FINAL REPORT

Determination of Indicators of Ecological Change

SERDP Project RC-1114A

SEMPTEMBER 2004

Ramesh Reddy
Joseph Prenger
William F. DeBusk
Wendy Graham
Jennifer Jacobs
Andrew Ogram
Deborah Miller
Suresh Rao
George Tanner
University of Florida
Perdue University

SERDP
DOD • EPA • DOE

REPORT DOCUMENTATION PAGE

Form Approved
OMB No. 0704-0188

1. REPORT DATE *(DD-MM-YYYY)*	2. REPORT TYPE	3. DATES COVERED *(From - To)*
30-09-2994	Technical Report	

4. TITLE AND SUBTITLE
Determination of Indicators of Ecological Change

5a. CONTRACT NUMBER

5b. GRANT NUMBER

5c. PROGRAM ELEMENT NUMBER

6. AUTHOR(S)
Ramesh Reddy, Joseph Prenger, William DeBusk, Wendy Graham, Jennifer Jacobs, Andrew Ogram, Deborah Miller, Suresh Rao, George Tanner

5d. PROJECT NUMBER
RC-1114A

5e. TASK NUMBER

5f. WORK UNIT NUMBER

7. PERFORMING ORGANIZATION NAME(S) AND ADDRESS(ES)
University of Florida, Purdue University

8. PERFORMING ORGANIZATION REPORT NUMBER

9. SPONSORING/MONITORING AGENCY NAME(S) AND ADDRESS(ES)
Strategic Environmental Research & Development Program

10. SPONSOR/MONITOR'S ACRONYM(S)
SERDP

11. SPONSOR/MONITOR'S REPORT NUMBER(S)

12. DISTRIBUTION/AVAILABILITY STATEMENT

13. SUPPLEMENTARY NOTES

14. ABSTRACT
The goal of this research is to develop indicators of ecosystem integrity and impending ecological change that include natural variation and human disturbance. We are evaluating parameters related to properties and processes in the understory vegetation, soil and surface hydrology as potentially sensitive indicators of ecosystem integrity and ecological response to natural and anthropogenic factors. The basic premise is that soil serves as the central ecosystem component that links the quality of the terrestrial habitats (by influencing vegetation and its stability) and the aquatic habitats (via control of soil erosion and overland runoff). We have evaluated potential ecological indicators for sensitivity, selectivity, and ease of measurement.

15. SUBJECT TERMS
SEMP, Ecology, Fort Benning, Terrestrial Ecology, Disturbance

16. SECURITY CLASSIFICATION OF: a. REPORT	b. ABSTRACT	c. THIS PAGE	17. LIMITATION OF ABSTRACT	18. NUMBER OF PAGES	19a. NAME OF RESPONSIBLE PERSON
U	U	U	UU	349	19b. TELEPHONE NUMBER *(Include area code)*

Reset

Standard Form 298 (Rev. 8/98)
Prescribed by ANSI Std. Z39.18

TABLE OF CONTENTS

	Page
Executive Summary	1
1.0 Introduction	9
2.0 Technical Objectives	9
3.0 Results	10
3.1 Soil Biogeochemistry	10
3.1.1 Response of Soil Organic Carbon Dynamics to Disturbance by Military Training in the Southeastern U.S.	11
3.1.2 Influence of military land management on extracellular soil enzymes.	28
3.1.3 Distribution of Methanotrophs in Managed and Highly Degraded Watersheds	51
3.2 Vegetation	68
3.2.1 Vegetative indicators of disturbance in a chronically-disturbed ecosystem	69
3.2.2 Understory Vegetation and Soil response to silvicultural activity in a southeastern mixed pine forest: A chronosequence study.	79
3.3 Hydrology	109
3.3.1 Characterization and Modeling of Throughfall Temporal Variability for Forest Communities in the Southeastern U.S.	110
3.3.2 Hydrologic Indices of Watershed Scale Military Impacts in Fort Benning, GA.	137
3.3.3 Impacts on Soil Saturated Hydraulic Conductivity: Mechanized Military Training Characterization Using Five Measurement Methods.	171

3.3.4 Mechanized Military Training Impacts on Hydraulic Characteristics of Ridgetop Soils in a Forested Watershed. 188

3.4 Stream Water Quality 203

3.4.1 Ecological Indicators in Forested Watersheds: Relationships between Watershed Characteristics and Stream Water Quality in Fort Benning, GA. 204

3.4.2 Microbial Nutrient Cycling in the Riparian Zone and its Influence on Stream Chemistry. 228

3.5 Modeling and Synthesis 246

3.5.1 Multivariate analysis of soil biogeochemical parameters for assessment of ecological condition. 247

3.5.2 Quantitative analysis of soil nutrient concentrations with near infrared spectroscopy and partial least squares regression. 277

3.6 Summary 303

4.0 Project Milestones (FY2003) 339

5.0 Close-Out Plan 339

6.0 Publications, Theses, Dissertations, and Presentations 340

6.1 Publications 340

6.2 Theses and Dissertations 342

6.3 Presentations 342

EXECUTIVE SUMMARY

Introduction:

The goal of this research is to develop indicators of ecosystem integrity and impending ecological change that include natural variation and human disturbance. We are evaluating parameters related to properties and processes in the understory vegetation, soil and surface hydrology as potentially sensitive indicators of ecosystem integrity and ecological response to natural and anthropogenic factors. The basic premise is that soil serves as the central ecosystem component that links the quality of the terrestrial habitats (by influencing vegetation and its stability) and the aquatic habitats (via control of soil erosion and overland runoff). We have evaluated potential ecological indicators for sensitivity, selectivity, and ease of measurement. Indicator selection was based on those that 1) show a high correlation with ecosystem state, 2) provide early warning of impending change and 3) differentiate between natural ecological variation and anthropogenic negative impacts. In addition, we have attempted to determine the range of natural variation for indicator variables, and compared those with the range of values under anthropogenic, especially mission-related, influences. Our research and monitoring plan addresses the following five tasks:

- Task 1 Soil/sediment quality indicators: Identification of physical, chemical and biological variables of soil that may be used as indicators of ecological change. Section 3.1
- Task 2 Vegetation indicators: Identification of species and community variables of vegetation that may be used as indicators of ecological change. Section 3.2
- Task 3 Hydrology: Identification of aspects of surface hydrology that may be used as indicators of ecological change. Section 3.3
- Task 4 Stream Water quality: Correlation of watershed hydrology and soil biogeochemistry in order to identify natural and anthropogenic influences on water quality. Section 3.4
- Task 5 Synthesis and modeling. Section 3.5

Findings and Accomplishments

Severe impacts to soil, vegetation and hydrologic processes are associated with mechanized training involving tracked (tanks and Bradley) vehicles. Moderate to severe impacts also occur in several areas of non-military land use, primarily due to forest clear-cutting activities. Hydrologic and ecological impacts observed in wetlands and streams downslope from clear-cut upland areas were similar in nature to those observed in association with severe military disturbance; however, since silvicultural activities are typically shorter duration, the extent and severity of these disturbances are less and recovery more rapid than those associated with mechanized military activity. The soil, vegetation and hydrologic parameters (potential indicators) that were most closely correlated with pre-determined site disturbance levels (low, moderate, severe) were those that reflected loss of vegetation biomass and community structure, disruption and/or compaction of soil, and loss of soil A horizon (and soil organic matter) in uplands; and accelerated sedimentation of clay and sand in wetlands. In wetland areas downslope from impacted uplands, relationships between soil biogeochemical indicators and upland impacts were less clearly defined. However, indicators that directly related to wetland soil organic matter content (and "dilution" by clay or sand) were useful in identifying sediment-impacted wetlands located below severely-disturbed upland areas. The potential value of wetland soil biogeochemical properties as indicators of nutrient loading in uplands (e.g., from excessive fertilization or waste disposal) was not realized at the Fort Benning study areas, due to the nature of the ecological impacts in upland areas.

Commonly observed impacts of mechanized training on soil and vegetation included:

- Disturbance or destruction of vegetation communities, including ground cover (especially litter cover), understory and canopy vegetation.
- Disruption of soil A horizon and effective burial or dilution of biologically-active topsoil with organic-poor lower horizons.
- Compaction of subsoil, reducing soil permeability and increasing runoff and erosion potential.
- Loss of A and E horizons in severely-impacted upland areas, rendering soil unsuitable for supporting native plant communities.
- Gulley erosion in downslope areas, with significant sedimentation in wetlands and streams.
- Short-circuiting of watershed flow paths with increased surface runoff and decreased subsurface detention in uplands (creating hydrologic and ecological imbalances in wetlands and streams).

Summary of Accomplishments
Soil Biogeochemistry

The most promising soil biogeochemical indicators for upland areas were highly correlated with soil organic matter content and carbon (C) quality (biodegradability):

- **Total organic C - indicator of soil disturbance resulting from loss of topsoil (erosion) or mixing of A and E horizons.**
    - Anthropogenic impacts on soil and ground cover in upland areas of the Fort Benning study site included (1) disturbance or destruction of vegetation, resulting in increased area of bare ground and a greater proportion of early successional species, (2) disruption of soil A horizon and effective burial or dilution of biologically-active topsoil with organic-poor lower horizons, (3) increased erosion in uplands and deposition of sediment in bottomland areas, and (4) loss of soil A horizon in severely-impacted upland areas. Impacts to bottomland soils were primarily associated with soil disturbance in adjacent upland areas, and typically involved accelerated deposition of clay and silt (moderately-impacted areas) or sand (severely-impacted areas). The primary impact of increased sedimentation, with regard to soil C and N dynamics, was dilution and/or burial of organic matter contained in the native wetland soils. For both upland and bottomland sites, the observed decrease in soil TC and TN with increasing level of impact was indicative of the reduction in soil organic matter content of surface horizons. (**Section 3.1.1**: DeBusk, W. F., B. L. Skulnick, J. P. Prenger, and K. R. Reddy. Response of Soil Organic Carbon Dynamics to Disturbance by Military Training in the Southeastern U.S.).
- **Microbial biomass (as C) - indicator of the size of the labile (readily bioavailable) soil C pool.**

- Microbial biomass C and soil respiration showed a significant decrease with increasing site impact, consistent with the trend observed for TC. However, changes in MBC with impact level were not directly proportional to changes in TC, as demonstrated by the significant increase in MBC:TC with site impact. (**Section 3.1.1**: DeBusk, W. F., B. L. Skulnick, J. P. Prenger, and K. R. Reddy. Response of Soil Organic Carbon Dynamics to Disturbance by Military Training in the Southeastern U.S.).

- **Soil (microbial) respiration - indicator of the amount of bioavailable soil C.**
  - Soil respiration rate was roughly correlated with TC concentration, as would be expected since organic C provides the metabolic substrate for soil microorganisms. Since soil respiration was determined by laboratory incubation of soil samples at a constant temperature, the measured rates represented (1) primarily microbial respiration rather than root respiration, and (2) potential respiration rates rather than actual *in situ* rates at the time of sampling. Therefore, soil respiration reported in this study were an indicatory of the size of the bioavailable pool of soil C. (**Section 3.1.1**: DeBusk, W. F., B. L. Skulnick, J. P. Prenger, and K. R. Reddy. Response of Soil Organic Carbon Dynamics to Disturbance by Military Training in the Southeastern U.S.).
- **Ratios of microbial biomass to organic C and respiration to biomass - relative bioavailability of the soil organic C pool.**
  - Metabolic quotient (qCO_2), or specific respiration rate (normalized to MBC), showed a significant decrease with increasing level of impact. In our study, it was apparent that decreasing qCO_2 with increasing site impact was related to substrate bioavailability, and was not a response to environmental (external) stress. Although the biochemical processes governing the relationship between qCO_2 and soil impact or condition are not known with any certainty, our study results suggest that this parameter may be a useful indicator of ecological condition or change, primarily for upland areas. The ratio of microbial biomass C to soil organic C, a.k.a. microbial quotient, has been related to soil C availability and the tendency for a soil to accumulate organic matter. Based on combined results of Phases 1 and 2 of this study, both DOC:TC and MBC:TC were found to be relatively good indicators of soil "quality" in upland areas, as related to site impacts or ecological condition. The potential value of the DOC:TC parameter as a robust indicator, i.e., beyond the scope of this study, for either soil quality or ecological condition cannot be determined solely from these results. The MBC:TC parameter, on the other hand, has been widely used as an indicator of bioavailability of soil organic C (Anderson and Domsch, 1989; Sparling, 1992). (**Section 3.1.1**: DeBusk, W. F., B. L. Skulnick, J. P. Prenger, and K. R. Reddy. Response of Soil Organic Carbon Dynamics to Disturbance by Military Training in the Southeastern U.S.).
- **Relative bioavailability of soil C was higher in disturbed areas due to depletion of older, more stable soil organic matter.**
  - The response of qCO_2 to soil disturbance was consistent with the responses of DOC:TC and MBC:TC, all of which suggest that resource (organic C) quality *increased* with soil disturbance, i.e. there was a lower proportion of recalcitrant soil organic matter, even as total soil C storage *decreased* with increasing

disturbance. (**Section 3.1.1**: DeBusk, W. F., B. L. Skulnick, J. P. Prenger, and K. R. Reddy. Response of Soil Organic Carbon Dynamics to Disturbance by Military Training in the Southeastern U.S.).

- **Beta-glucosidase activity - indicator of the amount of bioavailable soil C.**
    - β-glucosidase did distinguish the three levels of impact in bottomland transects, perhaps indicating a higher ratio of available carbon to TC at intermediate levels of disturbance. Separation of moderate from low and severe impacts by β-glucosidase was less effective in upland soils. (**Section 3.1.2**: Prenger, J. P., W. F. DeBusk, and K. R. Reddy. 2004. Influence of military land management on extracellular soil enzymes.)
- **Methanotrophic bacterial communities differ in highly impacted bottomlands.**
    - Terminal restriction fragment length polymorphism (T-RFLP) analysis of *pmoA* genes was applied to samples taken from transects located in upland and bottomland sites within the two watersheds. Principal components analysis (PCA) revealed that T-RFLPs from upland and for the most part bottomland samples clustered together in both watersheds. However, some Bonham Creek bottomland T-RFLPs clustered within the upland cluster, suggesting mixing of upland with bottomland soils. (**Section 3.1.3:** Ogram, A., H. Castro, E.A. Stanley, W. Chen, and J. Prenger. Distribution of Methanotrophs in Managed and Highly Degraded Watersheds)

- **Depth (thickness) of the A horizon - indicator of soil disturbance resulting from loss of topsoil (erosion) or mixing of A and E horizons.**
    - A-horizon depths decreased with increased level of disturbance category: bottomland sand-loam, Low to Medium; upland clay, Medium to High; upland sand, Low to Medium to High. (**Part of integration effort, not included in this report. This portion to be published as: Krzysik, A., et al. 2004.** Soil A-Horizon Depth as a Reliable, Integrative, and Ecosystem Relevant Ecological Indicator of Landscape Disturbance. To be submitted to Environmental Management)

Vegetation

Vegetative indicators that most accurately reflected the impacts of military training were:

- **Percent cover of herbaceous vegetation (ground cover, and litter cover), or in cases of more severe impacts, canopy cover.**
    - Woody plants did not differentiate well among the disturbance levels; however, there was a trend of decreased overstory canopy cover with increased disturbance. Herbaceous vegetation composition on severely-disturbed sites segregated from low and medium disturbances but no segregation was found between the two lower levels of disturbance. Chronic, landscape-scale disturbances have resulted in a very resilient flora. Coverage of bare ground and plant litter may best serve as indicators of disturbance. (**Section 3.2.1:** Tanner, G.W., D.L. Miller, and J. Archer. 2004. Vegetative indicators of disturbance in a chronically-disturbed ecosystem: Ft. Benning Military Reservation, Georgia.).
- **Plant species present only in severely disturbed sites identify the highest degree of disturbance.**

- Relative cover of *Rubus* sp. and *Rhus copallina* may be an important indicator of a shift from moderate to severe conditions. These two species are prolific seed producers, enhancing their ability to colonize disturbed sites, and they appear to withstand physical disturbance once established. Those herbaceous species most closely associated with severely disturbed sites were: *Digitaria ciliaris, Diodia teres, Stylosanthes biflora, Aristida purpurescens, Opuntia humifusa, Haplopappus dirasicatus*, and *Paspalum notatum*. Solid stands of *Paspalum notatum*, an exotic species of grass, occurred on sites that had been totally denuded in the past, and probably was planted to reduce erosion. (**Section 3.2.1:** Tanner, G.W., D.L. Miller, and J. Archer. 2004. Vegetative indicators of disturbance in a chronically-disturbed ecosystem: Ft. Benning Military Reservation, Georgia.).

- **Plant species indicating various stages of recovery from severe disturbance were identified that may be useful in tracking the progress of restoration efforts in highly-impacted areas.**
  - Herbaceous species composition and cover varied more with stand age than understory woody species. (Archer, J and D.L. Miller. 2004. Understory Vegetation and Soil response to silvicultural activity in a southeastern mixed pine forest: A chronosequence study. In Review. Submitted to Journal of Forest Ecology and Management).

  - Species richness did not differ among age classes for either woody or herbaceous species, while species distribution and abundance did. *Bulbostylis barbata* and *Pityopsis* spp were identified as indicators of younger sites (more recently disturbed). *Andropogon* spp., *Dichanthelium* spp., and *Aristida* spp. have all been found to be more abundant soon after a disturbance, followed by a slow decrease in frequency and abundance over. *Schizachyrium scoparium* and *Andropogon ternarius* were associated with 30-80 yr sites. *Schizachyrium scoparium* is considered a late successional plant throughout it range. While *S. scoparium* and *A. ternarius* occurred in all age classes, both increased with recovery time and had higher frequency and cover values on the oldest sites. (**Section 3.2.2:** Archer, J and D.L. Miller. 2004. Understory Vegetation and Soil response to silvicultural activity in a southeastern mixed pine forest: A chronosequence study.).

Indicators related to vegetation community composition in moderately or less impacted sites are often confounded by residual effects of prior soil disturbance related to agricultural land uses. Plant species potentially sensitive to low to moderate levels of disturbance probably have been extirpated from the sites due to historic levels of chronic disturbances. Indicator species to assess ecological condition may require an evaluation of "natural" or reference conditions prior to their use.

Hydrology

Hydrologic indicators are of significant value for analysis of disturbance or recovery on a watershed scale.

- **Correlation and regression analyses were performed to determine relationships among the watershed physical characteristics and the storm-based hydrologic indices.**
    - A number of significant relationships were found. The correlation results show that the increase in road density increased the variability in the peak discharges and the slopes of the rising limb. The increase in the military land increased the time of rise as well as the variability in the time base. The number of roads crossing streams is positively correlated with the response lag, whereas it is negatively correlated with the time base and the variability in the slopes of the falling limb. Increase in the bare land and the disturbance index increased the time of rise as well as the variability in the time base. Stepwise multiple correlations identified the relationships between the event indices and the management related watershed physical characteristics that are susceptible to the disturbances. Military land, road density, and the number of roads crossing streams predicted storm-based baseflow index, bankfull discharge, response lag, and time of rise well. (**Section 3.3.2:** Shirish Bhat, Jennifer M. Jacobs, Kirk Hatfield, and Wendy D. Graham. Hydrologic Indices of Watershed Scale Military Impacts in Fort Benning, GA.)
- **Analysis of hydrographs clearly reflect hydrologic imbalances resulting from soil and vegetation disturbance in uplands.**
    - In support of the finding that uplands in non-impacted areas do not contribute to the stream hydrograph, the contributing areas calculated by the stream hydrograph volumes and depth of rainfall events is less than the riparian/wetland area, suggesting that no area outside of the wetland/riparian area contribute to the stream hydrographs. In training areas, the K_{sat} is sufficiently low that overland flow could occur. Time of concentration for a 10cm/hr storm event was about 10 minutes. It is apparent that overland flow has gouged out deep gullies and transported sediment from the hilltops. The flow processes in these areas are observed to be different than those in less-impacted watersheds. Overland flow is conceived to usher water toward roads that channel the water directly to streams, thus by-passing or short-circuiting the natural watershed flow paths. (**Section 3.3.3:** D.B. Perkins, J.W. Jawitz, N.W. Haws, P.S.C. Rao. Impacts on Soil Saturated Hydraulic Conductivity: Mechanized Military Training Characterization Using Five Measurement Methods.)
- **Soil physical parameters (bulk density, porosity, texture, grain-size distribution, and saturated hydraulic conductivity) are potentially useful at small spatial scale.**
    - Smaller scaling factors imply smaller mean pore sizes of the training soils compared to the non-training soils. The higher soil bulk density values and lower infiltration rates of the training versus non-training areas are indications of the loss of organic matter combined with compaction from repeated tank track. The mean steady-state infiltration rate of the training sites (12.0 cm/hr) is less than half that of the non-training sites (26.8 cm/hr), but it is still greater than the maximum 100-yr, 24-hr rainfall intensity of 10 cm/yr. (**Section 3.3.4:** D.B. Perkins, N. W. Haws, B. S. Das, P.S.C. Rao, and J.W. Jawitz. 2004. Mechanized Military Training Impacts on Hydraulic Characteristics of Ridgetop Soils in a Forested Watershed.)

Stream Water quality

- **Stream TOC and TKN concentration decreased with increasing soil and vegetation disturbance (proportion of bare ground) in the watershed, reflecting depletion of soil organic matter and detritus in uplands and reduced leaching in soils due to short-circuited flow paths (gulleys) from uplands to streams.**
  - 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. (**Section 3.4.1:** Bhat, S., J.M. Jacobs, K. Hatfield, and J. Prenger. 2004. Ecological Indicators in Forested Watersheds: Relationships between Watershed Characteristics and Stream Water Quality in Fort Benning, GA.)**.**
- **Enzyme activities relative to patterns of biogeochemistry and soil water content in riparian wetlands varied with distance from stream edge and help explain temporal patterns of groundwater Total Kjeldahl Nitrogen (TKN) related to leaf fall and canopy loss in riparian forests.**
  - Riparian soils were sampled at approximately 80 meter intervals along two streams and in three transects normal to stream flow. Stream and groundwater water chemistry were monitored monthly in transects normal to stream flow in one second order watershed. Variability in microbial enzyme activities and soil total nitrogen (TN) were most closely associated to soil water content, while groundwater Total Kjeldahl Nitrogen (TKN) showed temporal patterns related to leaf fall and canopy loss in riparian forests and varied with distance from stream edge. Patterns of peptidase activity were complex, with minima observed at approximately 30% soil moisture content. (**Section 3.4.2:** Prenger, J.P., Bhat, S., J.M. Jacobs, and K. R. Reddy. Microbial Nutrient Cycling in the Riparian Zone and its Influence on Stream Chemistry.)

Modeling and Synthesis

- **Multivariate statistical analyses were applied to 20 biogeochemical parameters in order to discriminate samples based on landscape position, vegetation type, watershed of origin and disturbance class.**
  - Principal components analysis identified that the total organic matter present in the soil samples (measured as total carbon, total nitrogen, and total phosphorous) was the dominant contributor of variability between the soil samples. Canonical Discriminant Analysis showed that canonical variables could be successfully used to discriminate samples based on landscape position, vegetation type, watershed of origin and disturbance class. Logistic regression was used to predict the

probability of a specific site being disturbed or non-disturbed based on the observed categorical variables and measured biogeochemical variables that were found to effect disturbance. (**Section 3.5.1:** Dabral, S., W. D. Graham, J.P. Prenger, and W. F. DeBusk. Multivariate analysis of soil biogeochemical parameters for assessment of ecological condition.)

- **Near Infrared Reflectance Spectroscopy (NIRS) for soil analysis is rapid, low-cost technique for determination of several individual soil biogeochemical properties and direct evaluation of derived soil quality metrics or indices.**
    - Reflectance measurements and 20 soil biogeochemical variables measured on over 550 soil samples were used to develop a robust PLS model for independently predicting TC, TN, and TP of new observations based on the reflectance measurements. The results presented indicate that near-infrared spectroscopy coupled with partial least squares can be a useful and inexpensive alternative to expensive and time consuming lab analyses. (**Section 3.5.2**: Dabral, S., W. D. Graham, and J.P. Prenger. 2004. Quantitative analysis of soil nutrient concentrations with near infrared spectroscopy and partial least squares regression.).

