas to the stream. This connectivity is important as the nutrient concentrations can strongly control autotrophic production during inter-flood periods (e.g., Biggs, 2000). Also, individual high flow events greatly reduce the biomass and change the species composition of periphyton (e.g., Biggs and Stokseth, 1996), and invertebrates (e.g., Cobb et al., 1992).

Predictive relationships were identified for storm-based hydrologic indices based on watershed scale military management and suggest that the key variables related to hydrologic alteration are military training land, road density, and the number of roads crossing streams. Military training within a watershed can modify annual and storm generated runoff due to changes in drainage, vegetation, and soils. The impact is due to troop maneuvers and large, tracked and wheeled vehicles that traverse thousands of hectares in a single training exercise (Quist et al., 2003). These activities' impacts, ranging from minor soil compaction and lodging of standing vegetation to severe compaction and complete loss of vegetation cover in areas with concentrated training use (Wilson, 1988; Milchunas et al., 1999), are evident in the stream hydrographs and suggest impacts to stream ecosystems. Disruption in soil density and water content (Helvey and Kochenderfer, 1990), addition of sediment, nutrients, and contaminants in aquatic ecosystems (Gjessing et al., 1984), and impairment of natural habitat development and woody debris dynamics in forested floodplain streams (Piegay and Landon, 1997) are potential impacts on terrestrial and aquatic ecosystems. In a recent study in the same watersheds in Fort Benning, Houser et al. (2004) found that gross primary productivity, respiration, and the benthic organic matter were low in the stream

corresponding to the impacted watershed (higher disturbance index) as compared to the reference one (lower disturbance index). Road crossings commonly found in military training areas may act as barriers to the movement of fishes and other aquatic animals (Furniss et al., 1991). Roads also directly change the hydrology by intercepting shallow groundwater flow paths, diverting the water along the roadway and routing it efficiently to streams at crossings (Wemple et al, 1996). This can cause or contribute to changes in timing and routing of runoff (Jones and Grant, 1996).

Conclusion and Recommendations

In the present study, a reference watershed was compared to impacted watersheds using annual and storm-based hydrologic indices within the Fort Benning military installation. The results suggest a subset of the hydrologic indices recommended by Olden and Poff (2003) are necessary for the Fort Benning streams. It is recommended that mean annual runoff and its spread, baseflow index, high flow volume, frequency of low flow spells, high flood pulse count, variability in annual minima of 90-day means of daily discharges, and constancy be used in stream ecology studies in the Fort Benning streams to identify disturbances relative to a reference watershed at a watershed scale.

The indices based on storm data may be used to augment the annual indices or as surrogate to those indices. Storm-based magnitude and variability in peak discharge, baseflow index, and the bankfull discharge were consistent with the results from annual-based analysis. With respect to the potential influence of the frequency and duration aspects of the flow regimes on the stream ecology, duration and variability of time base, and duration of response lag are identified as critical hydrologic indices. The military management practices, military training land, road density, and the number of road crossing streams, were found to significantly affect these indices.

CHAPTER 4
PREDICTION OF NITROGEN LEACHING FROM FRESHLY FALLEN LEAVES: APPLICATION OF RIPARIAN ECOSYSTEM MANAGEMENT MODEL (REMM)

Introduction

Forest canopy litterfall initiates the major pathway for recycling of nutrients from plant to soil (Bubb et al., 1998). Nutrients are returned to the soil via litterfall and throughfall fluxes in forested ecosystem (Ukonmanaho and Starr, 2001). Litterfall, comprised predominantly of leaf litter, is usually the most important nutrient source (Herbohn and Congdon, 1998). Litter quantity, litter decomposition and nutrient release patterns are factors for understanding nutrient cycling in forest ecosystems (Rogers, 2002). Litterfall is primarily caused by natural senescence of leaves. Forest litterfall quantity and composition varies among tree species, stand age and development, and reflects environmental conditions, particularly water and nutrient availability (Binkley, 1986; Polglase and Attiwill, 1992). The amount and distribution of litterfall through time are also affected by season, rainfall amount and distribution, and wind speed (Crockford and Richardson, 1998).

Fluxes of dissolved organic matter are an important vector for the movement of nutrients both within and between ecosystems (Cleveland et al., 2004). In many forest ecosystems, more than half of the nitrogen is soil solution is in organic form (Qualls et al., 1991). Hydrology plays an important role in both production and mobilization of dissolved organic matter in the forest floor (Park and Matzner, 2003). A substantial amount of potentially soluble organic matter exists in an adsorbed phase (Christ and

David, 1996). The amount of percolating water mobilizes these soluble organic matters (Tipping et al., 1999).

Past studies have shown that the dissolved organic matter flux increases when the percolating water passes through the forest floor (McDowell and Likens, 1988; Qualls et al., 1991; Currie et al., 1996; Michalzik et al., 2001). In a recent study in India, Singh et al. (1999) reported that the rapid mass loss of leaf litter occurred during the rainy season and the rate of litter mass loss was positively correlated with the cumulative rainfall. In an early study, Bocock et al. (1960) observed a rapid leaching from oak and ash leaf litter, and suggested that the weight loss during the first month was largely due to a physical loss by leaching. The primary sources of dissolved nutrients are considered to be leaching of substance from fresh litter and the product of decomposition of the plant litters (Qualls et al., 1991). Water-soluble nutrients are more easily leached from the leaf litter of deciduous species, including birch, than from coniferous species, including Scots pine (*Pinus sylvestris*) and Norway spruce (*Picea abies*) (Harris and Safford, 1996; Hongve, 1999).

Depending on the species, leaf litter may release from 5 to 30% of original dry weight as dissolved organic matter within 24 hours (Cummins, 1974). In a woodland stream ecosystem in Massachusetts, McDowell and Fisher (1976) found that leaves lose weight rapidly by abiotic leaching of soluble constituents for 1-3 days and decay more slowly thereafter as a result of microbial decomposition. Gosz et al. (1973) estimated 15% mass loss for yellow birch leaves (*Betula lutea*) due to leaching within first month on the forest floor. Heath et al. (1966) reported a 26% weight loss from birch leaves (*B.*

alba) that remained for two months on the forest floor. Nykvist (1961) found an 8% weight loss in *B. verrucosa* after 24 hours of leaching in distilled water.

Annual fluxes of nitrogen in litterfall are greater than other nutrients regardless of forest type but the amounts vary among forest types (Ukonmaanaho and Starr, 1999). Dissolved organic nitrogen is the major form of nitrogen in stream water draining from many mature forest watersheds, comprising about 95% of the total nitrogen (Qualls et al., 1991). Nitrogen content of a hardwood forest litterfall on the Hubbard Brook experimental forest comprised approximately 37% of the total nutrients (Gosz et al., 1972). In a low nutrient system at the Coweeta Hydrologic Laboratory in North Carolina, where carbon (C) to nitrogen (N) ratios in freshly fallen litter varied from 100 to 220, Qualls et al. (1991) reported that the output of dissolved organic nitrogen from the forest floor was 28% of the nitrogen input from the litterfall.

The Riparian Ecosystem Management Model (REMM) was developed by the U.S. Department of Agriculture's (USDA) Agricultural Research Service (ARS) to characterize the role of the riparian area on stream water quality (Lowrance et al., 2000; Altier et al., 2002). In REMM, the riparian areas' water quality functions for control of N, phosphorus (P), and sediment transport into surface waters are simulated and analyzed. This model has primarily been tested and applied on high-nutrient riparian buffers adjacent to agricultural fields in the past (e.g., Inamdar et al., 1999a; Inamdar et al., 1999b; Lowrance et al., 2000).

The complex dynamics related to the nutrients in a watershed is particularly of interest in low nutrient systems as the release of nutrients from plant litters prior to decomposition may be an important aspect of characterizing stream water quality. A

watershed scale study related to surface water quality was conducted in the Fort Benning military installation located in southwest Georgia. Experimental data from Fort Benning shows an increase in total Kjeldahl nitrogen concentrations following litterfall. The release of nitrogen is relatively rapid as compared to that released from decomposition of organic matter. It is hypothesized that after leaves fall, the first precipitation events can be expected to result immediate nitrogen leaching and a corresponding increase in nitrogen levels in the stream water. The goal of this study is to determine whether freshly fallen leaves in a riparian area are a significant source of nitrogen in low nutrient system. The specific objectives of this study are to quantify nitrogen in streamflow using REMM and verify the modeled results with the observed data.

Basic Concepts of REMM

Originally, REMM was developed to simulate buffer systems that meet specifications recommended by the USDA Forest Service and the USDA Natural Resource Conservation Service as a national standard (Natural Resource Conservation Service, 1995). Three zones parallel to the stream characterize the riparian system. Thus REMM simulates a three-zone buffer. The riparian area's soil is characterized in three layers. In these soil layers, vertical and lateral movement of water and associated dissolved nutrients takes place. The top two soil layers correspond with the soil horizons A and B, respectively. Depth of the root zone or a restricting soil horizon defines the third soil layer. A litter layer that interacts with surface runoff covers the soil layer 1. The model takes upland water output supplied by the user and calculates loadings of water, nutrients, sediment, and carbon based on actual area of the zones of a riparian area (Lowrance et al., 2000).

The model input data consist of the physical conditions of the riparian area, external loadings of water and nutrients to the riparian area, soil, vegetation, and soil information. Altier et al. (2002) documents a detailed description of the algorithms, equations, and parameters used in the model.

Hydrology component of REMM

REMM measures water stores and fluxes using a daily water balance. The water balance includes interception, evapotranspiration, infiltration, vertical drainage, surface runoff, subsurface lateral flow, upward flux from water table, and deep seepage. Interception losses occur in the vegetation canopy and litter layer. Canopy interception is an exponential function of the canopy storage capacity and the amount of daily rainfall, and is simulated using a modified form of the Thomas and Beasley (1986) equation. Precipitation falling through the canopy (throughfall) is subject to litter interception. Similar to canopy interception, litter interception is determined by the mass of the litter at any given time and the litter storage capacity.

Infiltration at the soil surface depends on the total depth of water. This depth of water is the sum of the throughfall depth and overland flow depth from the upslope zone or field. If the sum of incoming upland runoff and throughfall depth is less than the infiltration capacity of the riparian soil, runoff infiltrates at the rate of application. During high intensity storms, the runoff and throughfall rate may exceed the infiltration rate. Surface runoff from the riparian buffer occurs when total water depth exceeds the infiltration depth. In such conditions, model simulates the infiltration using a modified form of the Green-Ampt equation (Stone et al., 1994).

When soil moisture exceeds the field capacity, vertical drainage from a soil layer occurs. This vertical drainage also depends on the available water storage capacity of the

receiving layer. The rate of drainage is controlled by the lesser of the vertical hydraulic conductivities of the draining and receiving layer. Vertical unsaturated conductivity is simulated as a function of the soil moisture content using Campbell's equation (Campbell, 1974). When a water table builds up over the restricting soil layer, subsurface lateral movement occurs. Lateral water movement is simulated using Darcy's equation. Saturation excess overland flow occurs when soil profile is completely saturated.

Nutrient component of REMM

Simulation of nutrients in REMM is based on the Century Model (Parton et al., 1987). The C cycle is fundamental for simulation of all organic matter dynamics and many nutrient cycling processes in REMM. Release and immobilization of N is in proportion to transformations of C. Soil and litter C is divided into residue (woody debris, leaf litter, and roots) and humus (soil organic matter) pools. Each C pool has an associated mineralization rate, transformation efficiency, and a specific range of C to N ratios (Inamdar et al., 1999b). A first order rate equation with respect to C determines the mineralization of C from each pool. Litter from leaves, stems, branches, coarse roots, and fine roots are allocated into a readily decomposable (metabolic) or a resistant (structural) residue pool based on lignin to N ratio (Pastor and Post, 1986). Each pools decomposes at different rates.

REMM assumes a fixed fraction of C in fresh plant residue. Dissolved C from metabolic residue and active humus pools differ by litter and soil layer (Altier et al., 2002). The sum of C in throughfall and incoming runoff is mixed with C on the ground surface to determine dissolved and adsorbed concentrations. Dissolved C moves with surface runoff, subsurface lateral flow, and vertical drainage (Lowrance et al., 2000).

The amount of dissolved C carried out of the litter or soil layer is a function of the water volume and the dissolved C concentration.

Nitrogen is input to the riparian area through precipitation, surface and subsurface water flow, and adsorbed to incoming sediment carried by surface flow. Presence of nitrogen in different forms and the associated degree of physical and chemical stabilization influences its availability for microbial transformations and plant uptake. Effective C to N ratios influencing residue decomposition are calculated in the model as a function of the content of N in the litter as well as the content of available inorganic N in the soil. As decomposition of the litter takes place, C and N are released (Lowrance et al., 2000).

Inorganic nitrogen, ammonium and nitrate, are both available for immobilization into soil organic matter. Immobilization of nitrate occurs only after all available ammonium has been used. Ammonium may be in solution or adsorbed to the soil matrix. Nitrification is calculated with a first-order rate equation influenced by temperature, soil moisture, and pH. Temperature and soil moisture are variables, recalculated each day but pH is a constant for each zone and layer combination. With increasing amounts of substrate, the rate of nitrification is lagged to represent a delayed microbial response. The calculation of denitrification is a function of the interaction of factors representing the anaerobic condition, temperature, nitrate, and mineralizable C. Denitrification mostly occurs as water-filled pore space gets above 60% (Inamdar et al., 1999b). Nitrogen transformation processes are shown in Appendix C.

Application of REMM

Study Area

The study area consisted of a second-order watershed, Bonham-South (221 ha), in the Fort Benning military installation, Georgia (Figure 4-1). A detailed description of this site along with the instrumentation and the type of data collected is provided in

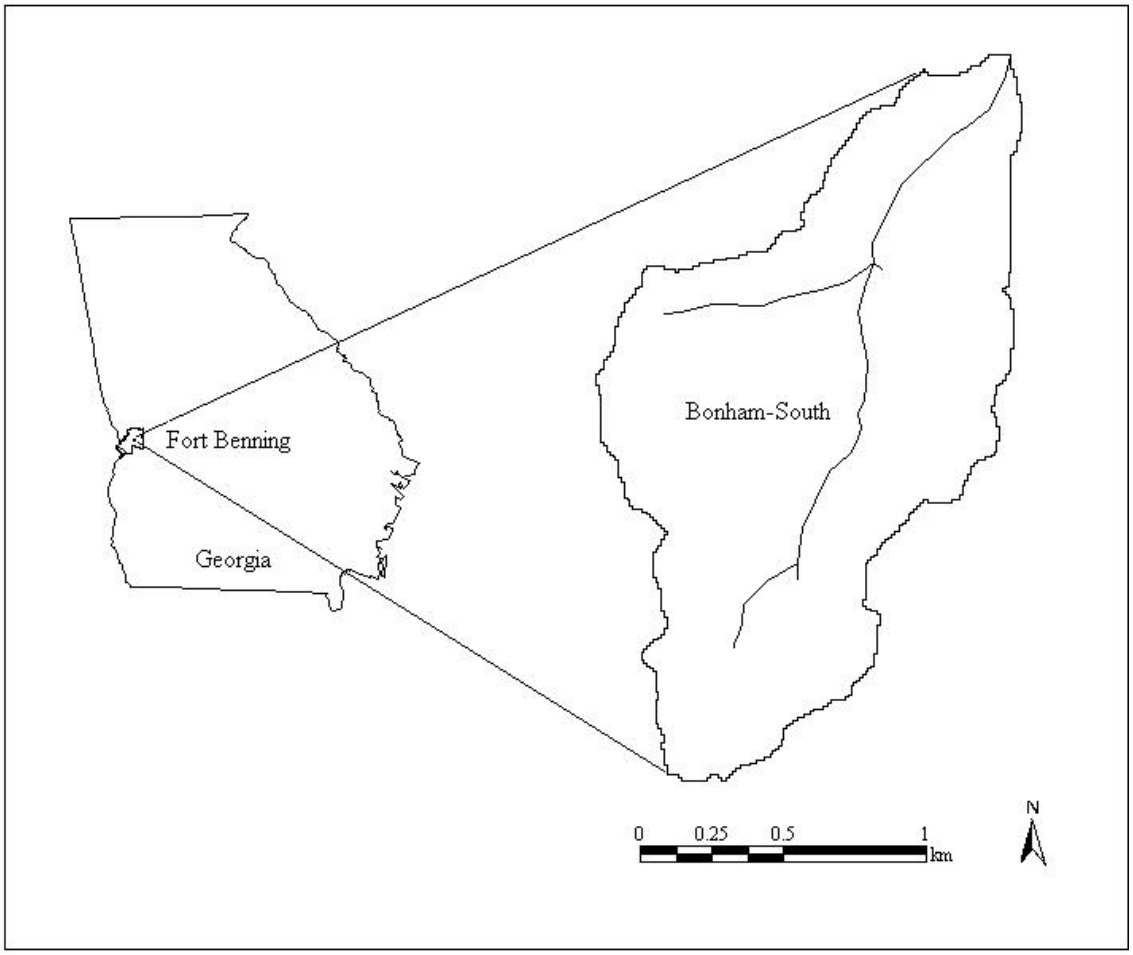


Figure 4-1. Study area. Bonham-South is a second-order watershed within the Fort Benning military installation in Georgia.

Bryant et al., in press and Bhat et al., in review. The riparian area covers approximately 2.5% of the total watershed area. The riparian length is 3,500 m. Based on the visual inspection in the watershed, a 12 m wide riparian buffer along the stream is considered.

This riparian buffer is divided into three equal zones (Zone 1, Zone 2, and Zone 3) of 4 m each. Corresponding soil layers (Soil layer 1, Soil layer 2, and Soil layer 3) are considered in each zone. A selective list of riparian area and hydrological parameters and their corresponding values are provided in Table 4-1. The riparian area consists primarily

Table 4-1. REMM model parameters values by the riparian zone for the Bonham-South watershed in Fort Benning, Georgia. The dimension of riparian area from the upland to the stream perpendicular to the stream is referred to as width, and the distance along the stream is referred to as length. The parameter values are identical for Zone 1, Zone 2, and Zone 3 except for the slope. The superscripts 1, 2, and 3 of the slope values represent the Zone 1, Zone 2, and Zone 3, respectively.

Parameters		Units	Values
Riparian zone length		m	3,500
Riparian zone width		m	4
Slope		%	3.0^1, 3.8^2, 4.2^3
Manning's n		unitless	0.06
Total soil profile thickness		m	3.3
Individual soil horizon thickness		m	
	Soil layer 1		0.3
	Soil layer 2		1.0
	Soil layer 3		2.0
Saturated hydraulic conductivity		cm/h	
	Soil layer 1		15
	Soil layer 2		13
	Soil layer 3		1
Porosity		cm/cm	
	Soil layer 1		0.40
	Soil layer 2		0.35
	Soil layer 3		0.35
Wilting point		cm/cm	
	Soil layer 1		0.07
	Soil layer 2		0.07
	Soil layer 3		0.07
Riparian area		ha	4.2
Field drainage area (surface)		ha	216.8
Field drainage area (subsurface)		ha	221.0

of hardwoods (approximately 60%) that are mostly scrub oak (*Q. rubra*), sweetbay (*M. virginiana*), sweet gum (*L. styraciflua*), water oak (*Q. nigra*), willow oak (*Q. phellos*), red maple (*A. rubrum*), and swamp tupelo (*N. aquatica*). Mixed species including southern scrub oak (*Q. rubra*) and yellow pine (*P. ponderosa*) cover approximately 21% of the riparian area, whereas loblolly (*P. taeda*) and longleaf (*P. palustris*) pine cover approximately 19%. The riparian area soils are predominantly sandy loam. The uplands are mostly on loamy sand.

Data and Input Parameters

REMM operates at a hillslope scale. To apply the model at watershed scale, it is necessary to convert the watershed into an equivalent hillslope by adjusting the geometry of watershed. As the riparian buffer has specified length (parallel to the stream) and width (perpendicular to the stream), upland contributing field width of the watershed was adjusted to equate the total area of the watershed.

REMM requires soil and vegetation nutrient pools for the riparian area of the watershed to initialize the model (Table 4-2). Required field data to run the model includes daily surface and subsurface runoff from the uplands to the riparian area (Table 4-3). The model also requires daily weather data and parameter values representing topographic, soil, and vegetative conditions within the riparian area. Strategic Environmental Research and Development Plan's (SERDP) Ecosystem Management Project (SEMP) continuously records the atmospheric data within the military installation. Daily weather data were obtained from SEMP data repository that included precipitation amount and duration, maximum and minimum air temperatures, solar radiation, wind speed, and dew point temperature. Daily streamflow for Bonham-South

Table 4-2. Soil and vegetation nutrient pools for Bonham-South watershed's riparian area. The litter and soil pools of nutrient consist of metabolic, structural, active, slow, and passive humus pools. The values in litter layer as well as individual soil layers in each zone are presented. Vegetation pools consist of nutrients in leaves, stems, branches, coarse roots, fine roots, and heartwoods. Total values of nutrients are reported in the vegetation pool. The parameter values are identical for Zone 1, Zone 2, and Zone 3.