General Conclusions:

1. Approximately 2-15% of throughfall shows up as stream flow. Median value is approximately 6%. Time to peak discharge is approximately 3 hours.
2. Storm intensities are usually $<K_{sat}$ at most places, except severely disturbed areas.
3. Soil cover plays an important role in determining the potential runoff and may be more important than K_{sat} of surface soil.
4. Biogeochemical cycling in soils and vegetation are influenced by soil-water content.
5. Soil organic matter and several biogeochemical properties associated with C cycling are important biogeochemical indicators.
6. Spectral analysis shows excellent promise to determine soil nutrient status.
7. Understory vegetation species composition correlates with disturbance. Clear indicators generally observed only at heavily impacted sites.
8. Nutrient and sediment loads in “low” and “medium” impact sites are not too large. Sediment may be the most important water quality attribute for “severe” impact sites.
9. Water quality measurements revealed low levels of most nutrients.
10. Decreased canopy cover in wetlands and hardwood communities of impacted areas increase the nutrient load to streams.
11. Riparian zones play an important role in determining water quality.
12. Multivariate Analysis, Principal Component Analysis, and Canonical Correspondence Analysis yielded combinations of factors that are useful in identifying impacts.

1.0 INTRODUCTION

Our research sought to develop suitable indicators of ecosystem integrity and impending ecological change resulting from both natural variation and anthropogenic activities. We have used a multidisciplinary and multi-scale approach, which produced a number of potential techniques for ecosystem monitoring and evaluation. Results of the study will enhance the ability to minimize, mitigate or remove major negative environmental impacts on DoD's ability to conduct the military mission. We have addressed the SEMP objective of identifying indicators that signal ecological change in intensively and/or lightly used ecological systems on military installations in an attempt to provide early indications of change associated with (1) natural ecosystem variability and (2) military activities, including training and testing, as well as other land management practices. Early indications of change, and an understanding of the likely causes, will improve installation managers' ability to manage activities that are shown to be damaging, and prevent long-term, negative effects.

The concept of ecosystem integrity, or "health", in the context of the military installation, encompasses not only the sustainability of the "natural" biota in the system, but also the sustainability of human activities at the installation, namely the military mission. Thus, changes in ecological condition are of great concern to both resource managers and military trainers. A suite of variables is needed to measure changes in ecological condition. Two types of indicators that may be useful are (1) variables that inform managers about ecosystem status and (2) variables that signal impending change. In many cases, these indicators may be the same. Both types are needed, but variables that serve as early warnings of impending changes outside the natural range of variation, and variables that are shown to be related to activities affecting the military mission, may be especially valuable.

2.0 TECHNICAL OBJECTIVES

We have evaluated a suite of parameters related to properties and processes in the understory vegetation, soil and surface hydrology as potentially sensitive indicators of ecosystem integrity and ecological response to natural and anthropogenic factors. In general, the soil hydrologic and biogeochemical parameters examined relate to changes in soil physical and chemical characteristics, and the response of soil microbial population and plant communities. We have attempted to identify cause and effect relationships between environmental changes, due to both natural variability and anthropogenic perturbation, and soil and vegetation responses, primarily as they relate to nutrient storage, nutrient turnover and population dynamics.

Our basic premise is that soil serves as the central ecosystem component that links the quality of the terrestrial habitats (by influencing the vegetation and its stability) and the aquatic habitats (via control of soil erosion and overland runoff). Thus, a careful study of soil parameters and processes and linking them to impacts on terrestrial/aquatic habitats was the basis for our experimental approach. Furthermore, we have attempted to establish a sound scientific basis for the empirical parameters that might be used as ecological indicators.

Our research and monitoring plan has address the following objectives:

- Identify physical, chemical and biological variables (properties and processes) associated with soil, surface hydrology and vegetation that may be used as indicators of ecological change.

- Evaluate potential ecological indicators based on sensitivity, selectivity, ease of measurement and cost effectiveness.
- Select indicators that most effectively 1) show a high correlation with a certain state in a specific ecosystem, 2) provide early warning of impending change and 3) differentiate between natural ecological variation and anthropogenic negative impacts.
- Determine the likely range of natural variation for indicator variables, and compare with the range of values under anthropogenic, especially mission-related, influences.

3.0 RESULTS

3.1 Soil Biogeochemistry

Results Soil Biogeochemical studies have been summarized in several manuscripts. These have been organized into subsections preceded by an abstract. The most promising soil biogeochemical indicators for upland areas were highly correlated with soil organic matter content and carbon (C) quality (biodegradability). These include:

- **3.1.1 Total organic C - indicator of soil disturbance resulting from loss of topsoil (erosion) or mixing of A and E horizons.**
    - **Microbial biomass (as C) - indicator of the size of the labile (readily bioavailable) soil C pool.**
    - **Soil (microbial) respiration - indicator of the amount of bioavailable soil C.**
    - **Ratios of microbial biomass to organic C and respiration to biomass - relative bioavailability of the soil organic C pool.**
    - **Relative bioavailability of soil C was higher in disturbed areas due to depletion of older, more stable soil organic matter.**
- **3.1.2 Beta-glucosidase activity - indicator of the amount of bioavailable soil C.**
- **3.1.3 Methanotrophic bacterial communities differ in highly impacted bottomlands.**
- **Depth (thickness) of the A horizon - indicator of soil disturbance resulting from loss of topsoil (erosion) or mixing of A and E horizons.**

3.1.1
Response of Soil Organic Carbon Dynamics to Disturbance by Military Training in the Southeastern U.S. DeBusk, W. F., B. L. Skulnick, J. P. Prenger, and K. R. Reddy.

ABSTRACT

A field study was conducted at the Fort Benning (Georgia, USA) military installation to evaluate changes in soil C storage and cycling in response to soil and vegetation disturbance associated with military training and natural resource management activities. Soil characterization was performed in two phases: Phase 1, a synoptic survey across the reservation to relate soil properties to predetermined levels of site impact (low, moderate or severe), and to landscape position (uplands vs. bottomlands), and Phase 2, a follow-up evaluation of soil properties and corresponding impact level, targeting sites with similar soil characteristics, i.e., grouped within the same upland or bottomland soil series. Phase 1 samples were analyzed for pH, total C (TC), total N (TN), dissolved organic C (DOC), and microbial biomass C (MBC). Phase 2 samples were analyzed for TC, TN, DOC, MBC, and soil respiration rate. Mean values for TC, TN, DOC, and MBC for Phase 1 samples were (for bottomland and upland sites, respectively) 38.3 and 10.9 g TC kg^{-1}, 2.03 and 0.42 g TN kg^{-1}, 121.3 and 65.5 mg DOC kg^{-1}, and 561 and 219 mg MBC kg^{-1}. Response of measured and derived parameters to level of site impact were similar for Phase 1 and Phase 2 of the study. Soil TC, TN, and MBC concentration and soil respiration rate decreased with increasing level of impact in both upland and bottomland sites. For upland areas, these trends reflected a loss of soil organic matter due to increased physical disturbance of soils and vegetation, and in areas of severe impact, loss of topsoil via erosion. Impacts in bottomlands were primarily related to sedimentation from disturbed areas in adjacent uplands, and were manifest as “dilution” of native soil organic matter, due to the deposition of mineral sediments (clay, silt or sand). The derived parameters DOC:TC and MBC:TC (a.k.a. microbial quotient) showed significant increases with increasing site impact. These parameters, especially MBC:TC, have been associated with soil C quality or bioavailability. Metabolic quotient qCO_2, or MBC-specific respiration rate, decreased significantly with increasing impact, indicating increased efficiency of microbial assimilation of organic C. These trends suggest that resource (organic C) quality *increased* with soil disturbance, i.e. there was a lower proportion of recalcitrant soil organic matter, even as total soil C storage *decreased* with increasing disturbance. Based on our study results, DOC:TC, MBC:TC, and qCO_2 were found to be relatively good indicators of soil quality in upland areas, as related to site impacts or ecological condition.

INTRODUCTION

Experimental data relating the impact of military training to soil biogeochemical characteristics is relatively scarce. While some work has been done on recovery of soils from encampments (Kade and Warren, 2002) and troop and mechanized training (Webb, 2002), these studies have involved desert ecosystems and focused primarily on alteration of soil physical properties. For example, a controlled study of mechanized military training activity in chalk grasslands (Hirst et al., 2003) showed significant changes in soil compaction but did not report changes in soil chemistry. Small but significant increases in soil bulk density and significant decreases in infiltration rate were associated with foot traffic from military training in Colorado (Whitecotton, et al. 2000). A recent study at Ft. Benning, Georgia (USA) by Garten et al. (2003) found increased soil bulk density, lower soil carbon concentrations, and less carbon and nitrogen in particulate organic matter at moderate use, heavy use, and remediated sites relative to reference sites.

In recent years, various soil biogeochemical properties and indices have been proposed for use in the determination of soil quality, for agricultural and other land uses. Soil quality refers to the suitability, or “fitness” of a soil to perform various hydrologic, biotic and ecological functions, including support for plant and animal productivity and enhancement of water and air quality (Arshad and Martin, 2002). Among the biogeochemical parameters that have been identified as potential indicators of soil quality are total soil organic C, nutrient availability, microbial biomass, and soil respiration (Nortcliff, 2002).

The present study was conducted as part of a multi-agency collaboration to identify indicators of ecological condition and change in landscapes dominated by military training. The specific objectives of our study were to (1) characterize soils under predominantly military land use by measuring a suite of biogeochemical parameters associated with C storage and cycling, (2) measure responses of these parameters to various levels and types of physical site disturbance, and (3) evaluate the potential utility of the parameters as indicators of soil quality, site impact and/or ecological condition.

METHODS

<u>Site Description</u>

The study area was located on the Fort Benning military reservation, near the city of Columbus in west-central Georgia, USA (Figure 1). The installation covers 74,370 ha of land along the Chattahoochee River, with most of the land area located in Georgia, and the remainder located across the Chattahoochee River in Alabama. Fort Benning lies within the Southern Mixed Forest Province (Bailey, 1995) along the Fall Line, which represents the transition between the Gulf Coastal Plain and Piedmont regions. The topography of the area is characterized by nearly level to gently sloping ridgetops, moderately steep and steep hillsides, and nearly level valleys along stream channels and other tributaries. Upland soils are primarily well- to excessively-drained Ultisols and Entisols (e.g., Troup and Lakeland series) that primarily support longleaf pine sandhill communities and longleaf and loblolly pine plantations. Wetlands are generally restricted to riparian areas, where bottomland hardwood forests are common. Soils commonly associated with wetlands and other bottomland areas of Fort Benning include Entisols (Fluvaquents, e.g., Bibb series), Inceptisols (Humaquepts), and Ultisols (Aquults). Climate in the study area is warm, humid and temperate. Monthly mean temperatures range from 7.7 °C in January to 27.6 °C in July, and average annual rainfall is 1321 mm.

<u>Assessment of Site Condition</u>

Fort Benning is the primary training facility for the US Army Infantry. Training-related impacts to natural communities range from low to moderate disturbance of understory vegetation and soils caused by foot and light vehicle traffic to severe localized vegetation clearing, soil compaction and erosion associated with site-intensive mechanized training (e.g., tracked vehicles). Other commonly-occurring, but unrelated to military training, impacts to vegetation and soil include fire and forestry activities, such as cultivation, tree thinning and clear-cutting. Fire-related impacts to vegetation and soil are fairly consistent and ubiquitous, due to the relatively high frequency of prescribed burns (typically a 3-year cycle). Wildfires also occur periodically, often resulting from ordnance detonation and other military training activities.

Ideally, evaluation of the response of soil C storage and partitioning to anthropogenic disturbance would entail correlation of response variables with known levels of site disturbance. Unfortunately, documentation of military training activity within the study area is poor with respect to frequency and duration of site disturbance. Therefore, it was necessary to define criteria for determining the current extent of impact to each site, as a surrogate for actual disturbance history, and to assign an impact level or class to each site. A standard set of criteria was developed for the purpose of determining the level of impact, primarily based on visual inspection of vegetation and soil properties. Study sites were thus placed in one of three categories of ecological impact: low, moderate or severe. In general, low-impact sites represented areas of low military use or mature managed forest; moderate-impact sites represented areas of moderate military activity (no tracked vehicles) or early to mid-successional managed forest; and severe-impact sites were areas of heavy military training (e.g., tanks) or non-military areas experiencing recent clear-cuts with severe soil disturbance and/or erosion. The complete set of criteria related to overstory vegetation, understory vegetation, ground cover, soils, and fire are listed in Table 1.

Sampling Protocol

Phase 1 Sampling:

Soil sampling was conducted in two phases, during January – August 2000 (Phase 1) and December 2000 – June 2001 (Phase 2). Phase 1 sampling was conducted at a multi-watershed scale, and covered a significant portion of the Fort Benning reservation, to capture the edaphic variability associated with landscape position, vegetation, hydrology and anthropogenic disturbance, both military and non-military. Soil sampling sites for Phase 1 were distributed among 6 third and fourth order watersheds, and located based on transects orthogonal to the main stream within each watershed (Figure 1). Sites along each transect were stratified according to landscape position, such that approximately equal numbers of sites were selected for bottomlands (wetlands and riparian zone), hilltop and ridge sites, and intermediate mid-slope sites. The intent of the stratified sampling scheme was to account for a portion of the "natural" variability related to differences among community or ecosystem types associated with landscape position. There were no significant differences in mean values between mid-slope and hilltop/ridge sites for the parameters in this study; therefore, for the purposes of this study all upland sites were combined into a single category ("upland").

Soil samples were obtained at approximately 50 locations in each watershed, for a total of 300 sites. Each sample point consisted of a 1 m^2 square plot, within which 5 individual samples were taken in a diagonal pattern. The individual samples were then composited for analysis as a single sample. Soil was sampled to a depth of 20 cm, using a soil push probe with an inside diameter of 1 inch. Triplicate samples were taken for QA at approximately 20% of the sites. At the triplicated sites, the 2 additional sampling points were located to either side of the primary sample point, at a distance of 5 m from the center of the primary point. Each sampling site was assessed using criteria in Table 1, and classified according to the corresponding level of impact (site disturbance), i.e., low, moderate or severe.

Phase 2 Sampling:

Phase 2 sampling was conducted within sub-regions of the Phase 1 sampling domain, typically less than 0.25 km^2 in area. Each of these Phase 2 sites represented a single type of land use and condition (level of impact), e.g., severe impact military. In addition, landscape position (upland/ridge top and bottomland) and soil type were uniform within individual Phase 2 sampling sites. Upland sites were underlain by Troup loamy sands (loamy, kaolinitic, thermic Grossarenic Kandiudults), while bottomland (wetland) sites were associated with Bibb sandy loams (coarse-loamy, siliceous, active, acid, thermic Typic Fluvaquents). The Phase 2 sampling layout was intended to provide greater control over the "natural" variability encountered among Phase 1 sampling sites.

Sampling points within each upland site were distributed along a single transect ranging in length from 80 to 400 m, depending on the total area of the site. Each transect contained sampling points spaced at 20 m intervals, resulting in a range of 5 to 21 sampling points per transect. Multiple transects spanning the riparian zone (orthogonal to stream, typically 25 m in length), were established for sampling bottomland sites. Sampling protocol for Phase 2 sites was identical to that used for Phase 1 sites, except that bottomland soils were sampled to a depth of 5 cm, using a 6.5 cm diameter polycarbonate corer.

Analytical Methods

All soil samples were analyzed for total C (TC), total N (TN), dissolved organic C (DOC), microbial biomass C (MBC), and pH. In addition, soil respiration rate was determined for Phase 2 samples.

Soil pH was measured in a 1:1 (mass basis) soil-water slurry, using deionized water and field-moist soil. Total C and total N were measured by combustion using a Carlo-Erba NA-1500 CNS analyzer (Haak-Buchler Instruments, Saddlebrook, NJ). Prior to analysis, samples were oven-dried at 40°C and finely ground; approximately 20 to 40 mg of sample were used for analysis.

Dissolved organic C was determined by extracting approximately 6 g of field-moist soil with 30 mL of de-ionized water on a reciprocal shaker for 1 hour. Samples were then centrifuged and the supernatant decanted and filtered through a 0.45-µm membrane filter. The resulting solution was analyzed for total carbon using a Dorman DC-190 TOC analyzer (Rosemount Analytical Inc., Santa Clara, CA).

Microbial biomass C was determined using a modification of the chloroform fumigation-extraction procedure of Horwath and Paul (1994). Ethanol-free chloroform was added directly to approximately 5g of field-moist soil to enhance distribution of chloroform within the sample (Ocio and Brookes 1990) prior to incubation of samples under a chloroform atmosphere in a vacuum dessicator. After incubation in darkness for a 24 hour period, chloroform was removed from the samples by repeatedly purging the dessicator, then the labile organic C in the soil samples was extracted with 0.5M potassium sulfate. Extracted samples were centrifuged, supernatant decanted, and filtered through Whatman #41 filter paper to remove particulate material. The sample solution was analyzed for total carbon using a Dorman DC-190 TOC analyzer. A parallel set of samples was extracted with potassium sulfate in the same manner, without prior chloroform fumigation; MBC was calculated as the difference between labile organic C content of the fumigated and non-fumigated samples.

Soil microbial respiration was determined under controlled conditions in laboratory incubations, providing a standardized (among sites) measure of organic C turnover. For each sample, approximately 5 g of field-moist soil and 50 mL distilled water were placed in a 150 ml glass serum bottle and sealed with a butyl rubber septum. Samples were placed in an incubator in the dark at 25°C with continuous shaking. Headspace samples were taken every 12 hours for a period of seven days, and analyzed immediately for CO_2 using a Shimadzu gas chromatograph (GC) equipped with a Poropak N (Supelco, Bellefonte, PA) column and thermal-conductivity detector (TCD). Soil respiration rate was determined from linear increase in headspace CO_2 over time. In addition, the metabolic quotient (qCO_2), or specific respiration rate, was calculated as the ratio of soil microbial respiration to MBC concentration (Anderson and Domsch, 1990).

Statistical Analysis

Statistical analyses were performed using the JMP statistical package (SAS Institute, Cary, NC). Analysis of variance and mean comparisons were determined for $P \leq 0.05$. Data sets with non-normal distributions were transformed to normal distributions using a log transformation procedure, prior to statistical analysis.

RESULTS AND DISCUSSION

Phase 1 Sampling

Soil chemical properties for Phase 1 sites (n=301) are summarized in Table 2 by landscape position. In contrast, for all parameters, means for bottomland sites were significantly different from means for upland sites. Total C, DOC, MBC, and total N concentrations were all greater in bottomland soils than in upland soils, as expected, given the elevated soil organic matter content characteristic of wetlands and other riparian areas.

In general, Phase 1 soil chemistry data were consistent with the low-fertility and poorly-buffered soils of the region. Concentrations of TC and TN in the topsoil (surface 20 cm) reflected relatively low organic matter content for both uplands and bottomlands. Organic matter in the sandy upland soils was generally restricted to a shallow A horizon (typically about 5-10 cm thick for Troup soil series), although an organic (O) surface horizon was observed in a few minimally-disturbed upland areas. Bottomland soils were generally loamy, with widely varying organic matter content relative to surrounding upland soils (inter-decile range for total C of 7.0 – 100.6 g kg^{-1}). Soils at many of the bottomland sites would neither be considered organic nor wetland soils; no significant accumulation of muck or peat was found at any sampling site. Increases in soil DOC, MBC and total N concentrations between upland and bottomland sites were roughly proportional to the increase observed in TC, although mean C:N ratio differed significantly between upland and bottomland sites. The mean C:N ratio of 26.8 for upland sites reflected the low fertility of these soils; in contrast the mean C:N ratio for bottomland soils was somewhat lower at 18.9.

Military impact at Fort Benning is patchy; severe disturbance is concentrated in a few areas of the base, primarily along sandy ridge tops. Many areas experience minimal disturbance from military training, since they serve as buffer zones or are used only to accommodate training of infantry foot soldiers. Severe and/or chronic impacts to soil and vegetation were primarily associated with areas of mechanized training with tracked vehicles (e.g., tanks and Bradley troop carriers). A wide range of impacts was also observed in areas of non-military land use, primarily associated with clear-cutting or selective thinning in stands of *Pinus spp.* (primarily longleaf and loblolly pine) in uplands. These activities resulted in varying degrees of soil disturbance and erosion.

The nature of physical impacts to soils among upland sites ranged from partial to total loss of vegetative ground cover; minor sheet or rill erosion to severe gully erosion; and disturbance of O and A horizons by foot or vehicular traffic to complete loss of topsoil through mechanized training and erosion. Elevated soil bulk density and decreased infiltration rate have been documented in areas of moderate to high foot traffic associated with military training, at Fort Benning and elsewhere (Garten et al., 2003; Whitecotton et al., 2000). Based on site observations, it was apparent that moderate to severe disturbance (e.g., training exercises) tended to disrupt the integrity of the topsoil, primarily the A horizon, where ground cover had been reduced by disturbance or was naturally sparse. Furthermore, it is likely that ground cover impacts from training were magnified because of the loose structure and sandy texture of the A horizon.

Ecological impacts to bottomland sites were, evidently, more often the result of soil disturbance in adjacent upland areas, rather than on-site activities. Consequently, the primary result of military and non-military disturbance to bottomland soils was accelerated sedimentation from uplands. In most cases this occurred as deposition of silt and clay, but in bottomlands located downslope from highly-impacted uplands significant deposition of sand was observed. Severely-impacted bottomland sites were relatively uncommon in the Fort Benning study area and, in fact, none of the bottomlands selected for Phase 1 sampling were classified as such.

The general trend for soil TC and TN among upland and bottomland Phase 1 sites was a decrease in concentration with increasing level of impact (Figure 2). A significant decrease in TC and TN with increasing level of impact was observed for upland sites, while a similar decrease in TC and TN in bottomlands was not significant. Soil C:N ratio increased slightly, but significantly, with level of impact in bottomland sites. In upland areas, C:N ratio was significantly greater for moderate-impact sites than in low- or severe-impact sites. Microbial biomass C concentration showed a similar trend to TC and TN, decreasing significantly with increasing impact in both upland and bottomland sites. Dissolved organic C concentration did not vary significantly with level of impact in bottomland or upland areas.

From the standpoint of soil C storage and partitioning, significantly impacted upland areas at Fort Benning were suggestive of early successional landscapes. Singh et al. (2001) observed an increase in soil MBC with increasing site age (from 1 to 58 years) across a chronosequence of successional sites impacted by landslides. Cook and Allan (1992a,b) found that soil DOC at 5 old field successional sites, ranging from 12 to 62 years since cultivation, was relatively invariant and that the quality (biodegradability) of soil DOC did not differ among sites, based on minimal differences in the relative amount of DOC fractions. O'Brien et al. (2003) observed an increase in total C and microbial C, as well as water soluble C (DOC), with increasing stand age in managed *Eucalyptus regnans* and *Pinus radiata* forests in Australia. Data from studies of ecological succession may have limited applicability to the present study, however, since the patterns of disturbance observed at our sites were not always consistent with successional ecosystems. For example, numerous moderately-impacts sites were characterized by disturbance or loss of ground cover, with minimal impact to canopy vegetation.

In contrast to the impact-related trends observed for soil DOC and MBC content, the relative proportion of DOC and MBC in the total soil C pool (i.e., DOC:TC and MBC:TC) increased significantly with level of impact in upland sites. The ratio of microbial biomass C to soil organic C, a.k.a. microbial quotient, has been related to soil C availability and the tendency for a soil to accumulate organic matter (Anderson and Domsch, 1989; Sparling, 1992). In this study, the positive relationship between these parameters (MBC:TC and DOC:TC) and site impact may be related to a disproportionate loss of stable soil organic matter (i.e., humus) in the O and A horizons due to physical soil disturbance. In addition to wholesale loss of soil organic matter in surface soils (decreasing TC and TC concentration) via erosion or mixing of soil horizons, it is likely that chronic physical disturbance of topsoil may stimulate decomposition of the more recalcitrant soil organic matter fractions. For example, tillage of agricultural soils favors breakdown of soil organic matter through increased aeration and breakup of soil aggregates with exposure of previously inaccessible organic matter to microbial attack (Haynes, 1999). Similarly, Neff and Asner (2001) suggested that disruption of soil aggregates through physical disturbance can indirectly increase DOC release by increasing the surface area of the aggregates, and Burford and Bremner (1975) found a high correlation between water soluble organic C and mineralizable C.

Among the parameters investigated during Phase 1 of this study, soil TC, TN, DOC and MBC all exhibited a similar response to site disturbance, consistent with changes in soil organic matter content. However, only TN and DOC:TC varied significantly and consistently among all levels of site impact in upland sites, and only MBC and C:N ratio showed significant responses to impact level in bottomland sites. The potential suitability of a response variable as an indicator of ecological condition, e.g. for gauging the extent of site degradation or recovery, depends to a great extent upon the degree of resolution it provides at the low-impact end of the site disturbance spectrum. The ability to discriminate among low to moderate levels of site impact or stress, for which more obvious signs of disturbance are absent, would be of particular value for early detection of impending significant ecological change.

<u>Phase 2 Sampling</u>

For Phase 2 sampling, multiple points were sampled at each of the 8 upland and 5 bottomland sites, thus, in contrast to the Phase 1 data set, within- and among-site variability for each parameter was known. In addition to landscape position (upland or bottomland) and level of site impact, the Phase 2 sites were designated as either predominantly military- or forestry-impacted (Table 3), based on information provided by resource management personnel as well as our own field observations. Analysis of variance indicated that among-site differences for parameters in Table 3 were related to both the type and level of site impact for bottomland sites, but were primarily a function of impact level in uplands.

For some parameters, in particular DOC, MBC and respiration, significant differences were found between sites where the *a priori* designation of both type and level of impact were the same. Obviously, any number of observed or unknown site characteristics may account for observed differences in soil chemistry among similar sites. However, it is highly probable that prior land use and management exerted a significant influence on present-day soil properties, particularly those related to soil organic matter storage and turnover. Although historical land use and disturbance regime are poorly documented for specific sites within the study area, it is known that much of the region now occupied by Fort Benning was under cultivation for a substantial period of time prior to the World War II era. Significant long-term (up to 100 years) effects of prior cultivation on soil organic matter properties, including decreased C content and lower C:N ratio, have been documented for forest soils (Compton and Boone, 2000; Compton et al., 1998).

Mean TC and TN concentration in bottomland soils were roughly 3 times higher for Phase 2 sites than for the more comprehensive Phase 1 data set. This discrepancy resulted from the fact that the comprehensive set of bottomland sites sampled in Phase 1 included many infrequently flooded wetlands and riparian zones of intermittent streams, with characteristically low accumulation of soil organic matter. The Phase 2 bottomland sites, on the other hand, were more representative of true wetlands, characterized by hydrophytic vegetation and flooded or saturated soils.

Aside from the difference in magnitude of TC and TN between Phase 1 and 2 sampling events, the general trends of C and N related parameters with respect to site impact were similar for Phase 1 and 2 (Figure 3). Total C showed an overall decrease with increasing level of impact, in both bottomland and upland areas, although mean TC concentrations for low and moderate levels of impact were not significantly different. Total N concentration followed a similar trend to TC in bottomlands; however, for upland areas the highest TN values were associated with moderate site impact rather than low site impact. Soil C:N ratio was significantly higher in severely-impacted bottomland sites than in low to moderate impact bottomlands. In contrast, C:N ratio decreased significantly with increasing level of site impact in upland areas. Dissolved organic C and MBC decreased significantly with increasing impact in bottomlands, but were highest at the moderate level of impact in uplands. As observed for Phase 1 sites, DOC:TC and MBC:TC increased significantly with level of impact in upland sites and, in contrast to Phase 1 results, showed a significant increase with site impact for bottomland areas.

Soil respiration decreased significantly with increasing level of impact for both bottomland and upland sites (Figure 3), although differences between low- and moderate- impact sites were not significant. Since soil respiration was determined by laboratory incubation of soil samples at a constant temperature, the measured rates represented (1) primarily microbial respiration rather than root respiration, and (2) potential respiration rates rather than actual *in situ* rates at the time of sampling. Although this approach does not capture the true magnitude (i.e., CO_2 evolved per unit area) or temporal variability of soil respiration under field conditions, it is a suitable method for inter-site comparison of soil microbial activity.

Soil respiration rate was roughly correlated with TC concentration, as would be expected since organic C provides the metabolic substrate for soil microorganisms. However, metabolic quotient (qCO_2), or specific respiration rate (normalized to MBC), showed a significant decrease with increasing level of impact (Figure 3). Conceptually, qCO_2 is a measure of microbial efficiency in the conversion of substrate C to MBC, and elevated qCO_2 is often viewed as an indicator of increased environmental stress or disturbance (Wardle and Ghani, 1995). Increased qCO_2 has also been associated with early successional ecosystems, under the premise that microbial efficiency is lower in these high resource systems with open cycling (Insam and Haselwandter, 1989; Wardle, 1993). However, Wardle and Ghani (1995) concluded that qCO_2 response is not always predictable, citing, for example, experimental evidence that this parameter may increase during the course of ecological succession. In our study, it is probable that decreasing qCO_2 with increasing site impact was related to substrate bioavailability and quality, and was not a response to environmental (external) stress. Accordingly, the response of qCO_2 to soil disturbance was consistent with the responses of DOC:TC and MBC:TC, all of which suggest that resource (organic C) quality *increased* with soil disturbance, i.e. there was a lower proportion of recalcitrant soil organic matter, even as total soil C storage *decreased* with increasing disturbance. This may be viewed as a corollary to the earlier observation that chronic soil disturbance may have created a more favorable environment, e.g., increased soil oxygenation, for decomposition of the more stable organic matter fractions, and therefore served to reduce, rather than increase, environmental stress for soil micoorganisms.

CONCLUSIONS

Anthropogenic impacts on soil and ground cover in upland areas of the Fort Benning study site included (1) disturbance or destruction of vegetation, resulting in increased area of bare ground and a greater proportion of early successional species, (2) disruption of soil A horizon and effective burial or dilution of biologically-active topsoil with organic-poor lower horizons, (3) increased erosion in uplands and deposition of sediment in bottomland areas, and (4) loss of soil A horizon in severely-impacted upland areas. Impacts to bottomland soils were primarily associated with soil disturbance in adjacent upland areas, and typically involved accelerated deposition of clay and silt (moderately-impacted areas) or sand (severely-impacted areas). The primary impact of increased sedimentation, with regard to soil C and N dynamics, was dilution and/or burial of organic matter contained in the native wetland soils.

For both upland and bottomland sites, the observed decrease in soil TC and TN with increasing level of impact (Figs. 2 and 3) was indicative of the reduction in soil organic matter content of surface horizons. While DOC responded inconsistently to site impact, microbial biomass C and soil respiration showed a significant decrease with increasing site impact, consistent with the trend observed for TC. However, changes in MBC with impact level were not directly proportional to changes in TC, as demonstrated by the significant increase in MBC:TC with site impact. The response of MBC:TC to site impact, and concomitant increase in DOC:TC, suggested an increase in the proportion of bioavailable C in the soil organic matter pool. This may have occurred via a simultaneous increase in decomposition of resistant soil organic matter fractions due to physical soil disturbance and increased fragmentation of newly-deposited plant litter. The decrease in qCO_2 with increasing disturbance is consistent with this scenario, where microbial efficiency of C assimilation increases as the proportion of recalcitrant soil organic matter fractions decreases (increased site impact).

The response of soil TC and related parameters to site impact level was similar for both Phase 1 and Phase 2 sites. Phase 2 sites were selected with the intent of minimizing variability resulting from differences in soil type and vegetation associations, thus clarifying the relationships between site impact and response variables. Nevertheless, Phase 2 results provided little additional insight into the relationship between site impact and TC, TN, DOC or MBC. This may have been an artifact of a biased site selection procedure (not representative of the total population) and small sample size, or the result of inaccurate determination of site impact, especially for non-military sites. On the other hand, the response of the derived variables MBC:TC and DOC:TC to site impact was more clearly illustrated by Phase 2 results. Similarly, a well-defined response was observed for metabolic quotient, qCO_2, particularly for upland sites.

Based on combined results of Phases 1 and 2 of this study, both DOC:TC and MBC:TC were found to be relatively good indicators of soil "quality" in upland areas, as related to site impacts or ecological condition. The potential value of the DOC:TC parameter as a robust indicator, i.e., beyond the scope of this study, for either soil quality or ecological condition cannot be determined solely from these results. The MBC:TC parameter, on the other hand, has been widely used as an indicator of bioavailability of soil organic C (Anderson and Domsch, 1989; Sparling, 1992). Although the biochemical processes governing the relationship between qCO_2 and soil impact or condition are not known with any certainty, our study results suggest that this parameter also may be a useful indicator of ecological condition or change, primarily for upland areas. Based on prior research, the utility of the metabolic quotient as an ecological indicator may be relatively region- or site-specific, and/or limited by the ability to identify appropriate assumptions and criteria for reliable use of this parameter (Wardle and Ghani, 1995).

The application of soil TC, TN and related parameters to evaluate sedimentation-related impacts to bottomland soils was not as straightforward as for upland soils. Level of impact was most closely related to those parameters directly associated with soil organic matter content, i.e., TC, TN, MBC, and respiration, and less related to the derived parameters that reflect C availability or soil quality. For the type of bottomland/wetland impact addressed by this study, i.e., accelerated deposition of mineral sediments, the severity of impact was primarily a function of the depth and texture (proportion of sand, silt and clay) of the allochthonous material. In this case, direct measurement of these parameters would be the most expedient means for analyzing site impact in bottomlands.

REFERENCES

Anderson, T.-H., and K.H. Domsch. 1989. Ratios of microbial biomass carbon to total organic carbon in arable soils. Soil Biol. Biochem. 21:471-479.

Anderson, T.-H., and K.H. Domsch. 1990. Application of eco-physiological quotients (qCO_2 and qD) on microbial biomasses from soils of different cropping histories. Soil Biol. Biochem. 22:251-255.

Arshad, M.A., and S. Martin. 2002. Identifying critical limits for soil quality indicators in agro-ecosystems. Agr. Ecosyst. Environ. 88:153-160.

Bailey, R.G. 1995. Description of the Ecoregions of the United States, 2d ed., USDA-Forest Service Miscellaneous Publication 1391, Washington, D.C., 108 pp.

Burford, J.R., and J.M. Bremner. 1975. Relationships between the denitrification capacities of soils and total, water-soluble and readily decomposable soil organic matter. Soil Biol. Biochem. 7: 389-394.

Compton, J.E. and R.D. Boone. 2000. Long-term impacts of agriculture on soil carbon and nitrogen in New England forests. Ecology 81:2314-2330.

Compton, J.E., R.D. Boone, G. Motzkin, and D.R. Foster. 1998. Soil carbon and nitrogen in a pine-oak sand plain in central Massachusetts: Role of vegetation and land-use history. Oecologia 116:536-542.

Cook, B.D., and D.L. Allan. 1992a. Dissolved organic carbon in old field soils: total amounts as a measure of available resources for soil mineralization. Soil Biol. Biochem. 24:585-594.

Cook, B.D., and D.L. Allan. 1992b. Dissolved organic carbon in old field soils: compositional changes during the biodegradation of soil organic matter. Soil Biol. Biochem. 24: 595-600.

Garten, C.T. Jr., T.L. Ashwood, and V.H.Dale. 2003. Effect of military training on indicators of soil quality at Fort Benning, Georgia. Ecol. Indicators 3:171-179.

Haynes, R.J. 1999. Size and activity of the soil microbial biomass under grass and arable management. Biol. Fert. Soils 30:210-216.

Hirst, R.A., R.F. Pywell, R.H. Marrs, and P.D. Putwain. 2003. The resistance of a chalk grassland to disturbance. J. Appl. Ecol. 40: 368–379. Blackwell Publishing Ltd.

Horwath, W. R., and E. A. Paul. 1994. Microbial Biomass. p. 753-773. In R. W. Weaver et al. (ed). Methods of soil analysis, Part 2. Microbiological and biochemical properties. Soil Science Society of America, Madison, WI.

Insam, H., and K. Haselwandter. 1989. Metabolic quotient of the soil microflora in relation to plant succession. Oecologia 79:174-178.

Kade, A., and S.D. Warren. 2002. Soil and plant recovery after historic military disturbances in the Sonoran Desert, USA. Arid Land Res. Manag. 16:231-243.

Neff, J.C., and G.P. Asner. 2001. Dissolved organic carbon in terrestrial ecosystems: synthesis and a model. Ecosystems 4:29-48.

Nortcliff, S. 2002. Standardisation of soil quality attributes. Agr. Ecosyst. Environ. 88:161-168.

O'Brien, N.D., P.M. Attiwill, and C.J. Weston. 2003. Stability of soil organic matter in *Eucalyptus regnans* forests and *Pinus radiata* plantations in south eastern Australia. Forest Ecol. Manag. 185:249-261.

Ocio, J. A., and P. C. Brooks. 1990. Soil microbial biomass measurements in sieved and unsieved soil. Soil Biol. Biochem. 22:999-1000.

Singh, K.P., T.N. Mandal, and S.K. Tripathi. 2001. Patterns of restoration of soil physicochemical properties and microbial biomass in different landslide sites in the sal forest ecosystem of Nepal Himalaya. Ecol. Eng. 17:385-401.

Sparling, G. P. 1992. Ratio of microbial biomass carbon to soil organic carbon as a sensitive indicator of changes in soil organic matter. Aust. J. Soil Res. 30:195-207.

Wardle, D. A. 1993. Changes in the microbial biomass and metabolic quotient during leaf litter succession in some New Zealand forest and scrubland ecosystems. Funct. Ecol. 7:346-355.

Wardle, D.A., A. Ghani. 1995. A critique of the microbial metabolic quotient (qCO_2) as a bioindicator of disturbance and ecosystem development. Soil Biol. Biochem. 27:1601-1610.

Webb, R.H. 2002. Recovery of severely compacted soils in the Mojave Desert, California, USA. Arid Land Res. Manag. 16: 291-305.

Whitecotton, R.C.A., M.B. David, R.G. Darmody, and D.L. Price. 2000. Impact of foot traffic from military training on soil and vegetation properties. Environ. Manage. 26:697-706.

Table 1. Criteria used for classification of site impact for the Fort Benning study area.

Site Disturbance Criteria	Site Impact Category		
	Low Impact	**Moderate Impact**	**Severe Impact**
General Criteria	Low use military; Managed forest (mature forest)	Moderate use military (no tracked vehicles); Managed forest (early-mid successional)	Heavy training (e.g., tanks); Recent clear-cuts with severe soil disturbance/erosion
Specific Criteria			
Overstory vegetation	None apparent	Minimal disturbance	Extensive tree removal or damage
Understory vegetation	None or minimal	Moderate physical impact or removal	Cleared understory
Ground cover	None or minimal	Moderate physical impact or patches of bare ground	Extensive bare ground
Soil	None apparent	Disturbance of O and A horizons, minimal erosion or other soil loss	Extensive soil disturbance, erosion and other soil loss, disruption of natural horizonation
Fire	Minimal (controlled burn, not recent)	Recent controlled burn or more severe burn in recent past	Not used as sole criteria for severe impact

Table 2. Summary statistics for Phase 1 soil characterization for bottomland (n=98) and upland (n=203) sites at Fort Benning. (pct = percentile)

	Bottomland			Upland		
	Mean	10th pct	90th pct	Mean	10th pct	90th pct
pH	4.99	4.19	5.85	5.28	4.73	5.77
Total C (g kg^{-1})	38.3	7.0	100.6	10.9	4.6	18.4
DOC (mg kg^{-1})	121.3	21.3	272.7	65.5	18.8	141.0
MBC (mg kg^{-1})	561.2	122.0	1435.0	218.9	76.4	400.0
Total N (g kg^{-1})	2.03	0.33	4.92	0.42	0.20	0.74
C:N Ratio	18.9	14.6	23.8	26.8	18.7	34.7

Table 3. Mean values for biogeochemical parameters analyzed for Phase 2 sampling at Fort Benning. Means followed by the same letter are not significantly different (P<=0.05); separate means comparisons were performed for bottomland and upland sites. MIL=military, FOR=forestry, L=low, M=moderate, S=severe.

Landscape Position	Site Name	Site Impact	Total C	Total N	C:N Ratio	DOC	MBC	Soil Respiration
		Type/Level	*g kg-1*	*g kg-1*		*mg kg-1*	*mg kg-1*	*µg C g-1 h-1*
Bottomland Sites	BW1	MIL/L	155.0 a	8.62 a	18.2 a	433.7 a	2705 a	12.79 a
	LW	MIL/L	111.4 bc	4.99 b	21.9 b	170.1 b	762 b	6.78 b
	BW2	MIL/M	129.2 ab	5.61 b	23.0 b	393.7 a	1148 c	9.12 b
	OW	FOR/M	70.1 c	3.9 b	17.7 a	132.9 b	1099 c	6.71 b
	HW	MIL/S	11.30 d	0.46 c	26.5 c	25.9 c	178 d	1.17 c
Upland Sites	BU1	MIL/L	9.6 a′	0.24 a′	40.1 a′	74.4 a′	128 a′	0.36 a′
	HU1	MIL/L	10.5 a′b′	0.34 a′b′	32.7 a′b′	16.4 b′	138 a′b′	1.14 b′
	LU	FOR/L	10.1 a′	0.27 a′b′	38.8 a′	27.4 c′	170 b′	0.81 c′
	HU2	MIL/M	9.7 a′	0.44 a′b′	31.4 a′b′c′	15.9 b′	144 a′b′	0.77 c′
	OU1	FOR/M	12.9 b′	0.43 b′	31.8 b′	132.0 d′	273 c′	0.78 c′
	OU2	FOR/M	9.8 a′	0.39 b′	28.0 b′c′	96.1 a′d′	202 b′	0.46 a′
	HU3	MIL/S	2.1 c′	0.10 c′	23.0 c′	8.9 e′	101 a′	0.12 d′
	RH	MIL/S	2.6 c′	0.10 c′	26.9 b′c′	72.6 a′	199 b′	0.14 d′

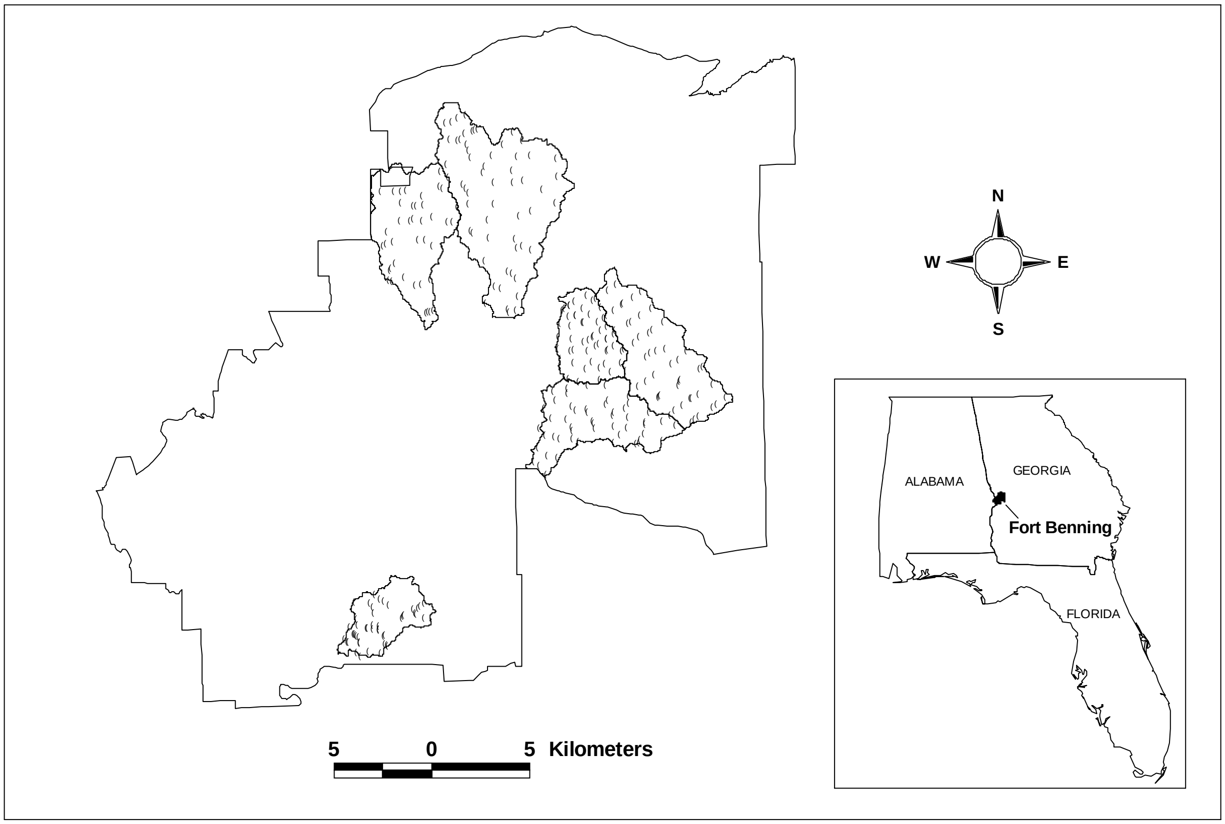


Figure 1
Location of Fort Benning, Georgia study area (inset) and Phase 1 watersheds with soil sampling locations.

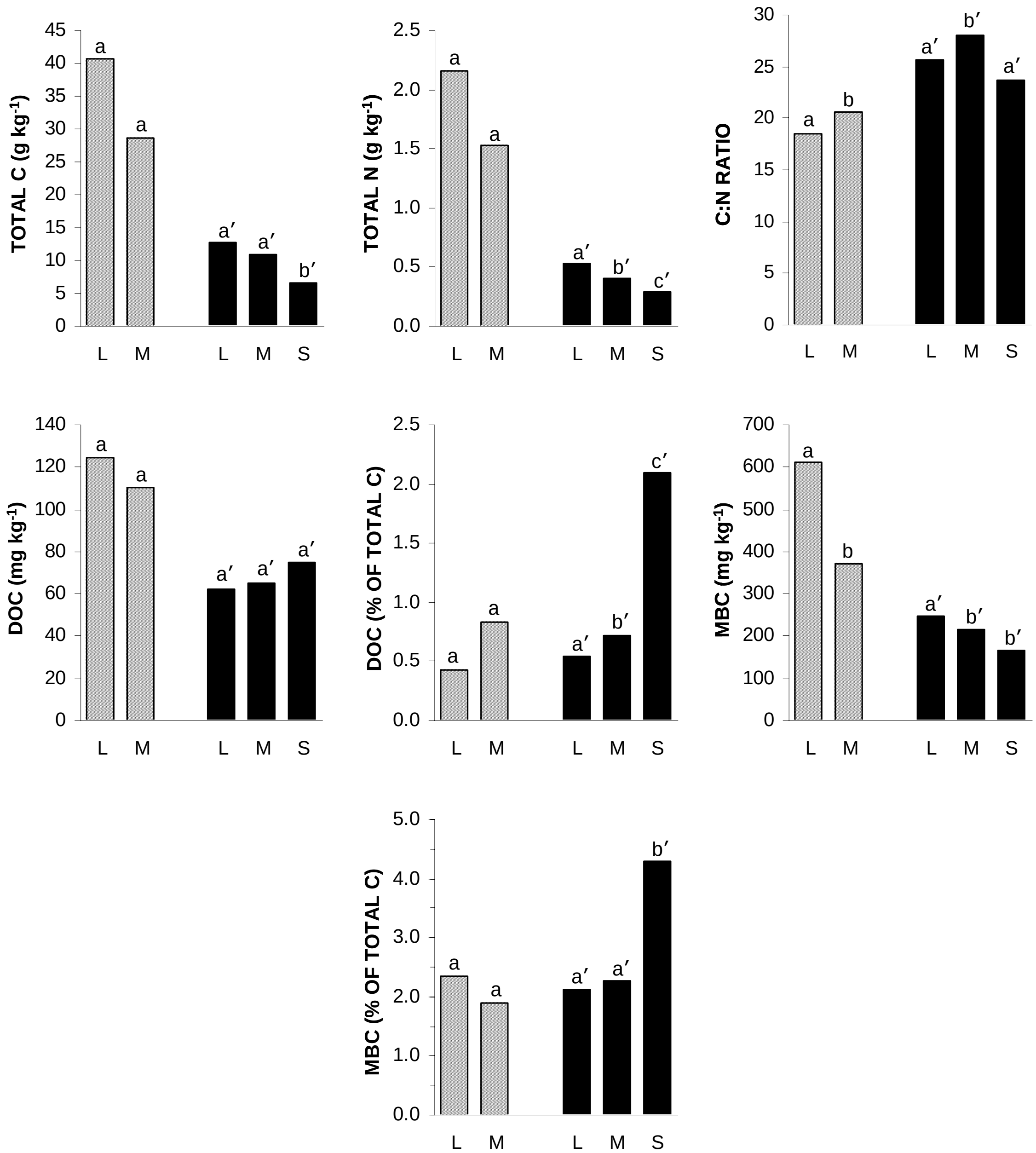


Figure 2
Summary results for the Fort Benning Phase 1 soil characterization study. Data points represent mean values for each level of site impact (L = low, M = moderate, S = severe) for bottomland (hatched bars) and upland (solid bars) sites. Means followed by the same letter are not significantly different (P<=0.05); separate means comparisons were performed for bottomland and upland sites. Severe-impact sites were not represented in bottom land areas for Phase 1.