Parameters		Depth (m)	Bulk density (g/cm^3)	Carbon (kg/ha)	Nitrogen (kg/ha)	Biomass (kg/ha)
Litter and Soil	Litter	0.01	1.0	18,100	1,134	
	Soil layer 1	0.30	1.4	29,280	655	
	Soil layer 2	1.00	1.5	24,640	644	
	Soil layer 3	2.00	1.5	11,560	252	
	Total			83,580	2,685	
Live vegetation				31,390	753	78,477

Table 4-3. Model simulated annual hydrologic budget for the riparian areas in Bonham-South watershed. * and ** represents the surface and subsurface flow input to the zone 3 from uplands. Values are rounded off to nearest whole numbers.

		Year				
	Units	2000	2001	2002	2003	4-yr average
Precipitation	mm/yr	475	736	731	1,014	739
Surface flow in*	mm/yr	-	-	-	-	-
Subsurface flow in**	mm/yr	119	134	143	253	162
Observed stream flow (total)	mm/yr	121	137	147	258	166
Simulated surface flow	mm/yr	2	5	4	6	4
Simulated subsurface flow	mm/yr	106	118	130	244	149
Simulated stream flow (total)	mm/yr	109	124	133	250	154

was calculated from 10-minute continuous stage records using rating curves and area-velocity method.

Typically, REMM uses daily surface runoff depth generated from the runoff collected by surface runoff samplers at the upland-Zone 3 interface. REMM uses the hydraulic gradient upland wells and Zone 3 wells, saturated thickness at Zone 1, and the saturated hydraulic conductivity to compute subsurface flow loading to the buffer. Due

to the lack of measured surface and subsurface flow from the upland to the riparian area, it is assumed that the baseflow fraction in the measured streamflow is a contribution of upland subsurface flow to the riparian area. Therefore, measured streamflow from January 2000 through December 2003 was partitioned into baseflow and surface runoff using constant slope base flow separation technique (McCuen, 1998). Surface runoff from the upland to the riparian area was not considered (Table 4-3). Parameters that describe the riparian area dimensions, soil, and vegetation characteristics were derived from measured data and previously published literature (Inamdar et al., 1999a, Inamdar et al., 1999b, Lowrance et al., 2000; Garten Jr., et al., 2003). Important parameters related to litter and soil layer are listed in Appendix D. Monthly canopy cover in the riparian area was determined by direct measurement with a Model-A spherical densiometer using the method outlined by Lemmon (1956) from June 2001 to September 2003.

Measured water and soil nutrient concentrations in the study area included the total Kjeldahl nitrogen (TKN), nitrate, and ammonium in precipitation, streamflow, soil water, and shallow groundwater. Two transects in the riparian area near the outlet of the watershed were considered for groundwater and soil water monitoring. Groundwater monitoring wells (1-3 m deep) and adjacent tension lysimeters at 20 and 60 cm depths were positioned on both sides of the stream along the riparian transects. Stream water, groundwater, and soil water samples were collected from October 2001 to September 2003; biweekly from October 2001 to November 2002 and monthly thereafter. Bonham-South stream was sampled during 16 storm events between September 2002 and September 2003 using an event triggered ISCO sampler that was programmed to collect

hourly samples based on the flow depth. Bhat et al. (unpublished manuscript), describes the sample analysis procedures for different chemical constituents.

As REMM simulations are performed on a daily time step, and observed TKN concentrations are of hourly time step, comparisons between the two were based on the TKN masses produced during the events that lasted 24 hours or longer. The model simulates the masses of dissolved and particulate organic nitrogen, dissolved ammonium and nitrate in surface and subsurface flows on a daily basis. The mass of simulated TKN is the sum of dissolved and particulate organic nitrogen and dissolved ammonium. Carbon dissolves from metabolic residue and active humus pools. In REMM, incoming C from precipitation, surface, and subsurface flow is assumed to be in dissolved form. Carbon available to be dissolved from the active humus pool is a fraction of the total C present in the same pool. This fraction of set at 0.31 based on an estimate by McGill et al. (1981) for the proportion of dying bacteria in metabolic form. The total amount of dissolved C is the sum of metabolic C residue and the dissolved C fraction in the active humus pool. In REMM, stoichiometric relationships are assumed between C and N in the organic matter. As C is transformed, corresponding N is also transformed. A Freundlich isotherm method determines the amount of dissolved ammonium from inorganic pool of N in REMM. Observed masses of TKN during storm events are calculated by multiplying the concentrations by the volume of water.

Model Calibration

REMM's hydrology and nitrogen components were evaluated using data collected in this study from riparian area as well as data from the literature. Data collected in nearby watersheds in Fort Benning, Georgia were used to initialize the C and N in the soil pools (Garten Jr. et al., 2003). The litter C and N pools are based on the literature

values (Silveira, M.L., Reddy, K.R. (Department of Soil and Water Science, University of Florida), Comerford, N.B. (Department of Soil and Water Science, University of Florida). Litter decomposition and soluble organic carbon and nitrogen release in a forested ecosystem. Unpublished manuscript). Soil and vegetation nutrient pools for the riparian area of the watershed are listed in Table 4-2. Bonham-South's soil layer thickness, soil porosity, saturated hydraulic conductivity, clay content, carbon decay rate, and denitrification rate were calibrated.

Model calibration involved the comparison of the simulated streamflow and TKN output with the measured values. While calibrating hydrology and nutrient components of the model, parameters such as soil and litter C and N pools, which are based on literature values, were fixed. Other fixed parameters included riparian length and width, surface and subsurface draining area. The remaining parameters including soil layer thickness, saturated hydraulic conductivity, soil porosity, clay content, carbon decay rate, and denitrification rate in each zone and soil layers of the riparian buffer were adjusted to match the observed and predicted results.

Fox (1981) recommends mean biased error (MBE) and mean absolute error (MAE) to measure the difference between observed and model predicted values. Mean biased error is calculated as the average error between predicted and observed values accumulated over the total number data points. Mean absolute error considers the absolute values of the errors. In the present study, MBE and MAE are modified to calculate the percent difference of modeled streamflow and TKN mass from the observed values. Mean bias difference (MBD) calculates the average difference accumulated over the total number of events, expressed as a percent of the observed value. Mean absolute

difference (MAD), expressed also as a percent, considers the absolute values of the differences. Mean absolute difference takes negative values and replaces them with their absolute values. Observed and modeled streamflow was also compared using the Nash-Sutcliffe efficiency. The Nash-Sutcliffe efficiency criteria is based on the normalized least square objective function that evaluates the sum of the squares of flow residuals (Nash and Sutcliffe, 1970).

Sensitivity Analysis

A sensitivity analysis was performed to determine the effects of key hydrological, soil, and vegetation parameters on streamflow and TKN fluxes. The parameters were canopy cover fraction, riparian zone width, soil layer thickness, maximum carbon decay rate for litter and humus, maximum denitrification rate, soil saturated hydraulic conductivity, soil porosity, and soil clay content. Each parameter was changed by +10% and –10% from the values used as the best estimates for the calibration simulations.

Results

Hydrology

Simulated daily flows for the study watershed closely matched the observed flows (Figure 4-2). Over the 4-year simulation period, MBD and MAD between observed and predicted streamflow were 2% and 12%, respectively. The model had Nash-Sutcliffe efficiency of 80%. REMM tends to underestimate the streamflow during low flow and overestimate during the storms. Approximately 1% of the streamflow was the contribution from surface runoff generated in the riparian area. Majority of streamflow output was through the subsurface (Table 4-2).

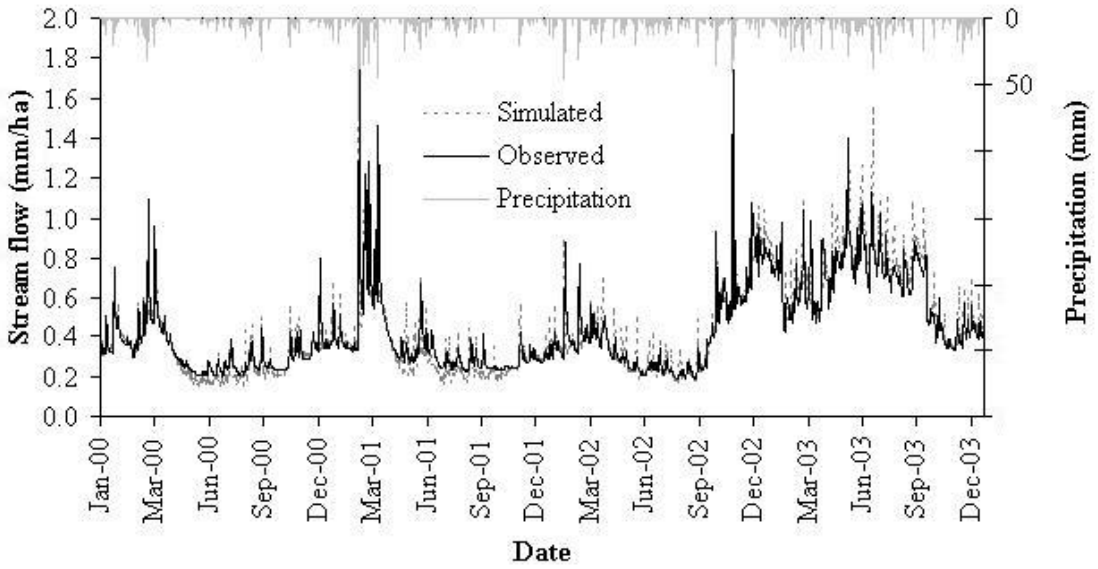


Figure 4-2. Comparison of REMM simulated daily flow with the observed daily flow for the Bonham-South watershed.

There was considerable variability between and within years for measured precipitation and streamflow data during study period. It is evident from Figure 4-2 that an increase in the average flow during the year 2003. This increase was consistent with the variation in precipitation during the study period. Overall, model simulations for streamflow can be considered good for the calibrated parameters.

Nitrogen

Observed stream TKN concentrations during the study period showed a strong correspondence with the leaf fall (Figure 4-3). Approximately 50% of the canopy in the riparian area dropped during November-January each year. Higher TKN concentrations were observed during the same period of time. The simulated TKN concentrations also correspond to simulated leaf fall (Figure 4-4). Modeled TKN concentrations were higher during high leaf fall months. From October 2001 to December 2003, modeled leaf fall totaled 12,200 kg/ha, which corresponds to 5,400 kg/ha/yr. As depicted in the figure,

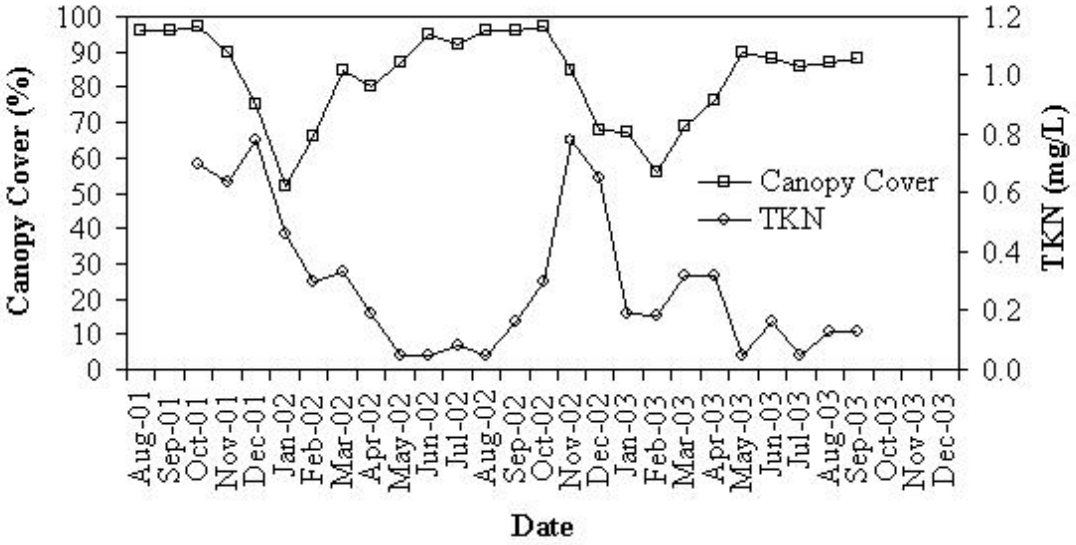


Figure 4-3. Observed canopy cover and TKN concentrations each month for the Bonham-South watershed.

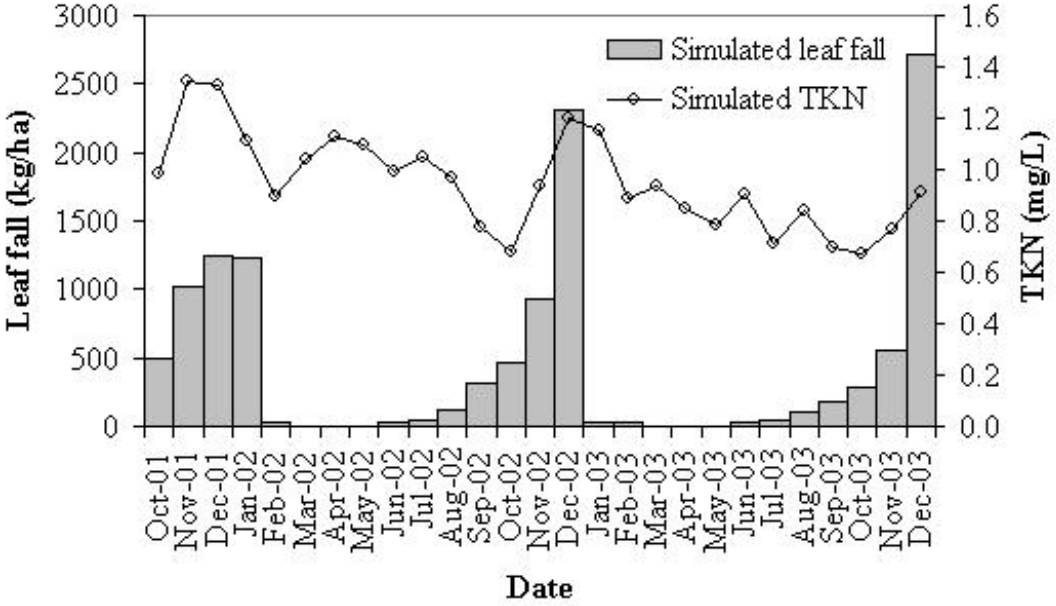


Figure 4-4. Monthly simulated TKN concentration and simulated leaf mass in the Bonham-South's riparian area.

simulated leaf fall began as early as June each year and completed by the end of December-January. As observed in the riparian area of the watershed, leaves were present in the canopy year-round. For example, during the 2001-2002 dormant season,

approximately 45% of the canopy was still covered with leaves. Simulated N present in the live vegetation showed an inverse relation with the simulated TKN concentrations in the stream (Figure 4-5).

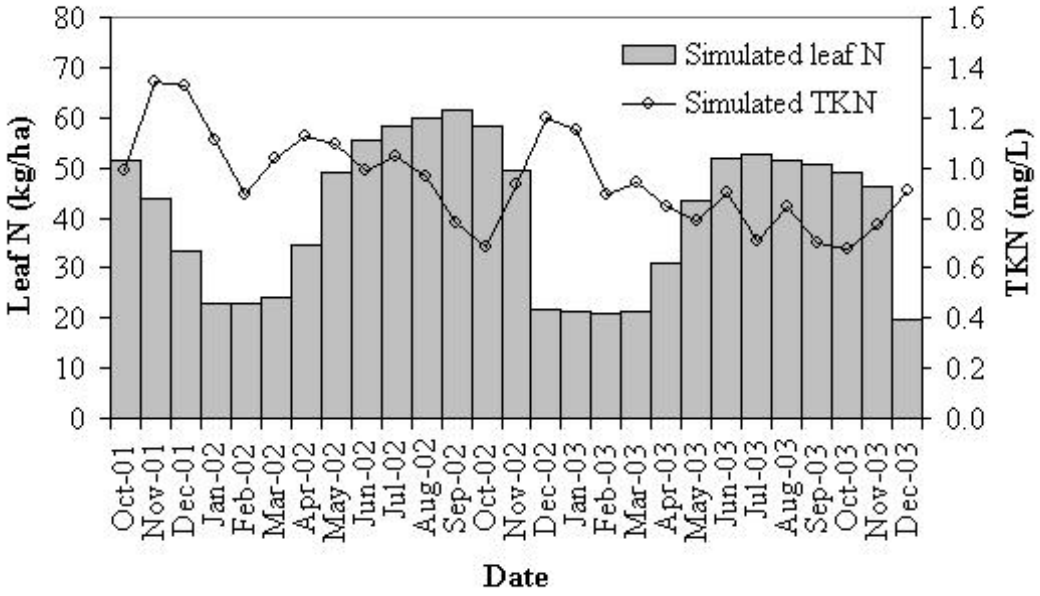


Figure 4-5. Monthly simulated TKN concentration and simulated nitrogen present in live leaves the Bonham-South's riparian canopy.

A careful analysis of model simulation results showed that a daily total precipitation of 6 mm or more produced surface runoff in the riparian area. This surface runoff is responsible for leaching and transporting TKN from the leaf litter in the riparian zone to the stream. Due to the irregular nature of surface runoff generation, the TKN variability during such events, the frequencies of observed data, and REMM's ability to simulate on a daily basis, storm events lasting 24 hours or longer were selected for further study. The masses of TKN were calculated for two different scenarios. First, total mass of TKN during an event was calculated for the total streamflow. Second, baseflow was separated from the total flow, and TKN mass corresponding to surface runoff was determined. The observed and simulated TKN masses during individual storm events

were compared for both the scenarios. Out of the 16 storm samples collected in the study area, 6 storms between October 2002 and May 2003 were suitable for the comparison with the daily output from REMM (Figure 4-6). The simulated TKN masses produced in surface runoff during the storms were comparable to the observed values (Figure 4-7). MBD and MAD between the observed and the simulated TKN masses were 8% and 23%, respectively. As the majority of the flow in the stream is subsurface flow, the effect of subsurface flow in carrying the TKN mass was also analyzed. The result showed that MBD and MAD between observed and simulated total TKN masses were 17% and 24%, respectively (Figure 4-8).

Sensitivity Analysis

The changes in streamflow and TKN outputs for the –10% to +10% parameter changes ranged from 0 to 1% and 0 to 7%, respectively (Table 4-4). Changes in soil porosity did not have significant effects on the streamflow and TKN. A decrease in soil porosity of 10% led to a total streamflow increase of only 0.5%, whereas the increase in TKN was only 0.6%. Increase in clay content decreased the TKN output by 0.7%. Increase in saturated hydraulic conductivity by 10% did not affect the streamflow but increased TKN by 7.2%. Increasing soil layer thickness slightly reduced the streamflow, but reduced TKN by 5.4%.