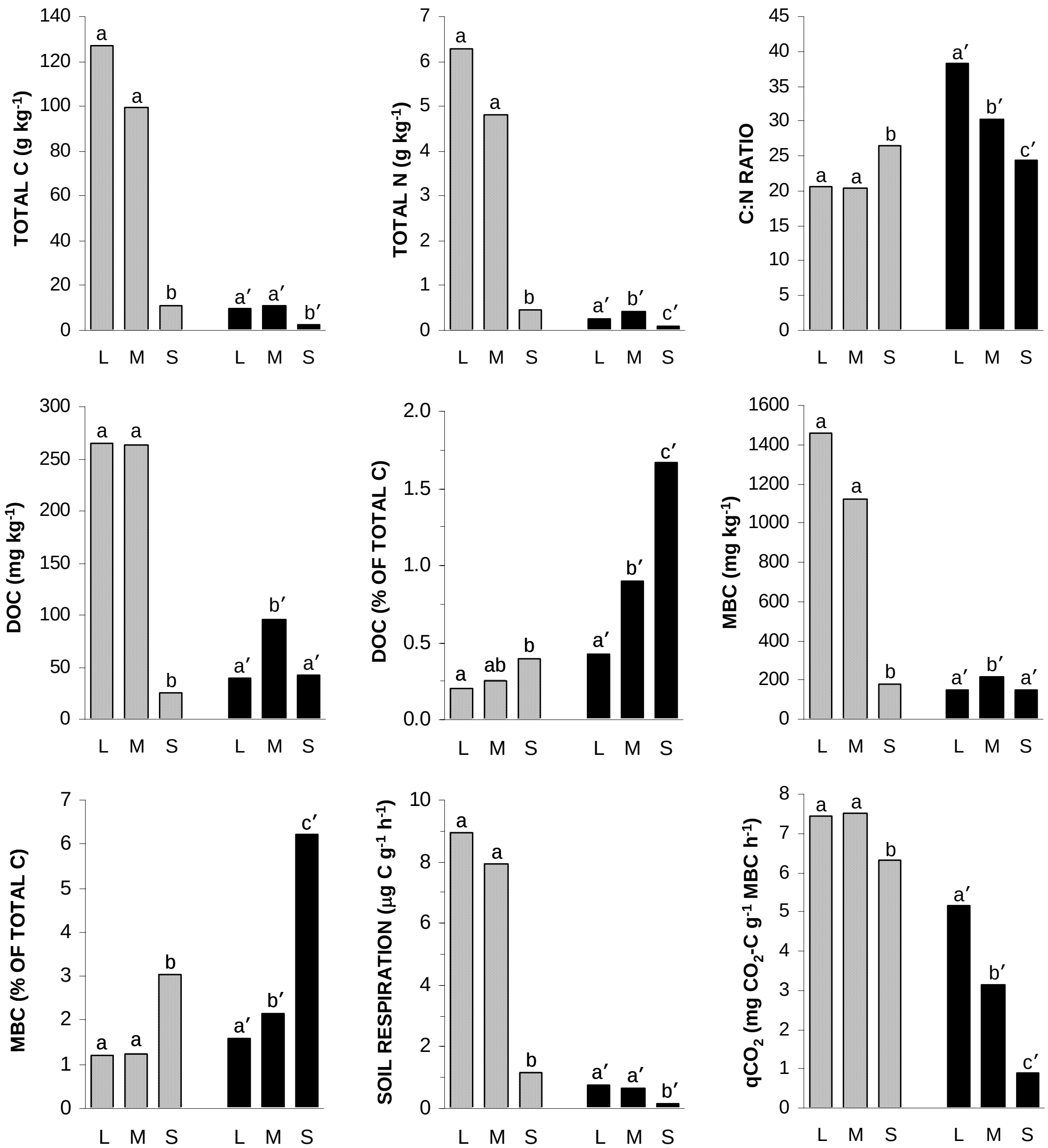


Figure 3
Summary results for the Fort Benning Phase 2 soil characterization study. Data points represent mean values for each level of site impact (L = low, M = moderate, S = severe) for bottomland (hatched bars) and upland (solid bars) sites. Means followed by the same letter are not significantly different (P<=0.05); separate means comparisons were performed for bottomland and upland sites.

3.1.2:

Influence of military land management on extracellular soil enzymes. Prenger, J. P., W. F. DeBusk, and K. R. Reddy.

ABSTRACT

In this study select soil chemical analyses and enzymatic activities were compared among watersheds and across community boundaries and disturbance gradients in order to examine differences due to anthropogenic influences. Activities of three soil enzymes (acid phosphatase, β-glucosidase, and dehydrogenase) and their relationship to soil total nitrogen (TN), phosphorus (TP), and carbon (TC), extractable cations, water extractable carbon and soil moisture were examined in soils from a range of disturbance by military training in a managed forest area at Ft. Benning, Georgia. Samples from 203 upland and 99 bottomland sites were collected from January to August 2000 in six 3rd to 4th order watersheds. Transects targeting more specific land uses and disturbance gradients were sampled in December 2000 and February 2002. Enzyme activities were generally related to TC, soil moisture, and season. Dehydrogenase best distinguished between qualitative disturbance levels based on vegetation and ground cover, soil disturbance, and other categorical factors.

INTRODUCTION

Ecological assessment of human perturbation at large spatial scales is often confounded by inherent differences among watersheds or adjacent ecosystems. While central to ecosystem function, soil quality has a high degree of natural variability due to landscape position, parent material, vegetation patterns, and other influences. However, the fundamental role of soil fertility in any ecosystem makes it an ideal focus for assessment despite the complexity of edaphic characteristics. Many anthropogenic impacts influence soil carbon stores through erosion, deposition, and modified sedimentation rates. Particularly in systems characterized by low productivity and nutrient stores, monitoring of soil carbon and nutrient dynamics may provide essential information regarding human impacts. Soil enzymes are potentially sensitive indicators of soil quality (Dick, 1994), particularly in upland systems (Ajwa et al., 1999; Saiya-Cork et al., 2002). While the response of enzymes in some wetland soils to nutrient disturbances has been examined (McLatchey and Reddy, 1998; Wright and Reddy, 2001a), enzyme response in riparian areas remains less studied. Bottomland riparian areas are extremely diverse ecosystems in which a variety of natural disturbances contribute to temporal and spatial variability (Naiman and Décamps, 1997).

Exogenous enzymes are critical to the turnover of organic matter and the soil nutrient cycle. Freshly deposited organic material (e.g., leaf litter) composed of high molecular weight compounds require hydrolysis by extracellular enzymes before becoming available to microbial or plant populations (Burns, 1982; Chrost, 1991; Sinsabaugh et al, 1991). Microbial cells release exogenous enzymes which are integral to degradation of organic matter and plant detritus (Halemejko and Chrost, 1984, Sinsabaugh et al, 1992), and thus help to determine soil quality and vegetation community structure. Soil enzyme activities are correlated with microbial biomass carbon (Dick et al., 1988), and the dynamic nature of microbial communities makes it sensitive to physical disturbance (Dick et al., 1988), nutrient loading (Powlson and Jenkinson, 1981) and other perturbation. Changes in organic C, total N, and bulk density affect enzyme activities in upland soils (Dick et al., 1988) and extracellular enzyme levels have been shown to reflect the nutrient status of wetland soils (Wright and Reddy 2001b).

As one of a suite of cellulose-degrading enzymes, β-glucosidase is involved in the generation of monosaccharides through hydrolysis of glycosides (Eivazi and Tabatabai, 1988), and can be correlated with detrital degradation rates (Sinsabaugh et al., 1994; McLatchey and Reddy, 1998). The activity of extracellular phosphatases often indicate impacts of phosphorus on wetland soils or aquatic ecosystems (Gage and Gorham, 1985; Wetzel, 1991; Newman and Reddy, 1993; Wright and Reddy, 2001a) since phosphatase activities are repressed by high concentrations of dissolved reactive P (DRP) (Cembella et al., 1984; Chrost, 1991). Other soil enzymes, particularly dehydrogenase (Casida 1977), may be used to indicate overall microbial biomass and activity levels.

There is a general paucity of information regarding the impact of military training on soil biogeochemical characteristics. While some work has been done on recovery of soils from encampments (Kade and Warren 2002) and troop and mechanized training (Webb 2002), these studies have involved desert ecosystems and focused on physical alterations, particularly soil compaction. A controlled study of mechanized military training activity in chalk grasslands (Hirst et al., 2003) showed significant changes in soil compaction but did not report changes in soil chemistry. A recent study at Ft. Benning, Georgia USA by Garten et al. (2003) found greater surface soil bulk density, lower soil carbon concentrations, and less carbon and nitrogen in particulate organic matter at moderate use, heavy use, and remediated sites relative to reference sites.

The current study describes changes in selected soil chemistry and enzyme activities due to soil disturbance related to military training in the context of a managed forest. Comparisons were made of chemistry in upland and riparian soils from six 3rd and 4th order watersheds impacted by soil disturbance and erosion due to mechanized training and forestry management practices. Enzyme activities were compared among 5 of the six watersheds. Transects across land-use gradients specifically targeting transitional areas were sampled in December 2000 and February 2002, and exhibited seasonal differences in enzyme activities in upland soils. In bottomlands, only dehydrogenase activity (a surrogate of microbial respiration) showed seasonal differences, particularly with respect to soil TC. Targeted sampling across community boundaries and disturbance gradients reduced geomorphological variability and allowed better discrimination of differences due to anthropogenic influences.

METHODS

Site Description

The study area is within the Ft. Benning military installation in Chattahoochee and Muscogee Counties in west-central Georgia, in the Carolina and Georgia sand hills major land resource area (USDA, 1997). Upland soils in the area are primarily well- to excessively- drained Ultisols and Entisols, supporting forests of slash (*Pinus elliotii*), longleaf (*P. palustris*), and loblolly (*P. taeda*) pines. Excessively-drained Lakeland soils (Entisol) of sandhill communities are associated with longleaf pine, turkey oaks (*Quercus laevis*), blackjack oaks (*Q. marilandica*), and post oaks (*Q. stellata*) near ridgetops in the central and northern portion of the installation. Loamy soils of relatively high clay content occur in upland areas in a band across the southern portion of the installation. Wetlands and hydric soils are generally restricted to bottomlands along streams and creeks. Military related impacts result from the direct removal of or damage to vegetation, digging activities, and ground disturbance from vehicles. The mechanized forces in particular use tracked and wheeled vehicles that cause soil disturbance and movement that may result in soil erosion and stream sedimentation. Forestry management activities occur throughout the base and include logging, thinning, and controlled burns.

Assessment of site condition

Soil samples were obtained from sites representing a wide range of military and non-military land uses and other anthropogenic disturbance regimes. Assessment of soil quality and its response to anthropogenic disturbance would ideally involve correlating response variables with known levels of site disturbance. Unfortunately, documentation of military training activity within the study area is inadequate to accurately estimate frequency and duration of site disturbance. It was therefore necessary to define criteria for determining the current level of impact at each site and to assign an impact level or class to each site. A standard set of criteria was developed for the purpose of determining the level of impact, primarily based on visual inspection of vegetation and soil properties. Because of the variety of disturbance modes, disturbance was categorized as severe, moderate, or low, based in general on the level of military usage (heavy, moderate, low) and/or the age of forest cover (mature or early successional). At each site the dominant vegetation type was noted, and the site was classified according to estimated overall level of disturbance based on both visual observation of soil and vegetation disturbance (Table 1) and documented land use. Accordingly, each site was placed in one of three disturbance classes: low, moderate or severe.

Soil Collection

Large-scale spatial sampling was conducted from January to August 2000 in both upland and bottomland areas of six 3rd or 4th order watersheds, including Bonham, Sally Branch, Halloca, Randall, Shell, and Wolf Creeks. Soil for chemical analyses was obtained at 99 bottomland and 203 upland sites along transects transverse to the orientation of the main stream channel, providing approximately uniform coverage of the watershed. Field replicates from all watersheds but Bonham Creek were examined for soil enzyme activity levels (approximately 15% of sites including 16 bottomland sites and 28 upland sites). Each sample site consisted of a 1 m^2 square plot, within which 5 subsamples were taken at the corners and center of the plot. Soil was sampled to a depth of 20 cm, using a soil push probe with an inside diameter of 2.5 cm. The individual sub-samples were then composited for analysis as a single sample. The spatially-intensive sampling scheme, stratified by landscape position, was designed to capture the full range of spatial variability of soil properties within each watershed.

The second phase of sampling, conducted in December 2000 and February 2002, was designed to characterize anthropogenic impact gradients on a smaller scale, at a greater spatial resolution, and to account for changes in soil moisture content and season. In uplands, transects focused on the transition from native oak community (turkey, blackjack, and post) to severely impacted (denuded) upland or from native oak to planted pine. Wetland transects were sampled in riparian areas of low and severe military impact watersheds. Bottomland riparian soils were sampled to a depth of 10 cm using a 6.5-cm diameter polycarbonate corer at 20 m intervals from 6 transects. Each of the 35 samples represented a composite of three subsamples taken within a 1 m^2 quadrat. Upland transects were sampled at 20 m intervals over a length of 400 m (21 samples in each of two transects), and samples were obtained as described above for spatial sampling. All soil samples were sealed in water-tight plastic bags and placed on ice immediately after sampling and stored at 4^o C in the lab until analyses were performed. In the lab, samples were homogenized and all visible plant fragments and roots were removed.

Chemical Analyses

Well mixed soil samples were analyzed for percent moisture, total C, N, and P, water extractable carbon (WEC), and Mehlich I extractable iron, aluminum, calcium, and potassium (Meh Fe, Meh Al, Meh Ca, Meh K). Percent moisture was calculated on a wet weight basis [(wet weight – dry weight) / wet weight]*100. Total C and N content was determined on soil dried for 24 hours at 105°C, ground by ball mill, and analyzed by dry combustion (Nelson and Sommers, 1996) using a Carlo-Erba NA-1500 CNS Analyzer (Haak-Buchler Instruments, Saddlebrook, NJ). Total P was determined by combusting approximately 0.2-0.5 g oven-dried, finely ground soil at 550^o C for 4 hrs, digesting the ash with 6 M HCl and continuous heating on a hot plate, and filtering through No. 41 Whatman filter (Anderson, 1976), followed by analysis of P by automated ascorbic acid method (Method 365.1, USEPA, 1983). Water extractable carbon was determined by extraction of the wet soil equivalent of 2.5 g soil dry weight in 25 mL of distilled deionized water with shaking (30 rpm) for 1 hr, followed by filtration through 0.45 μm membrane filter (Kuo, 1996) and analysis on a Dohrmann DC-190 TOC Analyzer. Mehlich extractable cations were determined by extraction of 5 grams soil in 20 mL 0.0125 M H_2SO_4, 0.05 M HCl with 5 min shaking (30 rpm), followed by filtration through No. 42 Whatman filter paper (Amacher, 1996) and analysis by ICP (EPA method 200.7).

Enzyme Analyses

Enzyme activities were assayed using the fluorescent model substrate 4-methylumbelliferone (MUF) (Chrost and Krambeck 1986; Hoppe 1993; Sinsabaugh et al. 1997) at the approximate ambient pH (6.0). All soil enzyme analyses were performed on well mixed fresh material from which all visible roots and living plant material was removed. Enzyme analyses were completed within 2 weeks of sampling, except December ’00 which were completed within 3 weeks. Enzyme activities were determined only on field replicate samples for five watersheds in the spatial study (approximately 15% of sites) and on all transect samples. Soil samples (~ 1 g) were placed in approximately 9 mL distilled water, and clumps were broken up by brief agitation with a Tissue Tearor Model 398 (Biospec Products, Bartlesville, OK). Immediately prior to enzyme assays, a 1/100 or 1/200 dilution of soil or detritus was prepared in water by serial dilution. Two hundred µL of well-suspended soil slurry was transferred by pipette into 8 wells of a 96 well microtiter plate, and 50 µL of substrate solution added to 4 wells (with 4 blanks). Samples were incubated (2 hr for phosphatase, 24 hr for all others) in the dark at room temperature except for dehydrogenase which was incubated at 30^{o}C. Phosphatase and β-glucosidase assays were stopped by addition of 10 µL 0.1 N NaOH. Substrate was added to blanks and immediately read on a Bio-Tek Model FL600 fluorometric plate reader (Bio-Tek Instruments, Inc., Winooski, VT). Dehydrogenase assays were stopped with 50 µL acetone, incubated an additional 2 hr and read. Substrate solutions were as follows: for acid phosphatase, 500 µM methyl-umbelliferyl (MUF)-phosphate in 5 mM MES pH 6.0; for β-glucosidase, 500 µM MUF-glucoside in 5mM MES pH 6.0; for dehydrogenase, 500 µM 5-cyano-2,3-ditolyl tetrazolium chloride (CTC) in 100mM Tris pH 7.8. Concentrations were calculated from a standard curve of MUF or CTC-Formazan. Excitation (Ext.) and emission (Em.) spectra for the two fluoro-chromes were: Ext. 360$\pm$40, Em. 460$\pm$40 (MUF-P, MUF-G); and Ext. 530$\pm$25, Em. 645$\pm$40 (CTC). Enzyme activities were calculated as µg product g $^{-1}$ dry soil hour $^{-1}$. All enzyme activities were normalized to their maximum value (separately for bottomland and upland sites) and reported on a scale of 0-1.

Statistical Analyses

Pairwise comparison of means with Tukey-Kramer HSD, multivariate correlation matrix, and least squares modeling were performed using JMP version 4.0.5 (SAS Institute, Cary NC). All variables were examined for normality of variance and log transformed where necessary. Outliers were identified by Mahalanobis distance from the multivariate mean (spatial soil chemistry data) or as points outside 1.5*(interquartile range) of log transformed data (correlations and transect data). Analysis of variance of univariate data included multiple comparison of means by Tuckey - Kramer and in all cases was at experiment-wise α=0.05.

RESULTS AND DISCUSSION

Spatial study

Comparison of Soil Chemistry by Watershed

Any attempt to assess impacts at large spatial scales must deal with inherent variability among watersheds or adjacent ecosystems. In this study soil chemical analyses were compared among six watersheds in the Fort Benning Military Installation (Fig. 1; Tables 2 and 3). About 40 to 60 soil samples were obtained from each watershed from January to August 2000, with approximately one third of these from bottomlands. Table 2 provides a summary of chemical analyses of bottomland and upland soil samples. Shell Creek bottomland soils were significantly higher than those of all others in mean Ca and K values. Shell Creek bottomland soils had the lowest mean WEC (significant only in two). Upland soils from the Shell Creek watershed were significantly higher in TP and TN, and TC was significantly higher than in Bonham, Halloca, and Randall watershed soils. These differences are most likely due to the prevalence of clay-loam soils in the Shell Creek bottomlands, as opposed to mainly sandy soils in bottomlands of the other five watersheds (Nankin Sandy Clay Loam vs. Bibb Sandy Loam and Troup Loamy

Sands; USDA-NRCS Soil Survey of Chattahoochee and Marion Counties). In uplands, sandy loams were prevalent in the Shell Creek watershed, while loamy sands dominated the other watersheds. Randall Creek watershed showed significantly lower Fe and TP levels than Bonham, Halloca, Sally Branch, and Wolf watersheds.

Figure 2 depicts differences in soil enzyme levels in bottomland and upland soil samples from five of the six watersheds in this study. The watersheds differ drastically in the level and areal extent of disturbance (Table 2), with the ridge dividing the Bonham and Sally Branch watersheds (Fig. 1) being the most heavily impacted by military vehicles and erosion. In general, soil enzyme levels did show variation among watersheds (Fig. 2), although it is not known to what extent, if any, these trends reflect seasonal variation since different watersheds were sampled over the period of March to August 2000. Dehydrogenase activity was highest in the Randall and Wolf watersheds, which were sampled in June 2000, indicating greater overall microbial activity. In contrast, acid phosphatase and β-glucosidase, two enzymes involved in nutrient recycling, were highest in Sally Branch and Halloca Creeks which were sampled in spring (March and May, respectively).

Comparison of Soil Chemistry by Disturbance Level

When soil analyses from the large-scale spatial sampling are compared based on level of disturbance, severely disturbed upland sites were significantly lower in Mehlich extractable Fe and Al, TC and TN (Table 3). Due to the high degree of within-class variability no significant differences in chemical content of bottomland soils are observed, although like uplands, severely disturbed sites tended to be lower in TC, TN, TP and extractable Fe and Al. Enzyme activities were not significantly different between the three disturbance classes in either landscape position (not shown).

Correlations

A correlation matrix of data from all sites of the spatial study (Table 4) indicates a weak correlation in upland soils between dehydrogenase activity and TC values (0.5545). There was a negative relationship between dehydrogenase and β-glucosidase activities (-0.6289). The strongest relationship was in bottomland soils between β-glucosidase and acid phosphatase (0.8742**)** and between dehydrogenase and TC and TN (0.8558 and 0.7464, respectively). WEC was most highly correlated with TC and TN in bottomlands. The relatively high correlation of WEC with dehydrogenase activity (0.6323) indicates the importance of soluble carbon in microbial activity, either as a substrate or the product of microbial activity. The high correlation of dehydrogenase with TC and TN is probably due to the quality and quantity of available carbon for microbial activity (Saggar et al., 2001). The lack of strong association of β-glucosidase or TC with WEC is somewhat surprising, since one would expect WEC to contain a relatively large proportion of easily degraded and readily available carbon; however, this may be due to the inherent variability in bottomland soils in the Ft. Benning area, which range from highly mineral to highly organic. In upland soils, the strongest associations were between TC and TN, TP and TN, and between TC and dehydrogenase activity, indicating the relationship between microbial activity and organic matter content and quality. There was a negative relationship between dehydrogenase and β-glucosidase activities (-0.6289), suggesting that microbial activity was greatest in soils with more labile carbon stocks that did not require extensive enzymatic degradation.