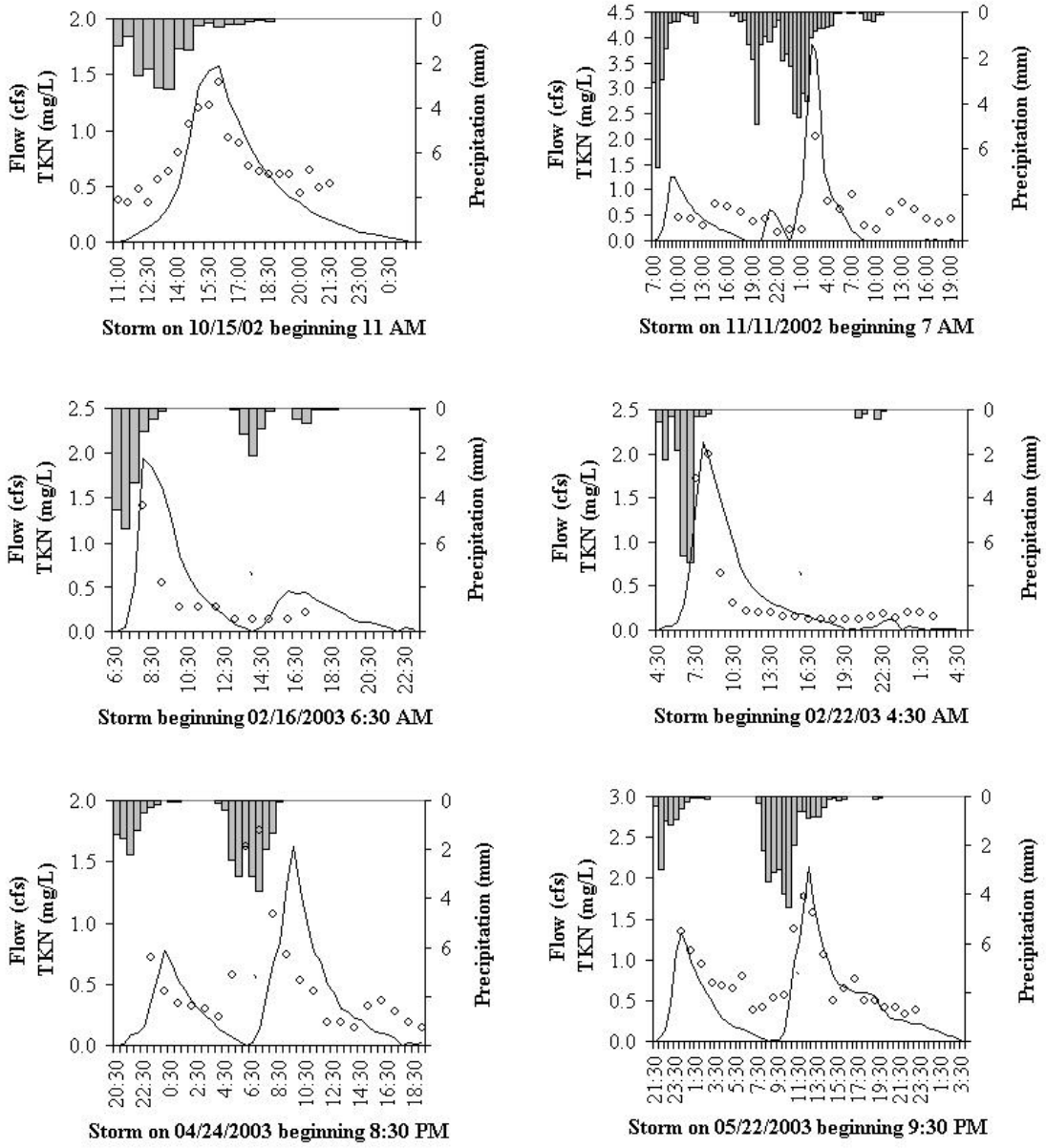


Figure 4-6. Observed TKN concentrations during the events. Vertical bars represent precipitation, solid lines represent the streamflow, and the hollow circles represent the TKN concentrations.

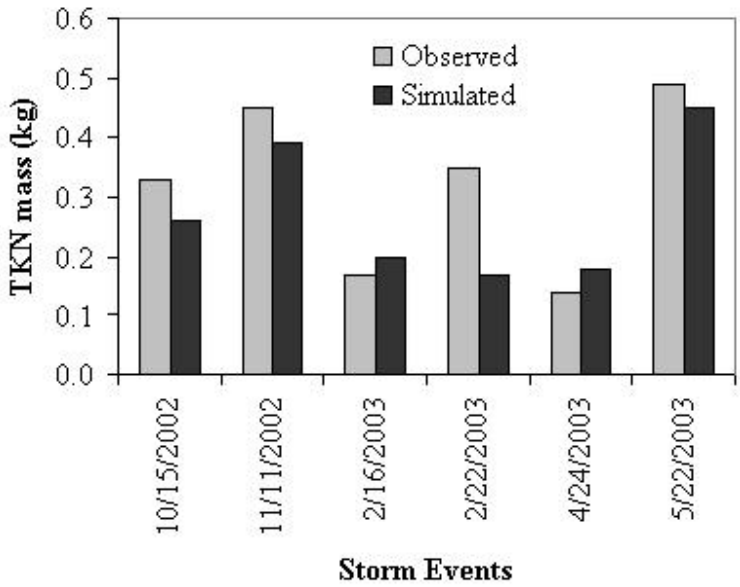


Figure 4-7. Comparison of the observed and simulated TKN masses during the events. These masses represent the contribution from the surface runoff only.

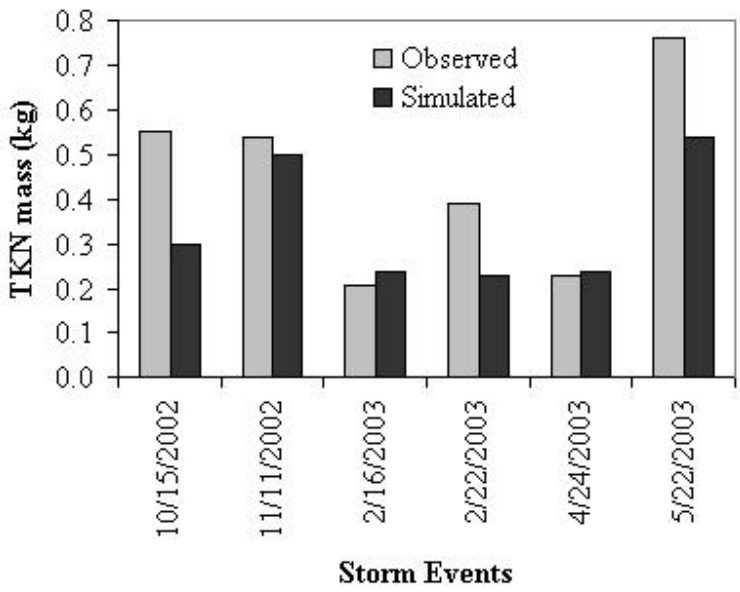


Figure 4-8. Comparison of the observed and simulated TKN masses during the events. These masses represent the contribution from the total flow.

Table 4-4. Sensitivity of modeled streamflow and TKN based on +/- 10% change in model parameters for Bonham-South watershed.

	Percentage change	
Parameters	Stream flow	TKN
Canopy cover (+)	-0.9	2.3
Canopy cover (-)	0.9	-0.2
Soil layer thickness (+)	-0.1	-5.4
Soil layer thickness (-)	0.1	5.7
Denitrification rate (+)	0.0	0.0
Denitrification rate (-)	0.0	0.0
Carbon decay rate (+)	0.0	0.9
Carbon decay rate (-)	0.0	-0.5
Saturated hydraulic conductivity (+)	0.0	7.2
Saturated hydraulic conductivity (-)	0.0	-0.2
Soil porosity (+)	-0.4	-1.5
Soil porosity (-)	0.5	0.6
Clay content (+)	0.0	-0.7
Clay content (-)	0.0	0.4

Discussion

Over the four years, the simulated average annual surface runoff that is generated in the riparian area and contributed to the streamflow was less than 1% of the annual observed precipitation (Table 4-3). The simulated surface runoff generated in the riparian buffer in the study watershed is small compared to earlier studies done in a similar watershed in Georgia. A study conducted by Shirmohammadi et al. (1984) in Little River watershed in Georgia reported approximately 4-12% contribution of annual precipitation to surface runoff. Inamdar et al. (1999a) reported the average annual surface runoff contribution to streamflow was approximately 8% of the annual precipitation for Gibbs Farm site within the Little River watershed. Surface runoff contribution to streamflow in our study watershed may not be significant in terms of

hydrology, but the modeled results suggest that it is capable of transporting nitrogen from the litter and top soil layer to the stream (Figure 4-7). Subsurface contribution to the streamflow for the Bonham-South watershed was approximately 20% of the average annual precipitation (Table 4-3). The simulated subsurface contribution in this study is close to the range of 14-22% reported by Shirmohammadi et al. (1984) for Little River watershed.

The riparian forest community found along Bonham-South watershed is representative of the Southeastern riparian forest (Shure and Gottaschalk, 1985; Lowrance et al., 2000). Slash pine (*P. elliotii*), longleaf pine (*P. palustris*), black gum (*N. sylvatica*), sweet gum (*L. styraciflua*), scrub oak (*Q. rubra*), sweetbay (*M. virginiana*), water oak (*Q. nigra*), willow oak (*Q. phellos*), red maple (*A. rubrum*), and swamp tupelo (*N. aquatica*) are often dominant canopy species in these riparian forests. The modeled total leaf fall in the study watershed from October 2001 to December 2003 was approximately 12,200 kg/ha, which corresponds to 5,400 kg/ha/yr. The modeled average litter mass produced during the same period was approximately 7,000 kg/ha/yr. Approximately 77% of the litter mass were leaves. Contribution of the modeled leaf fall to the total litter production falls in the neighborhood of 80% reported by Meentemeyer et al. (1982), and within the range of 72-84% reported by Shure and Gottaschalk (1985). The simulated litter mass produced in the study area is higher than the average 5,000 kg/ha for a mixed hardwood forest in the Northeastern U.S. (Gosz et al., 1972) and is comparable with the results found in the Southeastern U.S. Mulholland (1981) reported litter production of 6,100 kg/ha in a small stream swamp in eastern North Carolina. A bottomland hardwood forest and a cypress-tupelo stand located in Louisiana produced

litter masses of 5,750 kg/ha and 6,200 kg/ha, respectively (Conner and Day Jr., 1976). Brinson et al. (1980) reported a value of 6,500 kg/ha for an alluvial swamp forest in North Carolina.

Simulated results showed that the nitrogen content in the litter during the study period averaged 77 kg/ha of which approximately 59 kg/ha was the contribution from the leaves alone. This value is less than 83.4 kg/ha of nitrogen contribution from the deciduous leaves in the Northeastern U.S. (Gosz et al., 1972). As observed in the riparian area, 45% canopy was present even during the dormant seasons because of the presence of approximately 19% of conifers and 21% of mixed species. Modeled results also support the presence of the carbon in the leaves and the biomass in the canopy. The modeled result showed that on an average approximately 527 kg/ha of carbon was present in the canopy during the dormant season. This value of carbon suggests the presence of approximately 22 kg/ha of nitrogen in the canopy during the same time (Figure 4-5). Higher concentrations of TKN during the leaf fall and the inverse relation with the nitrogen present in the canopy support the hypothesis of flushing of nitrogen from the leaf litter.

The simulated TKN concentrations follow the trend of leaf mass accumulation in the riparian area suggesting a higher rate of release of TKN from freshly fallen litter during the fall and early spring. Brinson et al. (1980) reported higher nutrient fluxes following litterfall in an alluvial swamp forest in North Carolina. In a study of nutrient content of litterfall in the Southeastern U.S., Gosz et al. (1972) reported that approximately 56% of the total nutrient was released during September through December when the majority of litterfall (50%) occurred. A study conducted in cypress

swamp forest in Florida, Schlesinger (1978) reported that in the month of November when 56% litterfall occurred, 45% of nitrogen was released from the system.

Higher TKN concentrations were observed in the Bonham-South stream during the precipitation events (Figure 4-6). The storm events presented in Figure 4-6 were long enough to calculate the masses of TKN during the storms to compare REMM's simulated daily values. Comparisons of TKN masses produced during the events were comparable to REMM's simulated values. The masses of TKN during the storms were calculated for two different scenarios. When modeled TKN mass contribution of surface runoff that is produced in the riparian zone was compared with the observed value in the stream, MBD and MAD for the six storms were 8% and 23%, respectively. Mean biased difference value is less reliable as there is the risk that large outliers cancel each other out. As MAD takes negative values and replaces them with their absolute values, large outliers are de-emphasized; hence it is less sensitive to extreme values. A MAD of 23% in the surface runoff suggests that the model was able to produce comparable amount of TKN as observed during the storm. The comparison between observed and simulated TKN mass from the total flow showed comparable MBD (17%) and MAD (24%). On average, 43% of the observed total TKN mass was contributed by surface runoff. This result suggests that the surface runoff during the precipitation events is a major source of leaching nitrogen from the forest floor and transporting it to the stream.

Conclusion and Recommendations

Originally, REMM was developed to operate at a hillslope scale. The results in this study suggested that given appropriate upland inputs for a site, REMM can be used at different scenarios of riparian area width, length of zones, vegetation type, and soil properties. The trend and magnitude of the observed streamflow for the study watershed

was effectively simulated by REMM. The model, however, overestimated the streamflow during high flow and underestimated during low flow periods. The hydrologic budget showed a good agreement between observed and predicted streamflow.

The simulated litter and corresponding leaf masses in the study watershed were comparable to the values reported in the literature. Due to the model's longer simulation time step, a meaningful comparison of simulated and observed TKN was possible only through the masses produced during storms that were equal to or longer than a day. Comparison of TKN masses during six different storm events showed similar values both in the surface runoff and the total flow. The results supported the hypothesis of nitrogen leaching from freshly fallen leaves during the precipitation events. These results provided further insight into the nutrient dynamics of the riparian area. The model simulations respond as expected to precipitation and vegetation patterns over the study period. Results clearly indicated that the presence of fresh leaf litter in the riparian area increases the TKN concentration, and hence mass, in the stream. The model effectively captured the trends of leaf mass accumulation in the riparian area and subsequent high concentration and mass during those periods. With the present version of REMM, the comparison between observed and simulated values is possible only on a daily basis. However, given the fact that the precipitation events and the sampling frequencies are often shorter, simulation results using a smaller time scale would be useful. Therefore, it is recommended to modify REMM from its present version of daily time step to a smaller time step, preferably hourly, for more effective and meaningful interpretation and evaluation of riparian nutrient flushing.

CHAPTER 5
SUMMARY AND CONCLUSION

The National Research Council of the United States has proposed that reliable and comprehensive environmental indicators be developed to monitor ecological changes from natural and anthropogenic causes. The military installation in Fort Benning, Georgia offered a unique opportunity to study military impacts on water quality and quantity. Ecohydrological approaches were used to relate the effects of anthropogenic perturbations on water-vegetation-nutrient interactions in the Fort Benning watersheds. In this research, statistical relationships among water quality parameters and the watershed physical characteristics in low-nutrient watersheds were identified and examined. This research also identified and developed hydrologic indices that characterize the impact of military land management on watershed, and investigated the riparian corridor's role on water quality by quantifying the nitrogen leaching from freshly fallen leaves.

Relationships among watershed physical characteristics and water quality parameters in the study watersheds and the regression analysis showed that pH, chloride, total phosphorus, total Kjeldahl nitrogen, total organic carbon, and total suspended solids are useful indicators of watershed physical characteristics that are susceptible to perturbations. A comparison of the results between a reference and impacted watershed in terms of hydrologic indices that are derived from long-term daily flow as well as the storm-based data showed a clear distinction between the watersheds in terms of hydrological flow regimes. Storm-based magnitude of baseflow index, magnitude and

variability of peak discharge, and the frequency of bankfull discharge were consistent with the results from annual-based analysis. Results showed that these storm-based indices might be used as surrogates to the indices derived from long-term data. The analysis identified the relationships between the extent of military training land, road density, and the number of roads crossing streams with the storm-based baseflow index, bankfull discharge, response lag, and time of rise. For the low nutrient systems of this study, seasonal and storm variations in water quality were found to be strongly influenced by precipitation events which caused nitrogen leaching from recently fallen leaf litter and increased nitrogen levels in the stream water.

This study showed that the signatures of military alterations to watershed landscape are detectable in the water quality and the flow regime. The observed alterations to these regimes suggest impacts to aquatic ecosystems. The water quality related indicators and hydrologic indices presented in this research provide specific measures of stream water quality and instream flow, respectively, that respond to military impacts. Each indicator is linked to one or more readily measured watershed scale factors. The indicators could be useful in predicting effects of military land management and evaluating restoration activities on the quality as well as the quantity of the water.

This research has several implications for the Fort Benning military installation. The results of this study indicated that baseflow sampling of water quality can be used to assess the military training impacts on stream water quality. The water quality indicators identified in this study provide measures of the current watershed conditions. The water quality indicators identified in this study provide measures of the current watershed conditions. Changes to stream water quality due to military training within the Fort

Benning military installation can be identified by collecting long-term and routine stream water quality data and comparing indicator values to current conditions. Future activities should develop indicator thresholds beyond which improved management and restoration practices should be implemented. As nutrients are one major factor controlling the quality of the receiving aquatic ecosystems, the thresholds should be based on the ecosystems impacts.

The U.S. Army manages approximately 4.8×10^6 hectares of land for military training, and it has developed the Land Condition Trend Analysis (LCTA) program to systematically monitor terrestrial impacts from military training and to support the mitigation and remediation of severely impacted training lands (Quist et al., 2003). Although the magnitude of the training would differ from one installation to the other, the nature of the impact remains similar. Military training typically increases soil bulk densities and compaction, decreases infiltration, diminishes plant growth, degrades water quality, and affects water quantity. Therefore, the research findings in this study can be implemented for management and restoration practices to other military installations across the U.S. to minimize the impacts of the military training. A common finding between the water quality and quantity signatures is that military training impacts differ from those found due to agriculture and urbanization. In the former, water quality degradation impacts typically manifest as high nutrient loads. However, in military installations, the loss of topsoil and vegetation results in waters that have significantly lower nutrient levels. In a similar fashion, the hydrologic regimes in watersheds with significant military impacts suffer from a loss of variability, but exhibit higher annual discharge values.

APPENDIX A
ANNUAL-BASED INDICES DEFINITIONS AND CALCULATION PROCEDURES

Symbol	Definition	Method
M_A26	Coefficient of variation in monthly flows for December	Calculate mean monthly flow for December (q_{di}), i = 1,…, n; , n = no. of years Calculate mean of all December flows, $\bar{Q}_D = \sum_{i=1}^{n} q_{di} / n$ Calculate standard deviation (SD) of q_{di} Calculate coefficient of variation, $CV = SD / \bar{Q}_D$
M_A41	Mean annual flow divided by watershed area	Calculate mean yearly flow (q_{yi}), i = 1,…, n; , n = no. of years Divide q_{yi} by watershed area (A) Calculate $\bar{Q}_y / A = \sum_{i=1}^{n} \left(\frac{q_{yi}}{A} \right) / n$, i = 1,…, n; , n = no. of years
M_A10	Ranges in daily flows divided by median daily flows (where range in daily flows is the ratio of 20th/80th percentiles in daily flows across all years)	Combine daily flows of all years Calculate 20th percentile (Q_{p20}) Calculate 80th percentile (Q_{p80}) Calculate median flow (Q_m) Calculate $R = Q_{p20} / Q_{p80}$ Calculate R / Q_m
M_L17	Seven-day minimum flow divided by mean annual daily flows	Calculate mean yearly flow (q_{yi}), i = 1,…, n; , n = no. of years Calculate 7-day minimum $(q_{7\min})_i$ flow Calculate ratios $R_i = (q_{7\min})_i / q_{yi}$ Calculate $\sum_{i=1}^{n} R_i / n$

M_L14	Mean of the lowest annual daily flow divided by median annual daily flow averaged across all years	Extract lowest yearly flow ($q_{i,\min}$), i = 1,…, n; , n = no. of years Calculate $\bar{Q}_{\min} = \sum_{i=1}^{n} q_{i,\min} / n$ Calculate median flow (Q_m) of all years Calculate $\bar{Q}_{\min} / Q_m$
M_L16	Median of the lowest annual daily flows divided by median annual daily flows averaged across all years	Extract lowest yearly flow ($q_{i,\min}$), i = 1,…, n; , n = no. of years Calculate median (Q_{med}) of $q_{i,\min}$ Calculate median flow (Q_m) of all years Calculate Q_{med} / Q_m
M_H23	Mean of the high flow volume (calculated as the area between the hydrograph and the upper threshold during the high flow event) divided by median annual daily flow across all years. The upper threshold is defined as 7 times median annual flow	Calculate yearly median flow ($q_{i,med}$), i = 1,…, n; , n = no. of years Calculate 7 times $q_{i,med}$ Calculate the flow volume (Q_{Vi}) above 7 times $q_{i,med}$ value Calculate $\bar{Q}_V = \sum_{i=1}^{n} Q_{Vi} / n$ Calculate median flow (Q_m) of all years Calculate $\bar{Q}_V / Q_m$
M_H8	Mean of the maximum monthly flows for the month of May	Extract maximum flow for May (q_{mi}), i = 1,…, n; , n = no. of years Calculate mean of all May maximum flows, $\bar{Q}_{May} = \sum_{i=1}^{n} q_{mi} / n$
M_H14	Median of the highest annual daily flow divided by the median annual daily flow averaged across all years	Calculate median flow (Q_{mi}),i = 1,…, n; , n = no. of years Calculate $\bar{Q}_m = \sum_{i=1}^{n} Q_{mi} / n$ Extract highest daily flow (Q_{hi}) for each year Calculate median of highest daily flow (Q_{medh}) Calculate $Q_{medh} / \bar{Q}_m$