Organic matter inputs to riparian wetlands are determined by bottomland vegetation and contributions from the watershed (Naiman and Décamps, 1997; Craft and Casey, 2000). As open systems, floodplain and riparian wetlands receive water soluble inputs from the surrounding watershed (Naiman and Décamps, 1997), including dissolved metals, nutrients, and organic carbon. In Ft. Benning, areas of forestry and military disturbance have increased sediment deposition in some bottomlands, and labile nutrients (Naiman and Décamps, 1997) and dissolved and particulate organic material may also be increased. Nutrient dynamics and organic carbon accumulation in the riparian areas are therefore influenced by import and export, as well as decomposition and nutrient recycling (Craft and Casey, 2000). These relative microbial enzyme activities may be useful as indicators of decomposition rates and relative nutrient availability (Sinsabaugh and Moorhead, 1994).

Transect Study

In order to more clearly define soil changes due to anthropogenic impacts, soils were sampled along transects in low, moderate, and severely disturbed areas. The transect sampling strategy highlighted the difficulty in categorizing disturbance based on chemical properties versus visual or qualitative designation of disturbance. When targeted to specific bottomlands and uplands of varying impact level, more definitive differences between disturbance levels were evident (Table 5); however, bottomlands in this study were typically not directly impacted by military vehicles (the most severe disturbance). Rather, the most severely impacted bottomland transects suffered from burial by inorganic sediments from severely degraded upland areas disturbed by military vehicles. Values for the severely impacted bottomland sites were significantly lower than low impact sites for all parameters in Table 5, but did not differ significantly from moderate impacts due to high within-class variability. Severely disturbed upland sites were significantly lower in TC, TN, and water extractable C (Table 5). This indirect impact caused the concomitant decrease in organic material, resulting in severely disturbed bottomland soils that were significantly lower in Ca, Fe, Al, K, WEC, TC, TN, and TP.

There were significant differences ($\alpha < 0.05$) between sampling dates for all three enzyme activities in upland soils, but only for dehydrogenase in bottomlands (Table 6). There was not a clear relationship between enzyme activities and TC values in upland soils (Table 4) while in bottomland soils there was a strong relationship between TC and dehydrogenase activities. Fig. 3 indicates the relationship between the log of TC values versus the log of percent soil moisture and the log of activity levels for three enzymes from bottomland soils sampled in December 2000 (squares) and February 2002 (circles). TC for bottomlands was highly correlated to soil moisture, with an R^2 of 0.95 for December ’00 and 0.93 for February ’02 samples. The activity of β-glucosidase is closely related to TC levels for both sampling dates (R^2=0.93 and 0.87, respectively), while that of acid phosphatase is somewhat lower (R^2=0.86 and 0.72, respectively). Dehydrogenase showed two distinct slopes for the two sampling dates, with R^2 of 0.95

and 0.91, respectively, indicating higher activity per unit of carbon in December ’00 versus February ’02 soil samples. Water content for soils from the two sample dates were similar (42.2$\pm$20.8 % for December vs. 41.3$\pm$19.9% for February), as were meteorological conditions for the two months: mean precipitation <0.1 mm; December mean air temperature and range 4.4 ^{o}C, -9.3 to 20.3 ^{o}C; February mean air temperature and range 9.0 ^{o}C, -7.3 to 22.5 ^{o}C. Differences in TC and WEC between the two sampling times were not significant.

β-glucosidase activities distinguished well between the low and severe disturbance levels in bottomland transects (Fig. 4), but did not distinguish moderately from severely disturbed areas. In uplands β-glucosidase activities failed to distinguish low from moderate disturbance sites. β-glucosidase activities were significantly different between low and severe disturbance areas of uplands, due entirely to differences in the February ‘02 samples; December ’00 samples did not separate any of the three disturbance categories (Fig. 4). Dehydrogenase also failed to distinguish moderate from low and severe disturbance upland areas in February, but did distinguish moderate from severe disturbance in the December samples (α< 0.05, Fig. 4). Unlike β-glucosidase, dehydrogenase activity distinguished low from severe disturbance in both sampling periods, with consistently lower activities in February ’02 for each disturbance class as reflected in lower activity per unit TC in Fig. 3.

A statistical model was developed using transect data to investigate factors influencing soil enzyme levels. The general form of the model was: Enzyme Activity = sampling date + % moisture + TC + date*TC + TC*% moisture + date*% moisture + date*TC*% moisture. Effects and their level of significance for β-glucosidase and dehydrogenase from the two landscape positions are given in Table 7. Time of year samples were collected was a significant factor for the two enzyme activities in both landscape positions. Percent moisture was a significant predictor for both β-glucosidase and dehydrogenase levels in bottomlands (p <.0001 for both), but was not significant for either activity in uplands. TC and the interactions of TC with both date of sampling and % moisture were significant for dehydrogenase levels in uplands, but in bottomland soils TC was only a significant predictor of dehydrogenase activity as a cofactor with moisture content. The interactions of sampling date with TC and % moisture, as well as the three-way interaction were significant for β-glucosidase regardless of landscape position. The three-way interaction was not significant for dehydrogenase.

The activity of extracellular enzymes may be useful indictators of impacts on wetland and aquatic ecosystems (Gage and Gorham, 1985; Wetzel, 1991; Newman and Reddy, 1993; Wright and Reddy, 2001a). In the southeastern coastal plain and, in particular, the Ft. Benning Military Installation, soils are often relatively low in carbon content. The availability of carbon can be limiting even in relatively carbon-rich bottomlands where recalcitrant forms may dominate. Most of the chemical parameters and enzyme activities determined on bottomland soils in this study failed to distinguish well between low and moderate impacts. β-glucosidase and dehydrogenase activities did distinguish low from severe impact in bottomland and upland transects (Fig. 3), but failed to separate moderate from low and severe impacts in upland soils.

The reduced internal variability in data from targeted sampling allows relationships between the various components to be observed more readily. The linear relationships observed in log transformed data (Fig. 4) support the conclusion that microbial activity is dependent on available carbon. Log dehydrogenase activity versus log TC showed two distinct slopes based on date of sampling (Fig. 4), suggesting a possible dependence on carbon quality or soil temperature, both of which would be expected to decrease during winter months. The log of acid phosphatase activity demonstrated a somewhat weaker correspondence to log TC values, probably due to acid phosphatase levels being more dependent on available phosphorus rather than available carbon. β-glucosidase has been correlated with detrital degradation rates (Sinsabaugh et al., 1994; McLatchey and Reddy, 1998), and the unique slopes for it and acid phosphatase indicate persistence of these enzymes and lack of direct dependence on current microbial activity. The observation of independence from seasonal factors suggests the utility of these enzymes for monitoring impacts.

CONCLUSION

This study has demonstrated changes in select soil chemistry and enzyme activities due to soil disturbance by mechanized military activity. Seasonal differences in soil microbial activities probably confound the interpretation of enzyme activity levels in the large scale spatial study with regard to loss of soil TC. While microbial activity as indicated by dehydrogenase levels differed depending on time of sampling, extracellular enzyme activities appear to be persistent and dependent on total soil carbon and soil moisture in erosion and sediment impacted bottomlands and uplands influenced by military training activity. While the activity of β-glucosidase often did not significantly differ in moderately disturbed areas in this study, increased sampling density would increase the discriminatory power and thus may be a promising indicator of disturbance, particularly in bottomland soils. Extracellular enzyme activities relative to overall microbial activity and carbon availability can potentially be used as an efficient indicator of soil disturbance. This may help to better understand soil changes during and following physical disturbance of soils at the watershed scale.

REFERENCES

Amacher, M.C. 1996. Double Acid (Mehlich I) Method. In: Bigham, J.M. (Ed.), Methods of Soil Analysis Part 3: Chemical Methods. SSSA-ASA, Madison, Wisconsin, p. 757.

Anderson, J. M. 1976. An ignition method for determination of total phosphorus in lake sediments. Water Research 10: 329-331.

Ajwa, H.A., C.J. Dell, and C.W. Rice. 1999. Changes in enzyme activities and microbial biomass of tallgrass prairie soil as related to burning and nitrogen fertilization. Soil Biology and Biochemistry 31: 769-777.

Burns, R.G. 1982. Enzyme activity in soil: location and a possible role in microbial ecology. Soil Biology and Biochemistry 14: 423-427.

Casida L.E. 1977. Microbial metabolic activity in soil as measured by dehydrogenase determinations. Applied Environmental Microbiology 34: 630–636.

Cembella, A.D., N.J. Antia, and P.J. Harrison. 1984. The utilization of inorganic and organic phosphorus compounds as nutrients by eukaryotic microalgae: A multidisciplinary perspective. Part1. Critical Reviews in Microbiology 10: 317-391.

Chrost, R.J. 1991. Environmental control of the synthesis and activity of aquatic microbial ectoenzymes. In: Chrost, R.J. (Ed.), Microbial enzymes in aquatic environments. Springer-Verlag, New York, pp. 25-59.

Chrost, R.J., and H.J. Krambeck. 1986. Fluorescence correction for measurements of enzyme activity in natural waters using methylumbelliferyl-substrates. Archiv für Hydrobiologie 106: 79-90.

Craft, C.B., and W.P. Casey. 2000. Sediment and nutrient accumulation in floodplain and depressional freshwater wetlands of Georgia, USA. Wetlands 20: 323-332.

Dick, R.P. 1994. Soil enzyme activities as indicators of soil quality. In: Doran, J.W., Coleman, D.C., Bezdicek, D.F., Stewart, B.A. (Eds.), Defining Soil Quality for a Sustainable Environment. American Society of Agronomy, Madison, WI, pp. 107-124.

Dick, R.P., D.D. Myrold, and E.A. Kerle. 1988. Microbial biomass and soil enzyme activities in compacted and rehabilitated skid trail soils. Soil Science Society of America Journal 52: 512-516.

Eivazi, F., and M.A. Tabatabai. 1988. Glucosidases and galactosidases in soils. Soil Biology and Biochemistry 20: 601-606.

Gage, M.A., and E. Gorham. 1985. Alkaline phosphatase activity and cellular phosphorus as an index of the phosphorus status of phytoplankton in Minnesota lakes. Freshwater Biology 15: 227-233.

Garten, C.T. Jr., T.L. Ashwood, and V.H. Dale. 2003. Effect of military training on indicators of soil quality at Fort Benning, Georgia. Ecological Indicators 3:171-179.

Halemejko, G.Z., and R.J. Chrost. 1984. The role of phosphatases in phosphorus mineralization during decomposition of lake phytoplankton blooms. Archiv für Hydrobiologie 101: 489-502.

Hirst, R.A., R.F. Pywell, R.H. Marrs, and P.D. Putwain. 2003. The resistance of a chalk grassland to disturbance. Journal of Applied Ecology 40: 368–379.

Hoppe, H.G. 1993. Use of fluorogenic model substrates for extracellular enzyme activity (EEA) measurement of bacteria. In: Kemp, P.F., et al. (Eds.), Handbook of Methods In Aquatic Microbial Ecology. Lewis Publishers, Boca Raton, FL, pp. 423-431.

Kade, A., and S.D. Warren. 2002. Soil and Plant Recovery After Historic Military
Disturbances in the Sonoran Desert,USA. Arid Land Research and Management 16: 231-243.

Kuo, S., 1996. Phosphorus. In: Bigham, J.M. (Ed.), Methods of Soil Analysis. Part 3, Chemical Methods. Soil Science Society of America, ASA, Madison, Wisconsin. pp. 891.

McLatchey, G.P., and K.R. Reddy. 1998. Regulation of organic matter decomposition and nutrient release in a wetland soil. Journal of Environmental Quality. 27: 1268-1274.

Naiman, R.J., and H. Décamps. 1997. The ecology of interfaces: Riparian zones. Annual Review of Ecology and Systematics 28: 621-658.

Nelson, D.W., and L.E. Sommers. 1996. Total Carbon and Total Nitrogen. In: Bigham, J.M. (Ed), Methods of Soil Analysis. Part 3,Chemical Methods. SSSA ASA. Madison, Wisconsin. p. 973.

Newman, S., and K.R. Reddy. 1993. Alkaline phosphatase activity in the sediment-water column of a hypereutrophic lake. Journal of Environmental Quality 22: 832-838.

Powlson, D.S., and D.S. Jenkinson. 1981. A comparison of the organic matter, biomass, adenosine triphosphate and mineralizable nitrogen contents of ploughed and direct-drilled soils. Journal of Agricultural Science 97: 713-721.

Saggar, S., C.B. Hedley, and G.J. Salt. 2001. Soil microbial biomass, metabolic quotient, and carbon and nitrogen mineralization in 25-year-old *Pinus radiata* agroforestry regimes. Australian Journal of Soil Research 39: 491-504.

Saiya-Cork, K.R., R.L. Sinsabaugh, and D.R. Zak. 2002. The effects of long term nitrogen deposition on extracellular enzyme activity in an Acer saccharum forest soil. Soil Biology and Biochemistry 34: 1309-1315.

Sinsabaugh, R.L., and D.L. Moorhead. 1994. Resource allocation to extracellular enzyme production: a model for nitrogen and phosphorus control of litter decomposition. Soil Biology and Biochemistry 26: 1305-1311.

Sinsabaugh, R.L., R.K. Antibus, and A.E. Linkins. 1991. An enzymatic approach to the analysis of microbial activity during plant litter decomposition. Agriculture, Ecosystems, and Environment. 34: 43-54.

Sinsabaugh, R.L., T. Weiland, and A.E., Linkins. 1992. Enzymic and molecular analysis of microbial communities associated with lotic particulate organic matter. Freshwater Biology 28: 393-404.

Sinsabaugh, R.L., M.P. Osgood, and S. Findlay. 1994. Enzymatic models for estimating decomposition rates of particulate detritus. Journal of the North American Benthological Society 13: 160-169.

Sinsabaugh, R.L., S. Findlay, P. Franchini, and D. Fisher. 1997. Enzymatic analysis of riverine bacterioplankton production. Limnology and Oceanography 42: 29-38.

USDA, 1997. Natural Resources Conservation Service Major Land Resource Area Boundaries. http://www.nrcs.usda.gov/technical/land/mlra/

Webb, R.H. 2002. Recovery of Severely Compacted Soils in the Mojave

Desert, California,USA. Arid Land Research and Management 16: 291-305.
Wetzel, R.G. 1991. Extracellular enzymatic interactions: Storage, redistribution, and interspecific competition. In: Chrost, R.J. (Ed.), Microbial enzymes in aquatic environments. Springer-Verlag, New York. p. 6-28.
Wright, A. L., and K.R. Reddy. 2001a. Heterotrophic microbial activity in Northern Everglades wetland soils. Soil Science Society of America Journal 65: 1856-1864.
Wright, A. L., and K.R. Reddy. 2001b. Phosphorus loading effects on extracellular enzyme activity in Everglades wetland soils. Soil Science Society of America Journal 65: 588-595.

Table 1. Criteria used for determination of site disturbance level.

Disturbance Level	Disturbance Category				
	Vegetation			Soil	Fire
	Overstory	Understory	Ground cover		
Low	None apparent	None or minimal	None or minimal	None apparent	Minimal (controlled burn, not recent)
Moderate	Minimal disturbance	Moderate physical impact or removal	Moderate physical impact or patches of bare ground	Disturbance of O and A horizons, minimal erosion or other soil loss	Recent controlled burn or more severe burn in recent past
Severe	Extensive tree removal or damage	Cleared understory	Extensive bare ground	Extensive soil disturbance, erosion and other soil loss, disruption of natural horizonation	Not used as sole criteria for severe impact

Table 2. Summary of soil chemical analyses by watershed. The number of low, moderate, and severely disturbed bottomland and upland sites included in the analysis are indicated. Values are mean ± std. dev. Different letters indicate significant differences at $\alpha <$ 0.05.

Watershed	Number of sites sampled			Mehlich Extractable				Water Ext.	Total		
	Low	Mod.	Severe	Ca mg kg^{-1}	Fe mg kg^{-1}	Al mg kg^{-1}	K mg kg^{-1}	C mg kg^{-1}	C g kg^{-1}	N g kg^{-1}	P mg kg^{-1}
Bottomlands											
Bonham	11	2	1	46 ± 43 ^{a}	246 ± 247 ^{a}	626 ± 659 ^{a}	27 ± 20 ^{a}	163 ± 101 ^{a}	34.2 ± 33.2 ^{a}	1.67 ± 1.53 ^{a}	181 ± 113 ^{a}
Halloca	10	3	0	251 ± 160 ^{a}	344 ± 392 ^{a}	386 ± 290 ^{a}	38 ± 18 ^{a}	88 ± 37 ^{b}	17.7 ± 9.8 ^{a}	1.10 ± 0.69 ^{a}	181 ± 105 ^{a}
Randall	14	5	0	136 ± 159 ^{a}	152 ± 93 ^{a}	459 ± 399 ^{a}	29 ± 14 ^{a}	77 ± 34 ^{bc}	22.2 ± 17.8 ^{a}	1.27 ± 0.99 ^{a}	118 ± 57 ^{a}
Sally Branch	7	4	2	84 ± 75 ^{a}	242 ± 181 ^{a}	556 ± 510 ^{a}	38 ± 22 ^{a}	39 ± 22 ^{bc}	29.2 ± 22.2 ^{a}	1.56 ± 1.25 ^{a}	191 ± 142 ^{a}
Shell	10	0	0	683 ± 512 ^{b}	193 ± 281 ^{a}	192 ± 85 ^{a}	73 ± 35 ^{b}	23 ± 4 ^{c}	13.3 ± 4.5 ^{a}	0.75 ± 0.24 ^{a}	137 ± 55 ^{a}
Wolf	12	1	0	46 ± 27 ^{a}	163 ± 119 ^{a}	654 ± 338 ^{a}	27 ± 10 ^{a}	56 ± 34 ^{bc}	29.0 ± 10.5 ^{a}	1.65 ± 0.67 ^{a}	190 ± 100 ^{a}
Uplands											
Bonham	9	18	8	30 ± 23 ^{a}	23 ± 10 ^{a}	169 ± 70 ^{a}	8 ± 7 ^{a}	131 ± 55 ^{a}	7.7 ± 3.5 ^{a}	0.30 ± 0.12 ^{a}	68 ± 27 ^{a}
Halloca	1	20	9	157 ± 213 ^{b}	42 ± 19 ^{b}	269 ± 118 ^{b}	29 ± 24 ^{b}	65 ± 26 ^{b}	8.8 ± 4.6 ^{ab}	0.36 ± 0.20 ^{a}	68 ± 38 ^{a}
Randall	14	19	3	73 ± 69 ^{ab}	37 ± 24 ^{b}	258 ± 101 ^{b}	16 ± 7 ^{ab}	62 ± 23 ^{bc}	9.7 ± 3.9 ^{ab}	0.34 ± 0.14 ^{a}	62 ± 30 ^{a}
Sally Branch	5	27	2	111 ± 169 ^{ab}	29 ± 12 ^{ac}	174 ± 79 ^{a}	22 ± 22 ^{ab}	41 ± 34 ^{cd}	10.9 ± 4.9 ^{bc}	0.37 ± 0.13 ^{a}	69 ± 25 ^{a}
Shell	16	8	2	532 ± 344 ^{c}	40 ± 19 ^{bc}	275 ± 67 ^{b}	84 ± 44 ^{c}	39 ± 23 ^{bd}	13.4 ± 4.5 ^{c}	0.62 ± 0.21 ^{b}	108 ± 36 ^{b}
Wolf	3	19	2	65 ± 51 ^{ab}	31 ± 12 ^{ab}	295 ± 101 ^{b}	17 ± 7 ^{ab}	34 ± 17 ^{d}	10.7 ± 4.0 ^{ac}	0.37 ± 0.12 ^{a}	74 ± 31 ^{a}

Table 3. Summary of soil chemical analyses for bottomland and upland sites of spatial study based on the level of disturbance.

Values are mean ± std. dev. Different letters indicate significant differences at α< 0.05, except Total Carbon α< 0.10

Impact	Mehlich Extractable				Water Ext.	Total		
	Ca mg kg^{-1}	Fe mg kg^{-1}	Al mg kg^{-1}	K mg kg^{-1}	C mg kg^{-1}	C g kg^{-1}	N g kg^{-1}	P mg kg^{-1}
Bottomlands								
Low	207 ± 309 [a]	216 ± 241 [a]	489 ± 444 [a]	38 ± 25 [a]	73 ± 52 [a]	25.2 ± 20.7 [a]	1.41 ± 1.06 [a]	168 ± 102 [a]
Moderate	93 ± 132 [a]	268 ± 212 [a]	552 ± 463 [a]	27 ± 18 [a]	95 ± 107 [a]	24.5 ± 18.2 [a]	1.29 ± 0.94 [a]	155 ± 96 [a]
Severe	105 ± 111 [a]	49 ± 53 [a]	200 ± 130 [a]	44 ± 40 [a]	75 ± 82 [a]	12.0 ± 8.8 [a]	0.56 ± 0.45 [a]	133 ± 111 [a]
Uplands								
Low	209 ± 290 [a]	38 ± 23 [a]	250 ± 79 [a]	36 ± 39 [a]	57 ± 38 [a]	11.2 ± 4.3 [a]	0.45 ± 0.20 [a]	83 ± 41 [a]
Moderate	132 ± 213 [a]	34 ± 15 [a]	239 ± 112 [ab]	25 ± 29 [a]	66 ± 48 [a]	10.4 ± 4.3 [a]	0.38 ± 0.16 [ab]	71 ± 32 [a]
Severe	111 ± 201 [a]	24 ± 16 [b]	189 ± 90 [b]	21 ± 26 [a]	75 ± 59 [a]	6.7 ± 4.4 [b]	0.29 ± 0.20 [b]	66 ± 24 [a]

Table 4. Correlation matrix for total P, C, N, WEC, and enzyme activities for bottomland soils from spatial study.

	TC g kg^{-1}	TN g kg^{-1}	WEC mg kg^{-1}	Acid Phosphatase Activity µg MUF/g dry wt. Soil/hr	β-Glucosidase Activity µg MUF/g dry wt. Soil/hr	Dehydrogenase Activity µg /g dry wt. Soil
Bottomlands						
TP mg kg^{-1}	0.4548	0.4697	-0.1465	0.0935	0.3267	0.1704
TC g kg^{-1}		**0.9488**	**0.6437**	0.3132	0.0886	**0.8558**
TN g kg^{-1}			**0.6367**	0.253	0.1101	**0.7464**
WEC mg kg^{-1}				-0.0482	-0.3058	**0.6323**
Acid Phosphatase Activity					**0.8742**	0.0295
β-Glucosidase Activity						-0.3075
Uplands						
TP mg kg^{-1}	0.4851	**0.6512**	-0.493	0.1647	-0.2022	0.1452
TC g kg^{-1}		**0.8145**	-0.2265	0.1895	-0.2226	**0.5545**
TN g kg^{-1}			-0.3382	0.3326	-0.1451	0.2743
WEC mg kg^{-1}				0.0969	0.0002	0.0583
Acid Phosphatase Activity					0.1835	-0.2635
β-Glucosidase Activity						**-0.6289**

Table 5. Summary of soil chemical analyses for bottomland and upland sites of transect study based on the level of disturbance. Data from transects include December 2000 and February 2002 samples. Values are mean ± std. dev. Different letters indicate significant differences at $\alpha < 0.05$. Outliers were discarded for means comparisons.