F_L3	Total number of low flow spells (threshold equal to 5% of mean daily flow) divided by the record length in years	Combine daily flows of all years Calculate mean daily flow (Q_{mean}) Determine the threshold of 0.05* Q_{mean} Determine the numbers of low flow counts below 0.05* Q_{mean} (N_{low}) Calculate N_{low}/n, n = record length in years
F_L2	Coefficient of variation in Fl1, where Fl1 is low flood pulse count, which is the number of annual occurrences during which the magnitude of flow remains below a lower threshold. Hydrologic pulses are defined as those periods within a year in which the flow drops below the 25th percentile (low pulse) of all daily values for the time period	Calculate 25th percentile value for each year (Q_{p25i}),i = 1,…, n; , n = no. of years Determine the numbers of low flow pulses below 25th percentile for each year (N_{l25i}) Calculate mean of low flow pulses, $\bar{N}_{l25} = \sum_{i=1}^{n} N_{l25i} / n$ Calculate standard deviation (SD) of N_{l25i} Calculate $CV = SD/\bar{N}_{l25}$
F_H4	High flood pulse count is the number of annual occurrences during which the magnitude of flow remains above an upper threshold where the upper threshold is defined as 7 times median daily flow, and the value is represented as an average instead of a tabulated count	Calculate yearly median flow ($q_{i,med}$), i = 1,…, n; , n = no. of years Calculate 7 times $q_{i,med}$ Determine the number of high flood pulses above 7 times $q_{i,med}$ (N_{7hi}) Calculate $\bar{N}_{7h} = \sum_{i=1}^{n} N_{7hi} / n$
F_H6	Mean number of high flow events per year using an upper threshold of 3 times median flow over all years	Calculate yearly median flow ($q_{i,med}$), i = 1,…, n; , n = no. of years Calculate 3 times $q_{i,med}$ Determine the number of high flood pulses above 3 times $q_{i,med}$ (N_{3hi}) Calculate $\bar{N}_{3h} = \sum_{i=1}^{n} N_{3hi} / n$

F_H7	Mean number of high flow events per year using an upper threshold of 7 times median flow over all years	Same as F_H4
D_L10	Coefficient of variation in annual minima of 90-day means of daily discharge	Determine 90-day minimum flow ($Q_{90\min i}$) for each year, i = 1,…, n; , n = no. of years Calculate $\bar{Q}_{90\min} = \sum_{i=1}^{n} Q_{90\min i} / n$ Calculate SD of $Q_{90\min i}$ Calculate $CV = SD / \bar{Q}_{90\min}$
D_L17	Coefficient of variation in low flow pulse durations	Calculate 25th percentile value for each year (Q_{p25i}),i = 1,…, n; , n = no. of years Determine duration between low flow pulses below 25th percentile for each year (D_{l25i}) Calculate mean duration of low flow pulses, $\bar{D}_{l25} = \sum_{i=1}^{n} D_{l25i} / n$ Calculate standard deviation (SD) of D_{l25i} Calculate $CV = SD / \bar{D}_{l25}$
D_L6	Coefficient of variation in annual minima of 1day means of daily discharge	Determine 1-day minimum flow ($Q_{1\min i}$) for each year, i = 1,…, n; , n = no. of years Calculate $\bar{Q}_{1\min} = \sum_{i=1}^{n} Q_{1\min i} / n$ Calculate SD of $Q_{1\min i}$ Calculate $CV = SD / \bar{Q}_{1\min}$
D_H13	Mean annual 30-day maximum divided by median flow	Determine 30-day maximum flow ($Q_{30\max i}$) for each year, i = 1,…, n; , n = no. of years Calculate $\bar{Q}_{30\max} = \sum_{i=1}^{n} Q_{30\max i} / n$ Calculate median flow (Q_{mi}),i = 1,…, n; , n = no. of years

		Calculate $\bar{Q}_m = \sum_{i=1}^{n} Q_{mi} / n$ Determine $\bar{Q}_{30\max} / \bar{Q}_m$
D_H16	Coefficient of variation in high flow pulse durations	Calculate 75th percentile value for each year (Q_{p75i}), i = 1,…, n; , n = no. of years Determine duration between high flow pulses above 75th percentile for each year (D_{h75i}) Calculate mean duration of high flow pulses, $\bar{D}_{h75} = \sum_{i=1}^{n} D_{h75i} / n$ Calculate standard deviation (SD) of D_{h75i} Calculate $CV = SD / \bar{D}_{h75}$
D_H24	Mean annual maximum number of 365 days over all water years during which no floods occurred over all years	Determine flow of magnitude exceeding a return interval of 1.67 years based on log-normal distribution ($Q_{1.67yri}$), i = 1,…, n; , n = no. of years Determine the maximum number of 1.67-year flow non-exceedance days each year ($D_{1.67yr\max i}$) Calculate $\bar{D}_{1.67yr\max} = \sum_{i=1}^{n} D_{1.67yr\max i} / n$
T_A1	Constancy [Colwell (1974)]	Standardize the daily flow values by median flow, and express as natural logarithm values Divide the log-transformed flow values into 4-6 classes so that these classes cover all the observed flow values Create a matrix of total number of days in a year (columns, t) by number of classes (rows, s) Define the column totals (X_j), row totals (Y_i), and the grand total (Z) as $X_j = \sum_{i=1}^{s} N_{ij}$, $Y_i = \sum_{j=1}^{t} N_{ij}$, and $Z = \sum_i \sum_j N_{ij} = \sum_j X_j = \sum_i Y_i$ where N_{ij} is the number of cycles for which the flow is in class i and time j. Determine uncertainty with respect to time $H(X) = -\sum_{j=1}^{t} \frac{X_j}{Z} \log \frac{X_j}{Z}$

		Determine uncertainty with respect to class $H(Y) = -\sum_{i=1}^{s} \frac{Y_i}{Z} \log \frac{Y_i}{Z}$ Determine uncertainty with respect to the interaction of time and scale $H(XY) = -\sum_{i} \sum_{j} \frac{N_{ij}}{Z} \log \frac{N_{ij}}{Z}$ Determine predictability (P) with the range (0, 1) $P = 1 - \frac{H(XY) - H(X)}{\log s}$ where s is the number of classes Determine the constancy (C) with range (0, 1) $C = 1 - \frac{H(Y)}{\log s}$
T_A3	Maximum proportion of all floods over the period of record that fall in any one of six 60-day 'seasonal' windows	Determine flow of magnitude exceeding a return interval of 1.67 years based on log-normal distribution ($Q_{1.67\,yr}$) over the period of record Divide every year into six 60-day 'seasonal' windows Determine the maximum number of $Q_{1.67\,yr}$ exceedance days ($D_{1.67\,yr\,\max}$) over period of record in any one of six 60-day 'seasonal' windows Determine the total number of $Q_{1.67\,yr}$ exceedance days over all years in one of six 60-day seasonal window ($D_{1.67\,yrtot}$) Calculate the ratio ($D_{1.67\,yr\,\max} / D_{1.67\,yrtot}$)
T_H3	Maximum proportion of the year (number of days/365) during which no floods have ever occurred over the period of record	Determine flow of magnitude exceeding a return interval of 1.67 years based on log-normal distribution ($Q_{1.67\,yr}$) over the period of record Determine the maximum number of $Q_{1.67\,yr}$ non-exceedance days ($D_{1.67\,yrnon}$) over period of record Determine the total number of $Q_{1.67\,yr}$ non-exceedance days over period of record ($D_{1.67\,yrtot}$) Calculate the ratio ($D_{1.67\,yrnon} / D_{1.67\,yrtot}$)

R_A9	Coefficient of variation in number of negative and positive changes in water conditions from one day to the next	Determine number of rises from one day to the other each year (N_{Ri}), i = 1,…, n; , n = no. of years Determine number of falls each year (N_{Fi}) Calculate $\bar{N}_R = \sum_{i=1}^{n} N_{Ri} / n$ Calculate SD_R of N_{Ri} Calculate $\bar{N}_F = \sum_{i=1}^{n} N_{Fi} / n$ Calculate SD_F of N_{Fi} Calculate $CV_R = SD_R / \bar{N}_R$ Calculate $CV_F = SD_F / \bar{N}_F$
R_A7	Median of difference between natural logarithm of flows between two consecutive days with decreasing flow	Transform the daily flow values for each year using natural logarithm, Q_i = ln Q_i, i = no. of days For all ln Qs each year, determine [ln Q_i – ln Q_{i+1}] Separate the values of decreasing flow differences for each year Determine the median of decreasing flow differences each year (Q_{dmj}), j = 1, …, n; n = no. of years Calculate $\bar{Q}_{dm} = \sum_{j=1}^{n} Q_{dmj} / n$
R_A6	Median of difference between natural logarithm of flows between two consecutive days with increasing flow	Transform the daily flow values for each year using natural logarithm, Q_i = ln Q_i, i = no. of days For all ln Qs each year, determine [ln Q_i – ln Q_{i+1}] Separate the values of increasing flow differences for each year Determine the median of increasing flow differences each year (Q_{imj}), j = 1, …, n; n = no. of years Calculate $\bar{Q}_{im} = \sum_{j=1}^{n} Q_{imj} / n$