Impact	Mehlich Extractable				Water Ext.	Total		
	Ca mg kg^{-1}	Fe mg kg^{-1}	Al mg kg^{-1}	K mg kg^{-1}	C mg kg^{-1}	C g kg^{-1}	N g kg^{-1}	P mg kg^{-1}
Bottomlands								
Low	61.3 ± 27.7 ^{a}	227.4 ± 171.6 ^{a}	443.2 ± 229.8 ^{a}	30.4 ± 24.2 ^{a}	170.1 ± 135.1 ^{a}	111.4 ± 80.6 ^{a}	4.99 ± 3.44 ^{a}	310.8 ± 153.0 ^{a}
Moderate	29.8 ± 22.7 ^{b}	105.8 ± 198.9 ^{ab}	83.2 ± 116.9 ^{b}	8.4 ± 16.1 ^{b}	47.1 ± 45.2 ^{b}	26.1 ± 33.1 ^{b}	1.06 ± 1.55 ^{b}	68.1 ± 57.9 ^{b}
Severe	29.5 ± 17.3 ^{b}	24.3 ± 26.7 ^{b}	24.0 ± 17.7 ^{b}	2.1 ± 1.6 ^{b}	15.4 ± 7.9 ^{b}	3.9 ± 1.8 ^{b}	0.16 ± 0.08 ^{b}	34.1 ± 8.8 ^{b}
Uplands								
Low	62.8 ± 56.3 ^{a}	24.2 ± 18.4 ^{a}	131.8 ± 144.5 ^{a}	8.2 ± 11.7 ^{a}	21.0 ± 9.2 ^{a}	10.2 ± 2.2 ^{a}	0.29 ± 0.10 ^{a}	59.1 ± 8.3 ^{a}
Moderate	92.4 ± 69.0 ^{a}	33.7 ± 41.6 ^{a}	183.1 ± 226.8 ^{a}	10.1 ± 13.6 ^{a}	12.8 ± 4.7 ^{ab}	7.9 ± 4.9 ^{a}	0.43 ± 0.40 ^{a}	58.2 ± 8.2 ^{a}
Severe	61.7 ± 12.7 ^{a}	10.2 ± 3.1 ^{a}	73.2 ± 17.4 ^{a}	4.9 ± 2.2 ^{a}	9.7 ± 7.5 ^{b}	2.1 ± 0.4 ^{b}	0.10 ± 0.04 ^{b}	55.1 ± 3.6 ^{a}

Table 6. Comparison of normalized enzyme activity levels for December 2000 and February 2002 sampling dates for bottomland and upland soils. Different letters between dates indicates significant differences at $\alpha < 0.05$.

Landscape Position	Date	Acid Phosphatase	B-glucosidase	Dehydrogenase
Bottomland	**Dec. '00**	0.27 ± 0.28 [a]	0.25 ± 0.21 [a]	0.39 ± 0.24 [a]
	Feb. '02	0.17 ± 0.15 [a]	0.25 ± 0.19 [a]	0.17 ± 0.09 [b]
Upland	**Dec. '00**	0.23 ± 0.20 [a]	0.64 ± 0.13 [a]	0.69 ± 0.17 [a]
	Feb. '02	0.10 ± 0.04 [b]	0.47 ± 0.30 [b]	0.37 ± 0.11 [b]

Table 7. Controlling factors for enzyme activities in bottomland and upland soils from December 2000 and February 2002 . The table indicates significant effects contributing to β-glucosidase and dehydrogenase enzyme activities in the two landscape positions as identified by the following model using a Least Squares Fit of data: Enzyme Activity = date + % moisture + TC + date*TC + TC*% moisture + date*% moisture + date*TC*% moisture; where date = date of sampling, % moisture = soil water content, and TC = total carbon. Significant effects are reported as p > F values. NS = not significant.

	Effects							R^2
	Sampling Date	**% moisture**	**TC**	**Date*TC**	**TC* % moisture**	**Date* % moisture**	**Date*TC *% moisture**	
Bottomland β-glucosidase	**0.0076**	**<.0001**	**0.0492**	**0.0002**	**<.0001**	**0.001**	**0.0287**	**0.96**
Bottomland Dehydrogenase	**<.0001**	**<.0001**	**NS**	**NS**	**<.0001**	**0.0007**	**NS**	**0.98**
Upland β-glucosidase	**<.0001**	**NS**	**NS**	**<.0001**	**NS**	**0.0033**	**0.0348**	**0.42**
Upland Dehydrogenase	**<.0001**	**NS**	**<.0001**	**<.0001**	**0.0872**	**NS**	**NS**	**0.81**

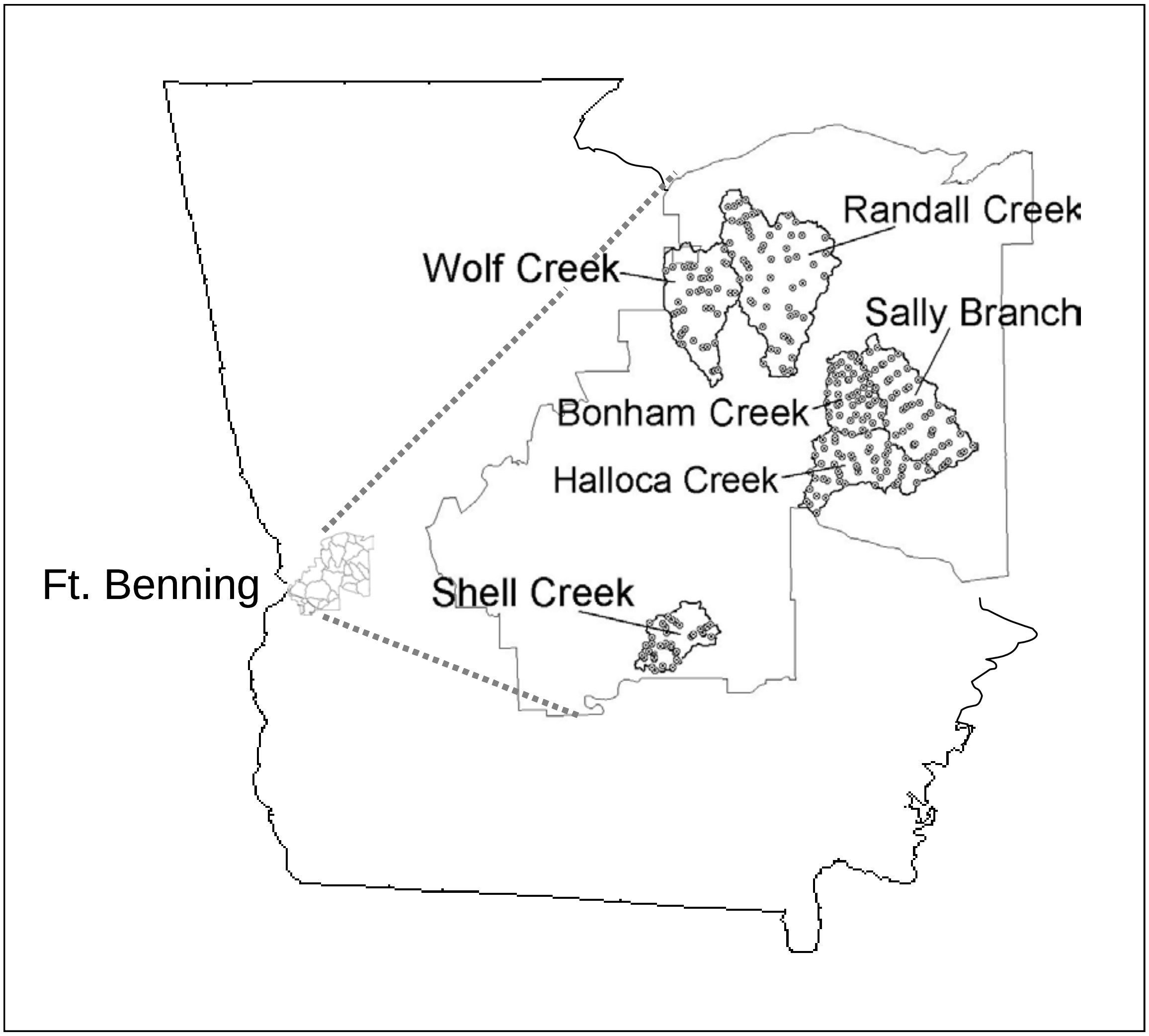


Figure 1.
Six watersheds of the Ft. Benning Military Installation from which soil samples were obtained during January to August 2000. Dots indicate sampling points along transects across watersheds at three landscape positions (Ridgetop, mid-slope, and bottomland).

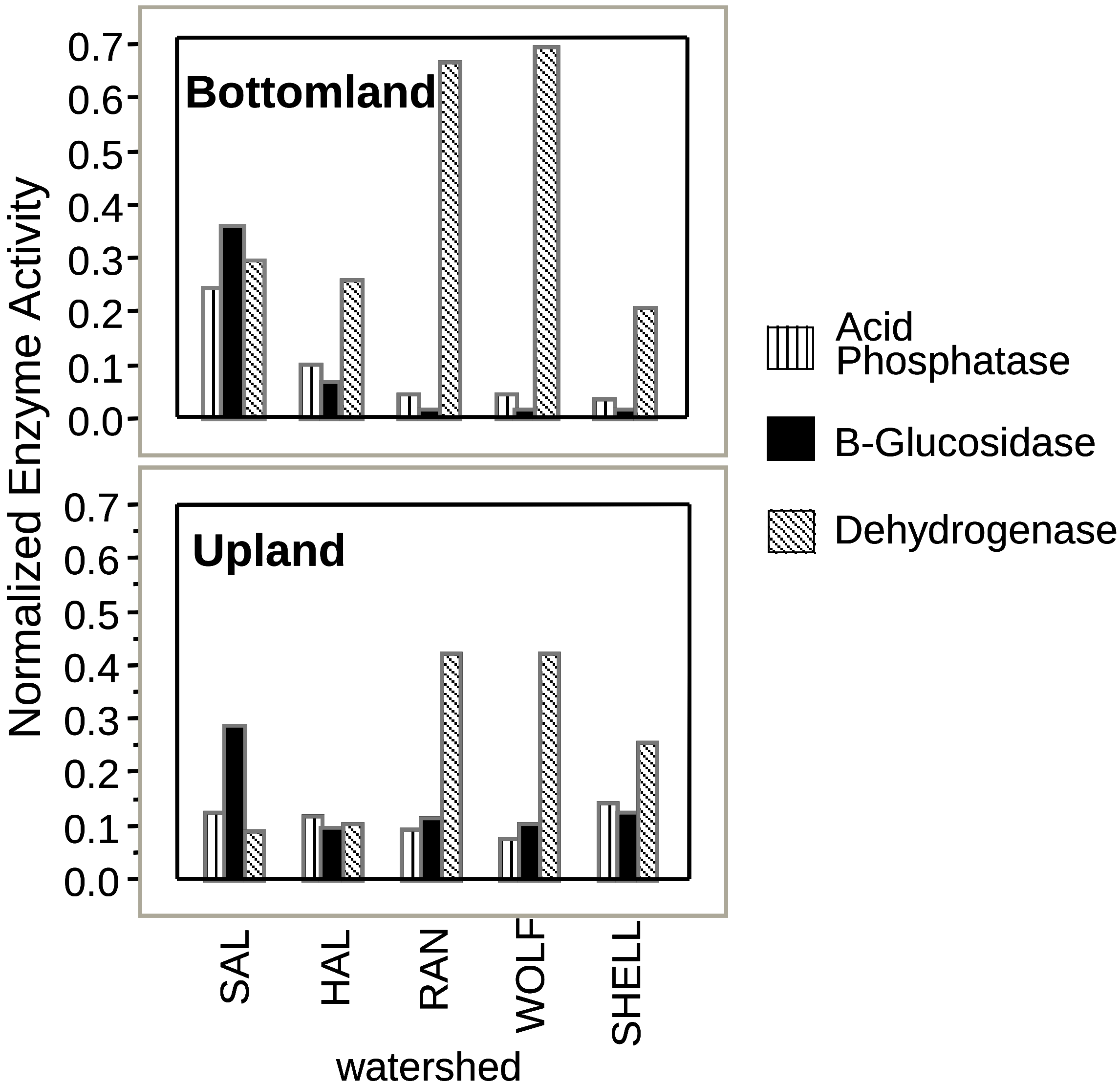


Figure 2.
Mean relative normalized activity of three microbial enzymes in five of the six watersheds sampled. Sampling occurred during the following months: Sally Branch (SAL), March ’00; Halloca Creek (HAL), May ’00; Randall Creek (RAN), June ’00; Wolf Creek (WOLF), June ’00; and Shell Creek (SHELL), August ’00.

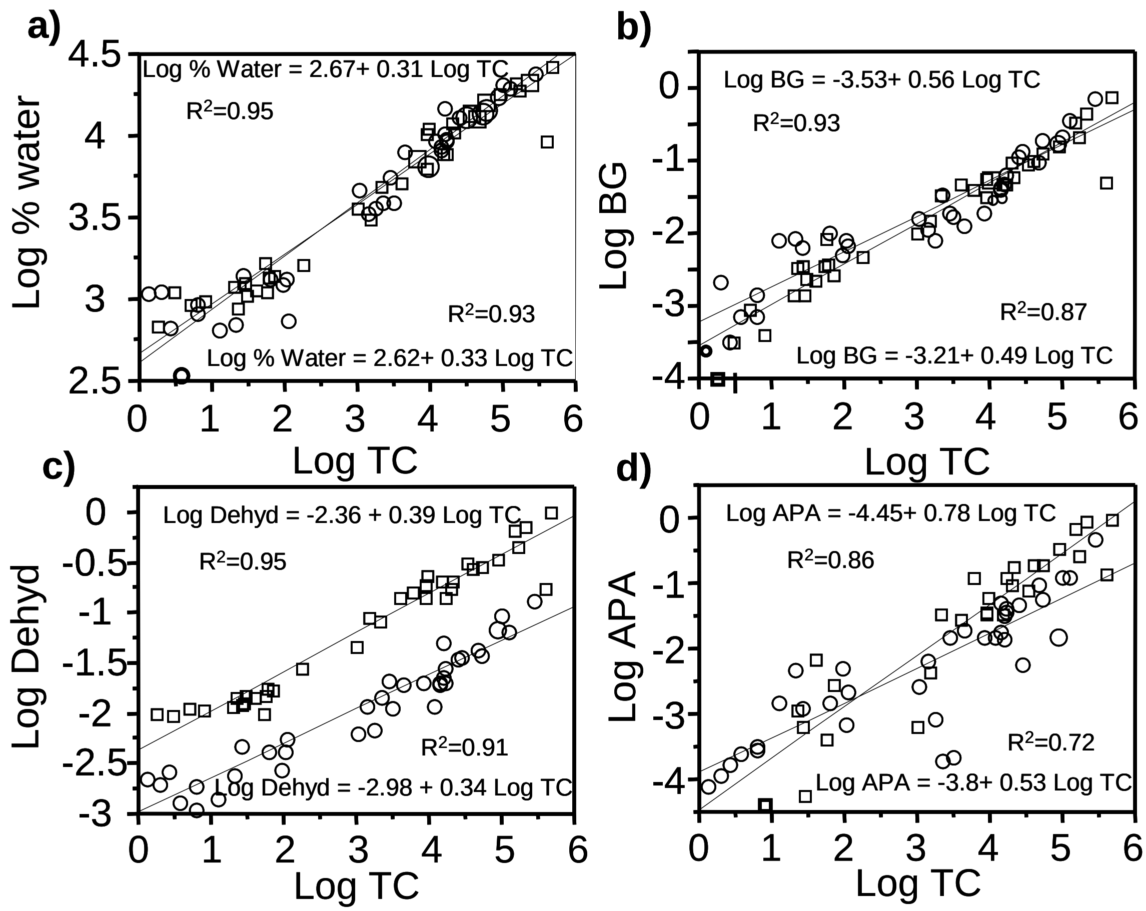


Figure 3.

Relationship between the log TC values and a) log percent soil moisture; b) β-glucosidase activity; c) dehydrogenase activity; and d) acid phosphatase activity. Data is from December 2000 (solid squares) and February 2002 (open circles) of the transect study. Trend lines and R^2 values are are shown for the two sampling times.

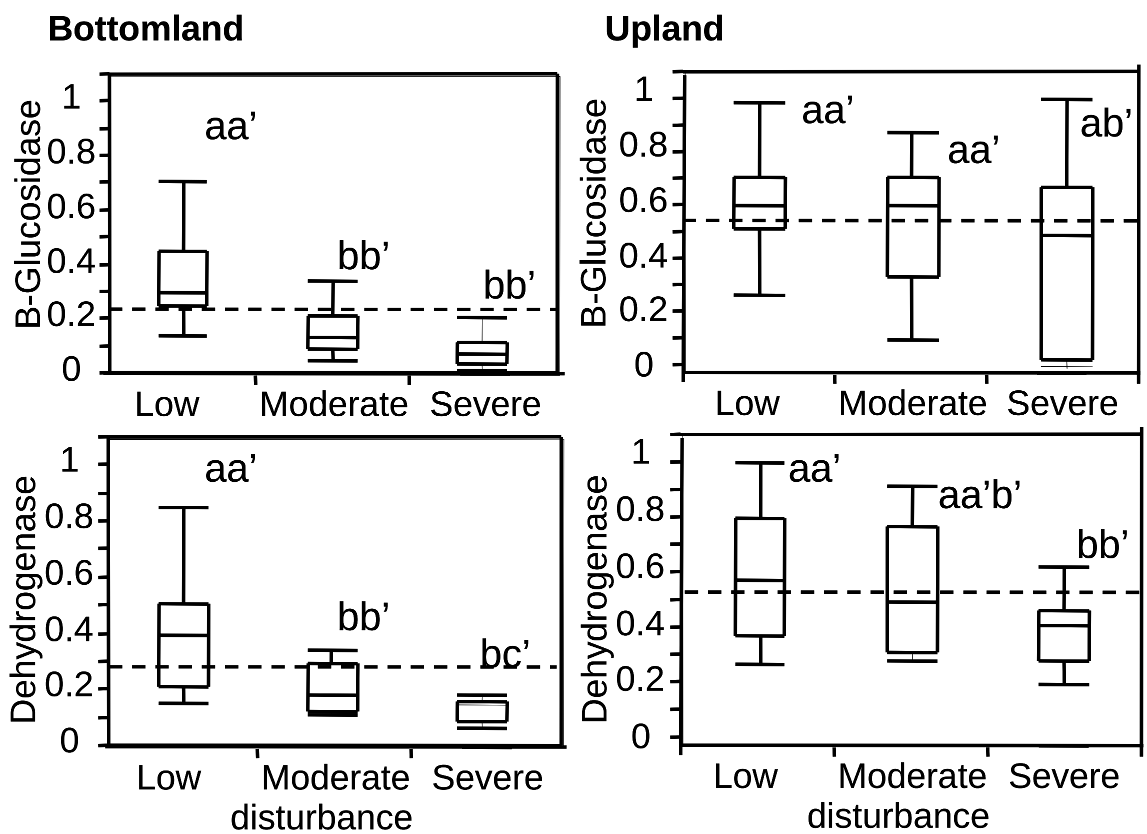


Figure 4.
Comparison of normalized β-glucosidase and dehydrogenase values for soils from Low, Moderate, and Severe disturbance bottomland and upland transects. Separate means comparisons were performed for December and February (prime) values; different letters indicate significant differences ($\alpha < 0.05$ except February bottomland dehydrogenase $\alpha < 0.10$). Dashed line indicates mean for all samples. Median and quantiles are indicated for data from both sampling dates combined. The line in the body of each box represents the median; top and bottom of the box represent the 75th and 25th quantiles; the lines above and below each box represent the 10th and 90th quantiles.

3.1.3
Distribution of Methanotrophs in Managed and Highly Degraded Watersheds.
Ogram, A., H. Castro, E.A. Stanley, W. Chen, and J. Prenger.

ABSTRACT

Potential impacts of infantry and tank training activities on the distribution of methanotrophs were investigated in two watersheds on the Ft. Benning US Army training facility in west Georgia, USA. The Bonham Creek watershed shows significant impacts from intensive infantry and tank training exercises, including severe erosion and deposition of sediments in bottomlands. The Sally Branch watershed is a managed watershed with limited exposure to tank training exercises and much less erosion. Sequence analysis of *pmoA* clone libraries constructed from DNA extracted from Sally Branch bottomland and upland samples were dominated by a deeply rooted lineage related to Type I methanotrophs (the "Benning soil cluster γ," BSC γ). Sequences clustering with known Type II methanotrophs were restricted to samples taken from the Sally Branch bottomland sample. Terminal restriction fragment length polymorphism (T-RFLP) analysis of *pmoA* was applied to samples taken from transects located in upland and bottomland sites within the two watersheds. Observed T-RFs matched well with T-RFs predicted from sequences obtained from the clone library, with few exceptions. Principal components analysis revealed that most T-RFLPs from upland samples clustered separately from bottomland samples in both watersheds. Upland and bottomland T-RFLPs clustered separately in the Sally Branch transect; however, some Bonham Creek bottomland T-RFLPs clustered within the upland cluster, suggesting mixing of upland with bottomland soils in this watershed. Assemblages were, in general, similar between the eroded and managed watersheds, although erosion likely resulted in mixing of upland assemblages with bottomland assemblages in the Bonham Creek watershed.

INTRODUCTION

The impact of land use practices on the structure and function of microbial communities is currently of great interest, particularly with regard to potential effects on nutrient cycling, carbon sequestration, and as indicators of ecosystem integrity. Among those microbial groups of interest are the methanotrophs, methane oxidizing bacteria that are widespread in most soils. Methanotrophs are important sinks for the almost 700 Tg methane emitted annually from various sources (23), and are an important part of the carbon cycle in many soils.

Most methanotrophs can be divided into two broad physiological and phylogenetic groups, the Type I and Type II methanotrophs. Type I methanotrophs belong to the γ-proteobacteria, and Type II are members of the α-proteobacteria (8). Both types coexist in many environments, although Type I methanotrophs typically dominate in high oxygen, low methane environments (11), and Type II methanotrophs typically dominate under high methane conditions. Representatives of both groups have the ability to fix nitrogen (1). Both types have been found in forests, although Type I methanotrophs have been reported to be more active than Type II strains in some forests (15). With one notable exception, (7), both types harbor characteristic forms of the particulate methane monooxygenase gene (*pmoA*), and Type II species also encode a soluble methane monooxygenase. These genes are sufficiently conserved between representative strains to be of use in constructing phylogenies that compare well with 16S rRNA phylogenies (1,5).

The availability of conserved genes such as those encoding the particulate and soluble methane monooxygenases make them attractive targets for non-culture based studies of the distribution and activities of methanotrophs in soils. Distribution of these genotypes has been studied in a variety of environments, including forests (15), peat bogs (16, 19), aquifers (20) and rice paddies (9). Non-culture based approaches have led to the discovery in forest soils of novel methanotrophic lineages that branch deeply within either the *pmoA* phylogenies of Type I or Type II groups (4,10,15), and have provided great insight into the activities and ecological functions of methanotrophs in upland and wetland soils. The relatively high diversity of methanotrophs, the availability of conserved genetic sequences, and their importance in ecosystem functioning make methanotrophs strong candidates for indicator organisms in studies on the effects of landuse on ecosystem properties.

Ecosystem stability is of great concern at military training facilities, particularly those involved in intensive training in forests with rolling terrain. Movement of troops and tanks across hillsides accelerates deforestation and associated processes, such as erosion. Decreased vegetation and increased erosion would significantly change rhizosphere nutrient dynamics and soil porosity that may impact rates of methanogenesis, and subsequently affect activities and compositions of assemblages of methanotrophs.

Ft. Benning, a large US Army base in western Georgia, USA, is home to a large infantry and armored warfare training facility. The entire area has been subjected to farming and disturbance since the early 19th century, but since incorporation into the military base many areas are recovering and are therefore relatively undisturbed. These relatively undisturbed areas of the base are adjacent to areas that have been highly impacted by training exercises (3,22). In this study, we investigated relationships between the degree of recent impact and the distribution of methanotrophs in a low impact and high impact watershed pair. The Sally Branch watershed has not been subjected to extensive training, and areas are managed as pine plantations. Bonham Creek watershed is highly impacted, with a high degree of erosion observed in upland sites with concomitant sedimentation in bottomland sites. A non-culture based approach, terminal restriction fragment length polymorphism (T-RFLP) analysis of the *pmoA* gene, was used to screen samples taken from two transects (upland and bottomland) in each watershed for comparison of potential effects of military training on methanotrophic assemblages.

MATERIALS AND METHODS

Site description and sampling

The study area is within the Ft. Benning military reservation in west-central Georgia (Fig. 1), in the Carolina and Georgia sand hills major land resource area (24). Upland soils in the area are primarily well- to excessively drained Ultisols and Entisols, supporting forests of slash (*Pinus elliotii*), longleaf (*P. palustris*), and loblolly (*P. taeda*) pines. Excessively-drained Lakeland soils (Entisol) of sandhill communities are associated with longleaf pine, turkey oaks (*Quercus laevis*), blackjack oaks (*Q. marilandica*), and post oaks (*Q. stellata*) near ridgetops in the central and northern portion of the reservation. Wetlands and hydric soils are generally restricted to bottomlands along streams and creeks. Military related impacts result from the direct removal of or damage to vegetation, digging activities, and ground disturbance from vehicles. The mechanized forces in particular use tracked and wheeled vehicles that cause soil disturbance and movement that may result in soil erosion and stream sedimentation. This study focused on two watersheds characterized by different land uses.

High and low impact regions were identified by land use, degree of disturbance and characteristic vegetation. The Bonham Creek watershed had been subjected to extensive tank warfare training, and was considered to be highly impacted. The Sally Branch watershed had not been subjected to mechanized training, and was considered to be minimally impacted. One upland transect and three bottomland transects were selected within each watershed (Table 1). Upland transects were 400 m long and soil samples were collected every 20 meters. Bottomland transects were 25 m long and samples were collected every 5 meters perpendicular to the creek. High and low impact bottomland transects were located downslope from the high and low impact upland transects, respectively. Five 20-cm deep cores were taken from each upland sampling site and manually homogenized. Similarly, five replicate 10-cm deep cores were taken at each bottomland site. A total of 80 samples were collected and manually homogenized, and samples were sub-sampled from composite samples and stored at –80 °C. Soil characteristics were provided by the Wetland Biogeochemistry Lab at University of Florida.

DNA extraction from soil.

DNA was directly extracted and purified from 250 mg soil samples with UltraClean Soil DNA kit (MoBio Laboratories, Solana Beach, CA.) according to manufacturer's instructions. The mass of DNA within 5 μl of extracted DNA was estimated by electrophoresis on 0.7% agarose gel stained in ethidium bromide. DNA was aliquoted and stored at -20°C.

PCR amplification of pmoA.

PCR were conducted using the primer sets A189 (12) combined with A650 (4) or mb661 (5). 20 μl PCR reactions were set up containing approximately 1 μl of soil DNA, 0.4 μM of each primer, 7.4 μl of distilled water, and 10μl × HotStarTaq Master Mix (Qiagen, Valencia, CA). PCR was carried out in a GeneAmp PCR system 2400 (Perkin-Elmer Applied Biosystems, Norwalk, CT) with the following cycling parameters: 15 min at 95°C, 30 cycles of 30 sec at 94°C, 30 sec at 55°C, 30 sec at 72°C, with a final extension step at 72°C, 7 min. PCR products were electrophoresed through 1% agarose gels to confirm the size.

Cloning, sequencing, and phylogenetic analysis.

Clone libraries for *pmoA* from representative low impact bottomland and low impact upland soils were constructed using the TOPO TA cloning kit (Invitrogen, Carlsbad, CA) according to the manufacturer's instructions. The clone libraries were screened by restriction fragment length polymorphism (RFLP) analysis using *Hha*I digestion of PCR amplification products of clones screened with the corresponding primer set. Clones representative of unique RFLP patterns were selected and their plasmid DNAs were purified and sequenced using cloning vector primers (TOPO TA cloning kit, Invitrogen, Carlsbad, CA) at the DNA Sequencing Core Laboratory at the University of Florida.

Sequences were aligned against available genes in the National Center for Biotechnology Information nucleotide database (NCBI; http://www.ncbi.nlm.nih.gov/) using the Pileup function of the Genetic Computer Group sequence analysis package (GCG; Madison, WI). Phylogenetic analysis was constructed using maximum parsimony with PAUP* version 4.0b8 (D. L. Swofford, Sinauer Associates, Sunderland, MA) and neighbor-joining using the Jukes and Cantor algorithms (25). Bootstrap analyses were conducted with 100 resamplings. Only branches with bootstrap values higher than 50% are shown. *pmoA* sequences for Sally Branch upland clones are listed under the GenBank accession numbers AY662347 through AY662367, and sequences for Sally Branch bottomland clones are listed under accession numbers AY662368 through AY662389. The sequences were later used in selection of restriction enzymes for T-RFLP.

T-RFLP analysis.

PCRs were performed on 20 μl reactions in triplicate using A189 and fluorescently labeled A650 with 6-FAM (6 carboxyfluorsecein) in the 5'position (Invitrogen, Carlsbad, CA). PCRs were carried out using the following cycling parameters: 15 min at 95°C, 30 cycles of 30 sec at 94°C, 30 sec at 49°C, 30 sec at 72°C, with a final extension step at 72°C, 7 min. The triplicate PCR reactions were pooled, and the pooled reactions were cleaned and concentrated using QIAquick PCR purification kit (Qiagen, Valencia, CA) to a final volume of 30 μl following the manufacturer's instructions. Restriction enzymes were selected *in silico* based on manually aligned sequences using CloneMap version 2.11 (CGC Scientific Inc., Ballwin, MO) on each sequence and the enzyme producing the greatest discrimination between sequences was selected for use in t-RFLP (the restriction enzyme with greater degree of discrimination for *pmoA* was *Hha*I). Approximately 50 ng of amplified PCR was added to the mixture with approximately 3 U (0.3 μl) of restriction enzyme (Promega, Madison WI), 1 μl restriction buffer, 1 μg bovine serum albumin, and deionized water to make a final volume of 10 μl. Enzymatic digestions were carried out at 37°C overnight.

One microliter of digestion product was used in terminal restriction fragment (t-RF) detection by the DNA Sequencing Core Laboratory (University of Florida). Briefly, 1 μl of digested DNA was mixed with 2.5 μl deionized formamide, 0.5 μl ROX-labeled GenScan 500-bp internal size standard (Applied Biosystems, Perkin Elmer Corporation, Norwalk, CT) and 0.5 μl of loading buffer (50 mM EDTA, 50 mg/ml blue dextran). The samples were denatured by heating at 95 ˚C for 3 minutes and immediately transferred to ice. One μl of denatured digests were electrophoresed in a 36 cm, 5% polyacrylamide gel containing 7 M urea for 3 h at 3000V on an ABI 377 Genetic Analyzer (Applied Biosystems, Perkin Elmer Corporation, Norwalk, CT) with filter set A and a well-to-read length of 36 cm. T-RFLP profiles were analyzed using GeneScan version 2.1 software (Applied Biosystems, Perkin Elmer Corporation, Norwalk, CT). The sizes in base pairs (bp) of terminal restriction fragments were estimated by reference to internal standard by using Local Southern method. T-RFLP results, including peak size, peak fluorescence height and area, and scan size were exported to Excel (Microsoft Corporation, Redmond, WA) for data analysis.

T-RFLP Data Analysis

Principal components analysis (PCA) was performed using the relative abundance of individual peaks normalized by the total area of all T-RFs with the Multivariate Statistical Package (MSVP version 3.12d, Kovach Computing Services, Wales, UK). All T-RFs observed were included in this analysis.

Relationships between the frequency of individual T-RFs and various site characteristics were statistically analyzed and plotted by PC SAS (SAS Inst., 2003). GLM analysis for the significant differences of means was performed. Significant differences were achieved at the 0.05 level. Population normality was tested for each variable before using parametric statistics for comparisons and testing.

RESULTS AND DISCUSSION

pmoA clone libraries:

pmoA clone libraries were constructed from samples representative of the Sally Branch upland and bottomland transects (Fig. 2). Sally Branch soils were chosen for library construction due to their low level of disturbance. Low disturbance sites are generally characterized by higher biological diversity than are high disturbance sites, and it is likely that most genotypes observed in Bonham Creek samples are present in the Sally Branch samples. This assumption was confirmed via T-RFLP analysis, presented below in the T-RFLP section.

The great majority (31 out of 41 sequenced) of clones grouped in Clusters I, II, and III and were not associated with cultured representatives (Fig. 2). The closest cultured relative for these sequences is *Methylocaldum gracile*, a Type I methanotroph in Cluster IV that branches outside of the major branch for Clusters 1, II, and III. The deep branch isolating Clusters I, II, and III from Cluster IV is supported by strong bootstrap values and indicates a major lineage within the Type I methanotrophs. These clusters, collectively termed the “Benning Soil Cluster γ” (BSC γ), branch outside the recently reported “upland soil cluster γ,” (uscγ)(15), which falls in Cluster IV of our tree (Fig. 2).

Representatives of this cluster have been reported previously (1,4), but this is the first report of a library dominated by these sequences.
Representatives of upland and bottomland libraries were segregated in most major branches, e.g. Cluster Ia is comprised exclusively of upland clones, and Cluster IIa is dominated by bottomland sequences. Type I methanotrophs have been reported to outcompete Type II methanotrophs under conditions of high oxygen and low methane (11), as might be expected from upland soils.

Five clones out of forty-one sequenced grouped within Cluster V, defined by the Type II methanotrophic genera, *Methylosinus* and *Methylocystis*. This group is not closely related to the previously described "forest sequence cluster," a significant group of uncultured Type II methanotrophs identified in European soils (4,10). Cluster V is exclusively comprised of bottomland clones; Type II methanotrophs typically dominate Type I methanotrophs in areas with higher methane concentrations, as might be expected for bottomlands (8). One bottomland clone (LIW- 516; Cluster VI) branched outside of Type I and Type II clusters, and was not associated with previously cultivated strains.

Most previous forest soil *pmoA* libraries have been constructed from European soils (13, 10) that contained much greater organic carbon contents than did the soils of this study (Table 1). Organic carbon content of soil is roughly correlated with methanogenesis rates, and it is likely that methane produced in this soil is low relative to most previous studies. Sandy, well drained, low organic carbon soils such as those of the Ft. Benning watersheds are likely to be characterized by low CH_4:O_2 concentration ratios relative to finer textured soils reported in other studies, which may select for methanotrophs with high affinities for methane. Bender and Conrad (2) hypothesized that, although low-affinity methanotrophs (micromolar K_m) dominate culture collections, high-affinity methanotrophs (nanomolar K_m) may be largely responsible for much of the methane oxidation that occurs in soils. A number of studies have demonstrated that Type I methanotrophs are active under low CH_4:O_2 concentration ratios (11,15). It is possible that BSCγ methanotrophs are particularly well adapted to the low methane concentrations likely to be present in the very well drained, low organic carbon forest soils of the southeastern United States. Without culturable relatives, it is not possible to make inferences regarding possible ecological roles for these groups beyond general inferences for Type I methanotrophs.

T-RFLP analysis:

Construction of *pmoA* clone libraries for all 70 samples taken from the four transects was not feasible due to the time expense that would be required for such an undertaking. T-RFLP is a useful approach to screening distributions of genotypes in large numbers of samples (14,17,18). While subject to limitations (6,21), T-RFLP analysis can provide information on complex assemblages of functional genotypes. Sequences of *pmoA* clones presented in Fig. 2 were used to design and evaluate a T-RFLP system for screening samples from the Sally Branch and Bonham Creek watersheds. Predicted T-RF lengths for individual clones are presented in Fig. 2 immediately to the right of the clones, and groupings within major clusters (e.g., Clusters Ia, IIa, IIIa, etc) represent clones predicted to yield similar T-RFs (+/- 2 bp). Predicted T-RFs were consistent with major phylogenetic groupings, although some overlap was observed between individual clones and broader groups: clone LIW-03 (Cluster I) shares a predicted T-RF of 36 bp with Cluster IIa; clone LIW-11 (Cluster V) shares T-RF 128 bp with Cluster Va; clone LIU-09 (Cluster III) shares T-RF 138-139 bp with clone LIW-517 (Cluster IV); and Clone LIW-11 (Cluster V) shares T-RF 128 bp with Cluster Va. In general, however, this T-RFLP system provides good resolution between major subgroups observed in the library.

T-RFLP analyses were conducted on a total of 71 soil samples (Table 1): from the Bonham Creek watershed, 13 samples were analyzed from the upland transect, and 16 from the bottomland transect; for Sally Branch watershed, 18 samples were analyzed from the upland transect, and 23 from the bottomland transect; and one highly eroded reference site from Bonham Creek (RH11). A total of 21 T-RFs from all samples were observed, with numbers of T-RFs ranging from 5 to 13 for individual samples. Most T-RFs predicted from the library were observed, although 5 primarily minor predicted groups were not observed: T-RFs 78, 114, 127, 128, and 131. The absence of these predicted T-RFs from observed T-RFLPs was likely due to lower sensitivity of T-RFLP than clone libraries. These strains were undoubtedly present in the samples because they were present in the clone libraries, but T-RFLP is not sufficiently sensitive to detect low abundance groups. Conversely, five T-RFs not predicted from the clone libraries were observed in T-RFLPs (Table 2).

Principal components analysis (PCA) was applied to T-RFLP data from the two watersheds, and significant differences between PCAs of methanotrophic assemblages within a given watershed were observed (Fig. 3). Upland samples were completely separated from bottomland samples in the Sally Branch watershed (Fig. 3A), although the bottomland samples formed two distinct clusters (1 and 2) that were largely independent of vegetation type, but no mixing of bottomland and upland samples was observed. This is contrasted with the Bonham Creek watershed, where significant overlap between bottomland and upland T-RFLPs was observed (Fig. 3B). Two separate clusters comprised exclusively of several bottomland samples were observed (Clusters 1 and 2), but several bottomland samples also grouped among upland samples (Cluster 3).

These general trends are supported when all data are combined (Fig. 3C); all bottomland samples cluster in two separate groups (Clusters 1 and 2), with some mixing of Bonham Creek bottomland samples within the diffuse upland cluster (Cluster 3). The separation of T-RFLPs into bottomland and upland clusters, independent of watershed, indicates that land management effects on the distribution of methanotrophs were not as strong as landscape position at these sites. These data suggest that methanotrophs in these watersheds are relatively stable, and that the eroded conditions are not significantly different from the managed areas to select for different groups of methanotrophs. Species presence does not imply activities (15), however, and it is quite possible that different groups are responsible for oxidation of methane in the different watersheds.

It is not known at this time why the bottomland assemblages of both watersheds split into two divergent clusters. Cluster 1 of the Bonham Creek watershed (Fig. 3B) was comprised of adjacent samples taken between ecotones (BB30-BB35; Table 1), and Cluster 2 (BB14, BB20, 21) were also adjacent samples, although not from similar vegetation. The Sally Branch bottomland Cluster 1 was composed of samples taken from adjacent sites (SB12, SB20, SB22, and SB24) within a depression, and Cluster 2 (SB23, SB34, and SB33) included samples from contiguous ecotones, although SB 23 appears to be an outlier from adjacent samples that grouped in Cluster 1.

The robustness of these groupings is not clear at this time. Groupings assigned to these PCAs were subjective, and one could imagine other ways to group the data points. It is clear, however, that considerable overlap exists between the Bonham Creek bottomland and upland points at the center of Figs. 3B and 3C, and that mixing is not observed in Sally Branch. Additional data from Sally Branch bottomlands would be of great value in testing these groupings.

Differences between the Sally Branch and Bonham Creek PCAs are likely related to difference in land use impacts between the two watersheds. Erosion of Bonham Creek uplands has been well documented (3,22), with transport of soil down slopes into bottomlands. Transport of soil bacteria, including methanotrophs, would be expected to occur with erosion, resulting in mixing of upland with bottomland communities. Uplands and bottomlands in forested watersheds not subjected to significant amounts of erosion would be more completely separated, as observed with the Sally Branch PCA.

Correlation of relative frequencies of individual T-RFs with environmental parameters, including transect, subjective level of disturbance, dominant plant community, and watershed is presented in Table 2. Individual T-RFs within the expected +/- 2 bp margin of error are presented as a single T-RF if both T-RFs exhibit similar trends. If trends between T-RFs with +/- 2 bp exhibited different trends (e.g., T-RFs 66 and 68; Table 1), both T-RFs are presented. Few T-RFs were significantly correlated at the P=0.05 level with individual parameters, although a number T-RFs showed clear trends.

The strongest correlations among predicted T-RFs were observed for: T-RF 23 (Cluster IIIc), which showed a clear enrichment in grassy patches of the upland Bonham Creek transect relative to Bonham Creek bottomlands or the managed pines of the Sally Branch upland transect; T-RF 68 (Cluster IIId), positively correlated with the bottomlands of both watersheds; and T-RFs 107 (Cluster Ia) and 139 (Cluster IV), positively correlated with the upland transects of both watersheds. Even though the transects crossed a number of plant community types (Table 1), the relative frequencies of most individual T-RFs did not correlate strongly with individual plant community types (Table 2). A few significant correlations between T-RF frequency and level of disturbance or general sample type was noted, e.g. the negative correlation between T-RF 23 (IIIC) and planted pines (Table 2); however, this was not typical.

These data indicate that individual genotypes were present in both transects, and that the distinctions between the different PCA clusters outlined in Fig. 2 were likely due to the presence or absence of a few genotypes in each assemblage. No significant differences in the frequencies of individual T-RFs were observed between the Bonham Creek and Sally Branch transects (Table 2), although a few T-RFs correlated with combined upland or bottomland transects. This indicates that our preliminary assumption that these Bonham Creek and Sally Branch could serve as paired watersheds for the purpose of comparison was appropriate.

CONCLUSIONS

T-RFLP analysis of *pmoA* sequences in DNA from soil samples taken from transects in two watersheds at Ft. Benning suggests impacts of land use on the distribution of methanotrophs largely resulted from transport of upland methanotrophs to bottomlands as a result of erosion in the highly impacted Bonham Creek watershed.

Upland and bottomland assemblages in the relatively undisturbed Sally Branch watershed were distinct, with no evident mixing. T-RFLPs from Sally Branch and Bonham Creek upland transects clustered together, indicating that land use differences between the two watersheds did not significantly impact the distribution of methanotrophs. Implications for impact of erosion on functioning of methanotroph assemblages suggest the likelihood of diminished methane oxidation rates as strains adapted to upland conditions are transported to bottomlands. Transects in both watersheds crossed a number of different plant community types, although the composition of methanotroph assemblages did not depend on the dominant vegetation or watershed. More work is required to investigate relationships between land use, transport of strains, methanogenesis, and methane oxidation in highly degraded forest soils.

REFERENCES

Auman, AJ, Stolyar, S, Costello, AM, Lidstom, ME (2000) Molecular characterization of methanotrophic isolates from freshwater lake sediment. Appl. Environ. Microbiol. 66:5259-5266.

Bender, M, Conrad, R (1992) Kinetics of CH_4 oxidation in oxic soils exposed to ambient air or high CH_4 mixing rations. FEMS Microbiol. Ecol. 101:261-270.

Bhat, S, Jacobs, JM, Hatfield, K, Prenger, J. Ecological indicators in forested watersheds: Relationships between watershed characteristics and stream water quality in Fort Benning, GA. Submitted to Ecological Indicators.

Bourne, DG, McDonald, Murrell, JC (2001) Comparison of *pmoA* PCR primer sets as tools for investigating methanotroph diversity in three Danish soils. Appl. Environ. Microbiol. 67:3802-3809.

Costello, AM, Lidstrom, ME (1999) Molecular characterization of functional and phylogenetic genes from natural populations of methanotrophs in lake sediments. FEMS Microbiol. Ecol. 65:5066-5074.

Dunbar, J, Ticknor, LO, Kuske, CR (2001) Phylogenetic specificity and reproducibility and new method for analysis of terminal restriction fragment profiles of 16S rRNA genes from bacterial communities. Appl. Environ. Microbiol. 67: 190-197.

Dunfield, PF, Khmelenina, VN, Suzina, NE, Trotsenko, YA, Dedysh, SN (2003) *Methylocella silvestris* sp. nov., a novel methanotroph isolated from an acidic forest cambisol. Int. J. Syst. Evol. Microbiol. 53: 1231 - 1239.

Hanson, RS, Hanson, TE (1996) Methanotrophic bacteria. Microbiol. Rev. 60:439-471.

Henckel, T, Friedrich, M, Conrad, R (1999) Molecular analyses of the methane oxidizing microbial community in rice field soil by targeting the genes of the 16S rRNA, particulate methane monooxygenase, and methanol dehydrogenase. Appl. Environ. Microbiol. 65:1980-1990.

Henckel, T, Jackel, U, Schnell, Conrad, R (2000) Molecular analyses of novel methanotrophic communities in forest soil that oxidize atmospheric methane. Appl. Environ. Microbiol. 66:1801-1808.

Henckel, T, Roslev, P, Conrad, R (2000) Effects of O_2 and CH_4 on presence and activity of the indigenous methanotrophic community in rice field soil. Environ. Microbiol. 2:666-679.

Holmes, AJ, Costello, Lidstrom, ME, Murrell, JC (1995) Evidence that particulate methane monooxygenase and ammonia monooxygenase may be evolutionarily related. FEMS Microbiol. Lett. 132:203-208.

Holmes, AJ, Roslev, P, McDonald, IR, Iversen, N, Henriksen, K, Murrell, JC (1999) Characterization of methanotrophic bacterial populations in soils showing atmospheric methane uptake. Appl. Environ. Microbiol. 65:3312-3318.

Horz, HP, Rotthauwe, JH, Lukow, T, Liesack, W (2000) Identification of major subgroups of ammonia-oxidizing bacteria in environmental samples by T-RFLP analysis of *amoA* PCR products. J. Microbiol. Meth. 39:197-204.

Knief, C, Lipski, Dunfield, PF (2003) Diversity and activity of methanotrophic bacteria in different upland soils. Appl. Environ. Microbiol. 69: 6703-6714.

Kravchenko, IK (2002) Methane oxidation in boreal peat soils treated with various nitrogen compounds. Plant and Soil 242:157-162.

Liu, WT, Marsh, TL, Cheng, H, Forney, LJ (1997) Characterization of microbial diversity by determining terminal restriction fragment length polymorphisms of genes encoding 16S rRNA. Appl. Environ. Microbiol. 63:4516-4522.

Marsh, TL (1999) Terminal restriction fragment length polymorphism T-RFLP an emerging method for characterizing diversity among homologous populations of amplification products. Curr. Opin. Microbiol. 2:323-327.

Morris, SA, Radajewski, Willison, TW, Murrell, JC (2002) Identification of the functionally active methanotroph population in a peat soil microcosm by stable-isotope probing. Appl. Environ. Microbiol. 68:1446-1453.

Newby, DT, Reed, DW, Petzke, LM, Igoe, AL, Delwiche, ME, Roberto, FF, McKinley, JP, Whiticar, MJ, Colwell, FS (2004) Diversity of methanotroph communities in a basalt aquifer. FEMS Microbiol. Ecol. 48:333-344.

Osborn, A M, Moore, ERB, Timmis, KN (2000) An evaluation of terminal-restriction fragment length polymorphism (T-RFLP) analysis for the study of microbial community structure and dynamics. Environ. Microbiol. 2:39-50.

Perkins, D, Haws, N, Das, BS, Rao, PSC. Soil hydraulic properties as indicators of land quality for upland soils in forested watersheds with military training impacts. Submitted to Soil Sci. Soc. Am. J.

Reeburgh, WS, Whalen, SC, Alperin, MJ (1993) The role of methylotrophy in the global methane budget, p. 1-14. In JC Murrell and DP Kelly (ed.), Microbial growth on C1 compounds. Intercept. Andover, UK.

USDA (1997) Natural Resources Conservation Service Major Land Resource Area Boundaries. http://www.nrcs.usda.gov/technical/land/mlra/.

Van de Peer, Y, de Wachter, Y (1994) TREECON for Windows: a software package for the construction and drawing of evolutionary trees for the Microsoft Windows environment. Comput. Applic. Biosci. 10: 569-570.

Figure 1. Location of Ft. Benning in Georgia, USA.