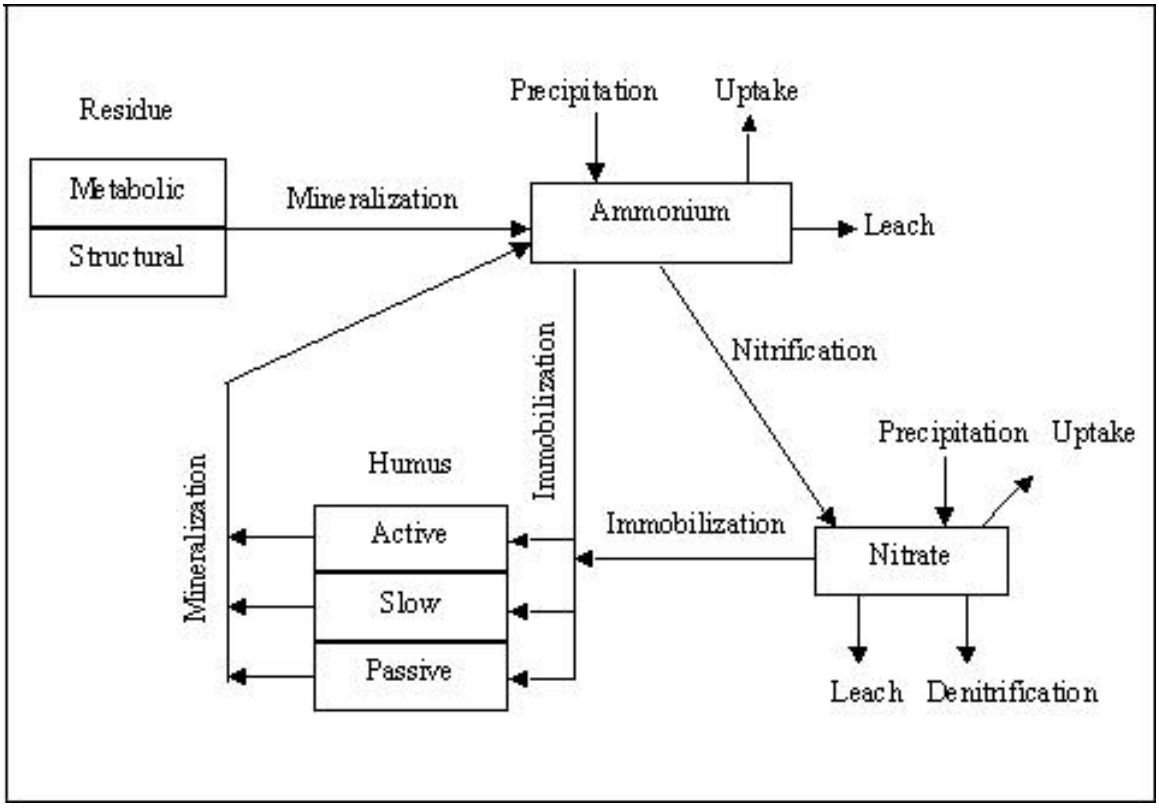
APPENDIX B
STORM-BASED INDICES DEFINITIONS AND CALCULATION PROCEDURES

Index Symbol	Definition	Methods
$M_M RF$	Mean value of the response factor	Calculate precipitation depth for each event (P_i), i = 1,…,n; n = no. of events Create hydrograph Separate baseflow Calculate DRO depth after deducting the baseflow portion from the hydrograph for each event (D_i) Calculate response factor for each event, $RF_i = D_i / P_i$ Calculate the mean, $\bar{RF} = \sum_{i=1}^{n} RF_i / n$
$M_V RF$	Coefficient of variation in response factors	Calculate $\bar{RF}$ Calculate standard deviation (SD) of RF_i Calculate $CV = SD / \bar{RF}$
$M_M BF$	Mean value of baseflow index	Create hydrograph Separate baseflow Determine baseflow volume (V_b) and total volume (V_t) for each events Calculate $BF_i = (V_b/V_t,)_i$, i = 1,…,n; n = no. of events Calculate the mean, $\bar{BF} = \sum_{i=1}^{n} BF_i / n$
$M_V BF$	Coefficient of variation in baseflow index	Calculate $\bar{BF}$ Calculate (SD) of BF_i Calculate $CV = SD / \bar{BF}$
$M_M PD$	Mean value of peak discharges divided by the watershed area	Determine $(q_{pk})_i$ of the event, i = 1,…,n; n = no. of events Normalize $(q_{pk})_i$ by the watershed area, $(q_{pk})_i / A$ Calculate mean, $\left(\bar{q_{pk}} / A\right) = \sum_{i=1}^{n} \left(\frac{q_{pk}}{A}\right)_i / n$

M_VPD	Coefficient of variation in peak discharges	Calculate $\left(\bar{q}_{pk} / A\right)$ Calculate (SD) of $(q_{pk})_i / A$ Calculate $CV = SD / \left(\bar{q}_{pk} / A\right)$
F_1FD	Percentage of peak discharge equals bankfull discharge	Determine bankfull discharge Count peak discharges equal to bankfull discharge for all events and express the count as a percentage of total events
F_2FD	Percentage of peak discharge 2 times above bankfull discharge	Determine bankfull discharge Count peak discharges equal to 2.0 times bankfull discharge for all events and express the count as a percentage of total events
D_MTB	Mean value of time base divided by the watershed response time	Determine time base for each hydrograph Normalize the time base by the response time (T_{bi}), i = 1,…,n; n = no. of events Calculate mean, $\bar{T}_b = \sum_{i=1}^{n} T_{bi} / n$
D_VTB	Coefficient of variation in time base	Calculate $\bar{T}_b$ Calculate (SD) of (T_{bi}) Calculate $CV = SD / \bar{T}_b$
D_MRL	Mean value of response lag divided by the watershed response time	Determine response lag for each hydrograph Normalize the response lag by the response time (T_{rli}), i = 1,…,n; n = no. of events Calculate mean, $\bar{T}_{rl} = \sum_{i=1}^{n} T_{rli} / n$
D_VRL	Coefficient of variation in response lag	Calculate $\bar{T}_{rl}$ Calculate (SD) of (T_{rli}) Calculate $CV = SD / \bar{T}_{rl}$
D_MTR	Mean value of time of rise divided by the watershed response time	Determine time of rise for each hydrograph Normalize the time of rise by the response time (T_{ri}), i = 1,…,n; n = no. of events Calculate mean, $\bar{T}_r = \sum_{i=1}^{n} T_{ri} / n$

D_VTR	Coefficient of variation in time of rise	Calculate $\bar{T}_r$ Calculate (SD) of (T_{ri}) Calculate $CV = SD/\bar{T}_r$
R_PPD	Mean rate of change in peak discharge in rising limb	Determine $(q_{pk})_i$ of the event, $i = 1,\ldots,n$; n = no. of events Normalize $(q_{pk})_i$ by the watershed area, $(q_{pk})_i / A$ Normalize the time of rise by the response time (T_{ri}), $i = 1,\ldots,n$; n = no. of events Calculate the ratio of $(q_{pk})_i / A$ to (T_{ri}) Calculate the mean of the ratios
$R_{PV}PD$	Coefficient of variation in the rate of change in peak discharge in rising limb	Calculate mean of R_PPD_i Calculate (SD) of R_PPD_i Calculate $CV = SD$/mean of R_PPD_i
R_NPD	Mean rate of change in peak discharge in falling limb	Determine $(q_{pk})_i$ of the event, $i = 1,\ldots,n$; n = no. of events Normalize $(q_{pk})_i$ by the watershed area, $(q_{pk})_i / A$ Normalize the time of rise by the response time (T_{ri}), $i = 1,\ldots,n$; n = no. of events Normalize the time base by the response time (T_{bi}) Calculate the difference (T_{bi} - T_{ri}) Calculate the ratio of $(q_{pk})_i / A$ to (T_{bi} - T_{ri}) Calculate the mean of the ratios
$R_{NV}PD$	Coefficient of variation in the rate of change in peak discharge in falling limb	Calculate mean of R_NPD_i Calculate (SD) of R_NPD_i Calculate $CV = SD$/mean of R_NPD_i

APPENDIX C
NITROGEN TRANSFORMATIONS IN REMM



APPENDIX D
PARAMETERS AND RATE CONSTANTS USED IN REMM

Parameters	Units	Values
Litter layer parameters (all zones)		
Layer depth	cm	1
Evaporation factor	unitless	4
Evaporation constant	unitless	-0.45
Litter moisture	mm	2
Litter bulk density	g/cm^3	1
Ammonium adsorption coefficient (a)	unitless	1.102
Ammonium adsorption coefficient (b)	unitless	0.956
Litter pH	unitless	6.2
Litter C structural pool	kg/ha	2613
Litter C metabolic pool	kg/ha	436
Litter C active pool	kg/ha	5
Litter C slow pool	kg/ha	3000
Litter C passive pool	kg/ha	1900
Litter C lignin	kg/ha	9100
Litter ammonium pool	kg/ha	1.34
Litter nitrate pool	kg/ha	0.24
Litter N structural pool	kg/ha	86
Litter N metabolic pool	kg/ha	3
Litter N active pool	kg/ha	1
Litter N slow pool	kg/ha	273
Litter N passive pool	kg/ha	173

Parameters	Units	Values		
		Layer 1	Layer 2	Layer 3
Soil layer parameters (all zones)				
Pore size distribution index	unitless	0.38	0.38	0.38
Bubbling pressure head	cm	1.8	1.8	5.6
Starting moisture content	cm/cm	0.20	0.31	0.35
Saturated conductivity	cm/hr	15	11	1
Sand content	%	75	60	51
Silt content	%	16	9	12
Clay content	%	9	31	38
Bulk density	g/cm^3	1.4	1.5	1.5
Start carbon structural pool	kg/ha	1114	135	135
Start carbon metabolic pool	kg/ha	1787	215	215
Start carbon active pool	kg/ha	400	400	200
Start carbon slow pool	kg/ha	12000	12000	6000
Start carbon passive pool	kg/ha	7600	7600	3800
Start carbon lignin pool	kg/ha	3360	1680	560
Start nitrogen ammonium pool	kg/ha	0.001	0.001	0.001
Start nitrogen nitrate pool	kg/ha	0.001	0.001	0.001
Start nitrogen structural pool	kg/ha	1	1	1
Start nitrogen metabolic pool	kg/ha	81	10	10
Start nitrogen active pool	kg/ha	16	8	8
Start nitrogen slow pool	kg/ha	303	157	157
Start nitrogen passive pool	kg/ha	191	81	81

Rate constants	Values
Carbon release rates (litter and soil layers)	
Metabolic residue pool	0.60
Structural residue pool	0.15
Active humus pool	0.02
Slow humus pool	0.005
Passive humus pool	0.00002
Denitrification rates	
Litter	0.02
Soil layer 1	0.02
Soil layer 2	0.01
Soil layer 3	0.002

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BIOGRAPHICAL SKETCH

Shirish Bhat was born in Pokhara, Nepal, on January 12, 1968. He graduated with a bachelor's degree in science from Tribhuvan University, Nepal, in 1989. He graduated with his second bachelor's degree in civil engineering from Regional Engineering College, Warangal, India, in 1994. He worked as a civil engineer for the Canadian Center for International Studies and Cooperation (CECI) in Nepal from July 1994 to August 1995. He also worked as a civil engineer consultant for the Department of National Parks and Wildlife Conservation, Government of Nepal, from September 1995 to August 1997. In Fall 1997, he began graduate studies in the Department of Civil and Environmental Engineering at South Dakota State University, Brookings, South Dakota, USA, and obtained his master's degree (thesis option) in Summer 1999. Shirish joined the Department of Civil and Coastal Engineering at the University of Florida, Florida, USA, in the Fall of 1999 and will obtain his Ph.D. degree in Spring 2005 under the supervision of Dr. Jennifer Jacobs and Dr. Kirk Hatfield. His primary research interests include land use impacts on surface water quality and quantity, riparian area nutrient dynamics, biogeochemistry of wetlands, and water resources management. He is a student member of the American Geophysical Union (AGU) and the ASCE.