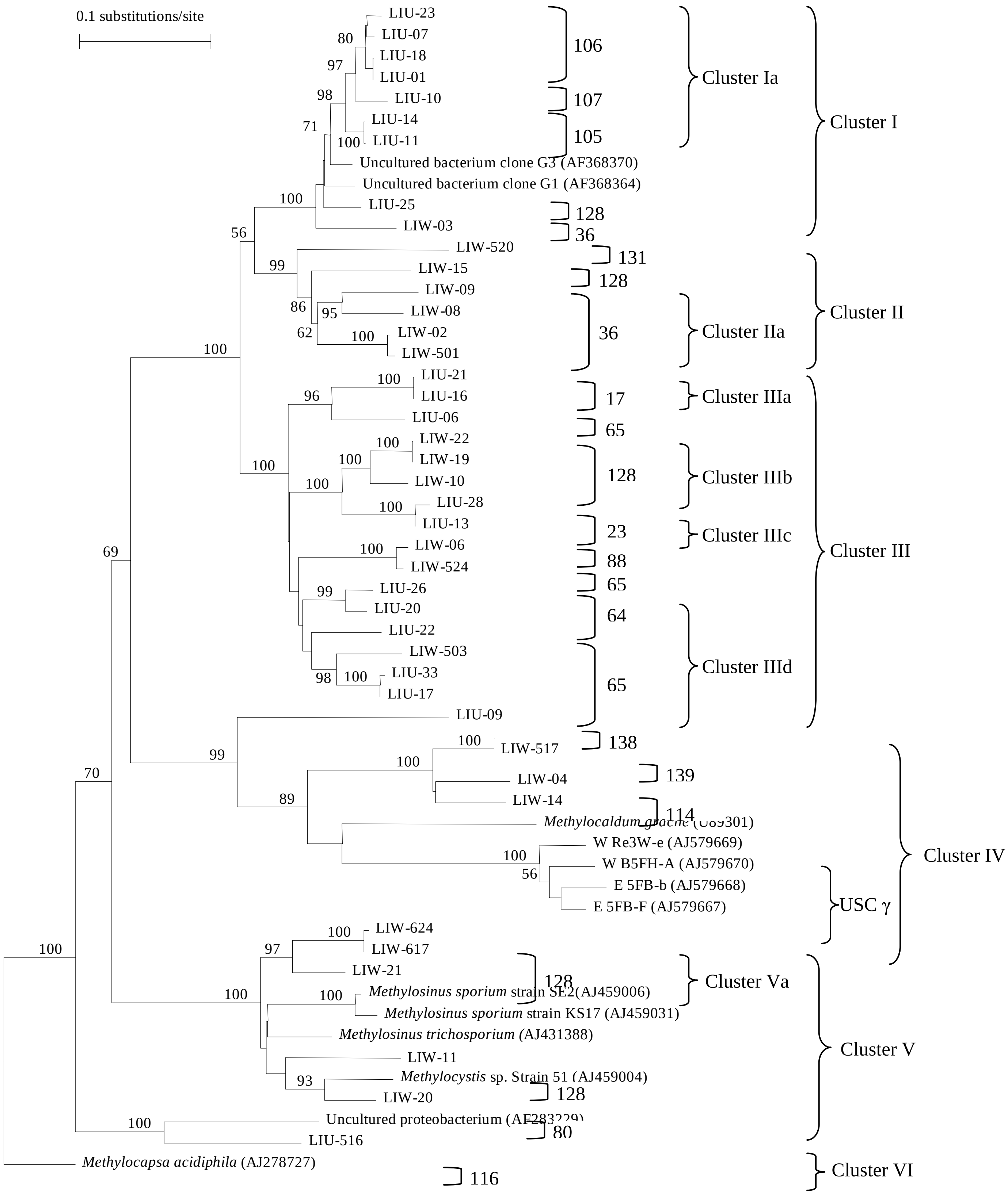
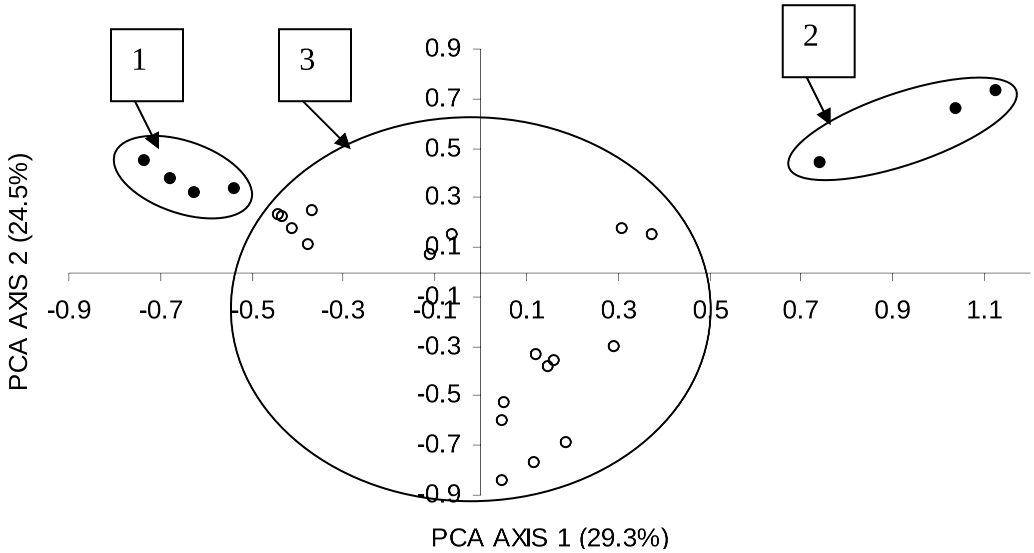


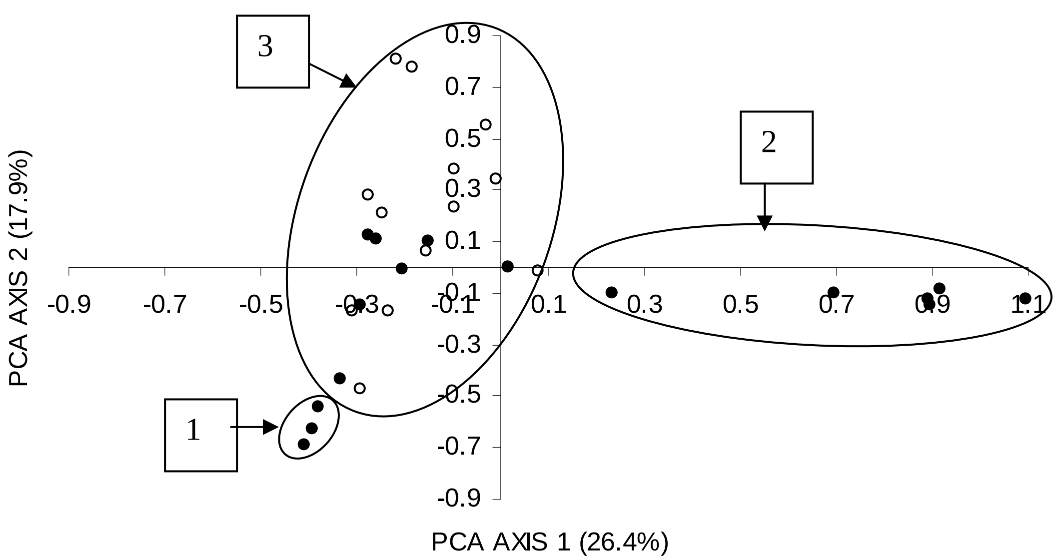
Figure 2. Neighbor-joining phylogenetic tree of Sally Branch *pmoA* sequences. LIU clone numbers refer to upland samples; LIW refers to bottomland samples. Numbers at branch points represent bootstrap values after 100 resamplings. Numbers outside of left most brackets indicate predicted T-RF sizes. Scale bar represents 10 nt change per 100 sequence positions.

A



1
3
2
0.9
0.7
0.5
0.3
0.1
-0.1
-0.3
-0.5
-0.7
-0.9
-0.9
-0.7
-0.5
-0.3
-0.1
0.1
0.3
0.5
0.7
0.9
1.1
PCA AXIS 2 (24.5%)
PCA AXIS 1 (29.3%)

B



3
2
1
0.9
0.7
0.5
0.3
0.1
-0.1
-0.3
-0.5
-0.7
-0.9
-0.9
-0.7
-0.5
-0.3
-0.1
0.1
0.3
0.5
0.7
0.9
1.1
PCA AXIS 2 (17.9%)
PCA AXIS 1 (26.4%)

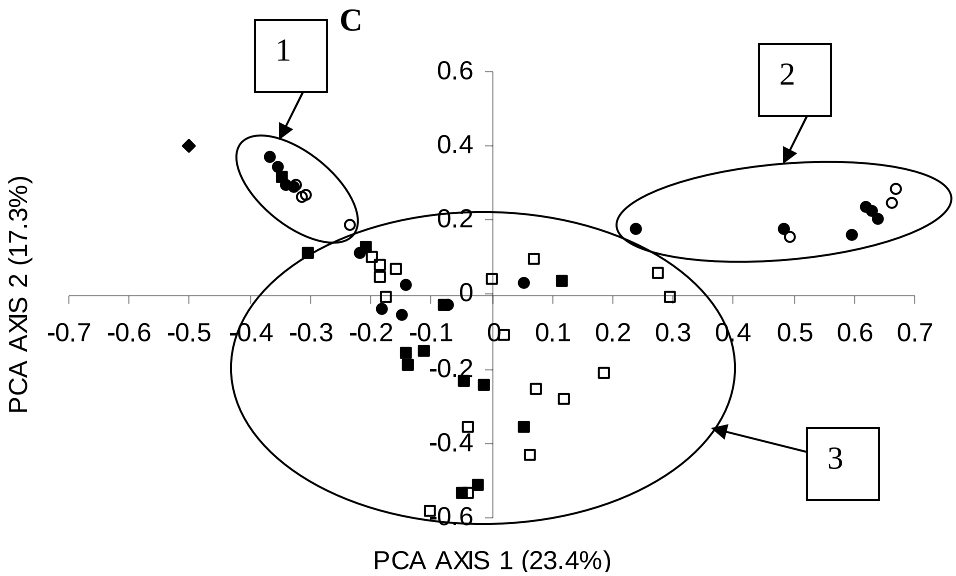


Figure 3. PCA of *pmoA* T-RFLPs. A: Sally Branch watershed; B: Bonham Creek watershed; C: combined Sally Branch and Bonham Creek watersheds. A and B: open circles represent upland samples; closed circles represent bottomland samples. C: Open squares represent Sally Branch upland samples, open circles represent Sally Branch bottomlands; closed squares represent Bonham Creek upland samples, closed circles represent Bonham Creek bottomland samples; closed diamond represents reference sample RH-1 (Table 1).

Table 1. Site characteristics.

Sample[a]	Site Description	Disturbance	pH	Total Carbon (g/kg)	Total Nitrogen (g/kg)
BU0	sandhill	Low	7.3	11.2	0.4
BU01-BU05	sandhill	Low	5.5	10.3	0.3
BU07	crater	Moderate	5.1	17.1	0.5
BU08-BU09	patchy grass	Moderate	5.7	4.3	0.6
BU10	sparse grass	Moderate	5.8	8.5	0.2
BU12	sparse grass	Severe	5.6	2.5	0.1
BU13	tree island	Severe	5.4	2.3	0.1
BU14	sparse grass	Severe	5.3	2.0	0.1
SU01-SU05	sandhill	Low	5.3-6.1	7.0-14.8	0.2-0.6
SU08-SU20	Planted pines	Low	5.1-5.6	7.1-13.0	0.1-0.4
BB10	edge, vegetated, seep	Severe	5.0	27.4	1.0
BB12	hummock, sparse litter	Severe	5.3	4.1	0.2
BB14	hummock, few litter	Severe	5.4	3.9	0.2
BB15	hummock, heavy litter	Severe	4.9	4.9	0.2
BB20	ecotone, litter, seep	Severe	5.1	91.0	4.2
BB21	hummock, grass, moss	Severe	5.6	5.9	0.3
BB22	hummock, heavy litter	Severe	5.0	3.6	0.2
BB23	streamside, sand, little litter	Severe	5.3	5.5	0.2
BB24	hummock, sand/loam	Severe	5.4	6.3	0.2
BB25	edge, litter, seep	Severe	5.1	19.8	0.6
BB30	ecotone	Severe	5.3	9.3	0.3
BB31	elevated, bamboo	Severe	5.4	4.4	0.2
BB32	elevated, shrubs, heavy litter	Severe	5.6	2.4	0.1
BB33	sand bar, sparse vegetation	Severe	5.5	1.6	0
BB34	hummock	Severe	5.8	2.0	0.1
BB35	ecotone, some bamboo	Severe	5.7	4.2	0.1
SB12, SB20, SB22-SB24	depression	Low	4.4-4.8	50.7-290.1	2.5-12.7
SB33	flat	Low	5.5	74.9	3.5
SB34	hummock	Low	4.7	266.0	12.0
RH11[b]	bare ground	Severe	5.3	2.9	ND*

[a] Sample designations refer to watershed and transect; B is Bonham Creek, S is Sally Branch; U is upland transect; B is bottomland transect.
[b] is reference sample; bare ground, highly eroded.

Table 2. Correlations between frequencies of individual T-RFs and sample type.

TREATMENTS	23 (IIIc)[a]	26	30	36 (I, IIa)	48	66 (IIId)	68 (IIId)	80 (V)	84	107 (Ia)	139 (IV)	161
	T-RFs											
Upland	0.028 A	0.018 A	0.086 B	0.0194 A	0.0202 A	0.0251 A	0.0008 B	0.0488 A	0.009358 A	0.07379 A	0.1501 A	0.017 A
Bottomland	0.031 A	0.030 A	0.243 A	0.0078 A	0.0338 A	0.0024 A	0.0225 A	0.0006 A	0.004248 A	0.0000 B	0.0136 B	0.0008 A
LSD (0.05)	0.044	0.033	0.158	0.0951	0.0432	0.0307	0.0184	0.0936	0.0137	0.0577	0.1287	0.0424
	SIGNIFICANCE (*P* Value)											
Landscape	0.883	0.4547	0.046	0.7993	0.5162	0.1319	0.0177	0.3307	0.4454	0.011	0.0411	0.4323
Low	0.0229 A	0.0183 A	0.1025 A	0.0203 A	0.0246 A	0.0226 A	0.0045 A	0.0459 A	0.0094 A	0.060776 A	0.1414 A	0.0169 A
Severe	0.0275 A	0.0227 A	0.1226 A	0.0036 A	0.0165 A	0.0161 A	0.0000 A	0.0285 A	0.0000 A	0.07971 A	0.0164 A	0.0000 A
Moderate	0.0852 A	0.0301 A	0.1588 A	0.0018 A	0.0025 A	0.0172 A	0.0018 A	0.0285 A	0.00606 A	0.07284 A	0.0952 A	0.0019 A
LSD (0.05)	0.0684	0.0518	0.2473	0.1494	0.0678	0.0482	0.0289	0.1599	0.0215	0.0906	0.2022	0.0665
	SIGNIFICANCE (*P* Value)											
Disturbance	0.0770	0.8355	0.8531	0.9278	0.6904	0.9294	0.9355	0.8104	0.6133	0.8838	0.4378	0.7479
Sandhill	0.0160 AB	0.0119 A	0.0417 A	0.0019 B	0.0359 A	0.0488 A	0.0020 A	0.1500 A	0.0025 B	0.0483 A	0.0952 AB	0.0368 A
Sandhill, slight disturbance	0.0346 AB	0.0000 A	0.0101 A	0.2687 A	0.0086 A	0.0281 A	0.0000 A	0.0011 A	0.0298 A	0.0754 A	0.2105 AB	0.0019 A
Depression	0.0315 AB	0.0298 A	0.2427 A	0.0018 B	0.0338 A	0.0024 A	0.0226 A	0.0006 A	0.0043 B	0.0000 A	0.0136 B	0.0008 A
Sparse Grass	0.0275 AB	0.0227 A	0.1226 A	0.0036 B	0.0165 A	0.0161 A	0.0000 A	0.0000 A	0.0000 B	0.0797 A	0.0164 AB	0.0000 A
Patchy Grass	0.0852 A	0.0301 A	0.1588 A	0.0018 B	0.0025 A	0.0172 A	0.0018 A	0.0285 A	0.0061 B	0.0728 A	0.0952 AB	0.0019 A
Sandhill, moderate disturbance	0.0627 AB	0.0506 A	0.2024 A	0.0000 B	0.0105 A	0.0182 A	0.0000 A	0.0649 A	0.0085 B	0.0548 A	0.2236 A	0.0043 A
Planted pines	0.0117 B	0.0122 A	0.0677 A	0.0000 B	0.0216 A	0.0185 A	0.0004 A	0.0178 A	0.0118 AE	0.0890 A	0.1823 AB	0.0192 A
LSD (0.05)	0.0695	0.0523	0.2558	0.1171	0.0721	0.0484	0.0312	0.1501	0.0207	0.0940	0.2082	0.0699
	SIGNIFICANCE (*P* Value)											
Vegetation type	0.1953	0.5333	0.3159	0.0045	0.9259	0.3368	0.5167	0.1395	0.0220	0.2173	0.0148	0.8043
Bonham Creek Upland	0.0454 A	0.0301 A	0.13680 A	0.0392 A	0.0096 B	0.0259 A	0.0004 B	0.0230 A	0.0094 A	0.0803 A	0.1432 A	0.0016 A
Bonham Creek Bottomland	0.0061 B	0.0129 A	0.0999 A	0.0101 A	0.0451 A	0.0094 A	0.0390 A	0.0310 A	0.0121 A	0.0038 B	0.0366 B	0.0066 A
Sally Branch Upland	0.0137 B	0.0128 A	0.0625 A	0.0006 A	0.0262 A	0.0263 A	0.0009 B	0.0629 A	0.0089 A	0.0711 A	0.1487 A	0.0262 A
Sally Branch Bottomland	0.0226 AB	0.0213 A	0.1780 A	0.0134 A	0.0536 A	0.0036 A	0.0364 A	0.0460 A	0.0117 A	0.0000 B	0.0197 B	0.0084 A
LSD (0.05)	0.0312	0.0256	0.1341	0.0598	0.0407	0.0241	0.0269	0.0723	0.0128	0.0383	0.0889	0.0285
	SIGNIFICANCE (*P* Value)											
Transect	0.042	0.3856	0.3806	0.5161	0.1535	0.1554	0.0017	0.5943	0.9279	<.0001	0.0039	0.2098

[a] Numbers within parentheses refer to cluster presented in Fig. 2.
Within each column, means that share the same letter are not significantly different at P = 0.05

3.

3.2 Vegetation

Indicators related to vegetation community composition in moderately or less impacted sites are often confounded by residual effects of prior soil disturbance related to agricultural land uses. Plant species potentially sensitive to low to moderate levels of disturbance probably have been extirpated from the sites due to historic levels of chronic disturbances. Indicator species to assess ecological condition may require an evaluation of “natural” or reference conditions prior to their use.

Vegetative indicators that most accurately reflected the impacts of military training were:

- **Percent cover of herbaceous vegetation (ground cover, and litter cover), or in cases of more severe impacts, canopy cover.**
- **Plant species present only in severely disturbed sites identify the highest degree of disturbance.**
- **Plant species indicating various stages of recovery from severe disturbance were identified that may be useful in tracking the progress of restoration efforts in highly-impacted areas.**

3.2.1
Vegetative indicators of disturbance in a chronically-disturbed ecosystem: Ft. Benning Military Reservation, Georgia. Tanner, G.W., D.L. Miller, and J. Archer.

Abstract:

Military reservations receive many types of anthropogenic disturbances associated with training missions and general land management practices. Some of these disturbances may elicit degradation of the ecological conditions on the site. Thus, identification of indicators of ecological change is needed to guide the development of alternative land use practices. Foliar cover of understory woody and herbaceous plants and overstory canopy cover were estimated within 6 watersheds and across a disturbance gradient from low to severe on the Ft. Benning Military Reservation, GA. Woody plants did not differentiate well among the disturbance levels; however, there was a trend of decreased overstory canopy cover with increased disturbance. Herbaceous vegetation composition on severely-disturbed sites segregated from low and medium disturbances but no segregation was found between the two lower levels of disturbance. Chronic, landscape-scale disturbances have resulted in a very resilient flora. Coverage of bare ground and plant litter may best serve as indicators of disturbance.

INTRODUCTION:

Some contemporary ecologists view disturbance as a natural process and a source of heterogeneity within ecosystems (Wu and Loucks 1995). Disturbances, however, come in many shapes and sizes in the spatial and temporal context and result in varying degrees of ecosystem response, also within a spatial and temporal context (Turner et al. 2001). An ecosystem's level of resilience to disturbances will be associated with a complex scalar integration of disturbance impacts with the antecedent state of the ecosystem (Holling and Gunderson 2002). Humans, whether considered part of nature or not, have the capability to disturb ecosystems at virtually any perceivable scale (Westley et al 2002). A mixture of social and ecological consciousness and extant environmental laws, however, dictate the need for recognition and management of our impacts.

In 1999, Ft. Benning Military Reservation, near Columbus, GA, was selected by the US Strategic Environmental Research and Development Program (SERDP) to be the study location to investigate ecological impacts of anthropogenic disturbances (SERDP's Ecosystem Management Project- SEMP). The initial objective of SEMP was to identify indicators of ecological change across a gradient of disturbance. Information reported herein focuses on vegetative relationships with this disturbance gradient. A goal is to provide land managers guidelines on recognizing early stages of ecosystem degradation.

STUDY AREA

Ft. Benning Military Reservation is located in Muscogee, Marion and Chattahoochee Counties, Georgia and Russell County, Alabama and covers approximately 73,000 ha. Our study was situated entirely within Georgia in the Upper Coastal Plain Ecoregion (Bailey 1995). Average elevation was about 240 m, and the terrain was rolling. Average annual precipitation is 124 cm with average daily temperature ranging from –1° to 37°C (Dale et al. 2002). Major soil series include Troup-Cowarts-Nankin, Cowarts-Nankin, Troup-Lakeland, Troup-Vaucluse-Pelion, Ailey-Troup-Vaucluse and Dothan-Orangeberg-Esto associations (USDA/NRCS 1997).

Much of the Reservation was cultivated for row crops and grazed in the 1800's and early 1900's prior to the military's acquisition in 1921 (Kane and Keeton 1998). Upland sites probably were once dominated by longleaf pine (*Pinus palustris*) forests (Cooper 1989) but are now a mixture of longleaf, loblolly pine (*P. taeda*), shortleaf pine (*P. echinata*) and an assortment of hardwoods. Upland understory vegetation is a mixture of bluestem grasses (*Andropogon* spp., *Schizachrium* spp.), forbs and woody shrubs and vines. Bottoms are characterized by single or braided streams and are dominated by hardwoods species. Bottom understory vegetation is dominated by woody shrubs and vines with occasional stands of switchcane (*Arundinaria gigantea*). Current silvicultural practices within the pine forests include thinning, clear cut harvest and regeneration with longleaf and loblolly pine and maintaining an active 3-yr cycle of prescribed burning in both the dormant and growing seasons.

MATERIALS AND METHODS

Six, order-2 and 3 watersheds (Halloca, Randall, Sally Branch, Bonham, Shell, and Wolf Creek) located across the Ft. Benning Reservation complex were selected to measure soil and vegetation attributes. Sampling points (n=273) were randomly assigned along transects distributed from the top to the bottom of each watershed and were located across an elevational gradient to represent sites in upland, mid-slope and stream bottom positions. Each sampling site was given a disturbance rating of low, moderate or severe based on the presence of military tracked vehicle use, recent logging activity (e.g., skid trails) and/or evidence of ground-based troop training (e.g., bivouac areas). Soil texture at each sampling point was evaluated and assigned in the field by experts from the University of Florida Department of Soil and Water Sciences, and a subset of points (n=56) were selected to core the top 15 cm of soil for laboratory analyses. GPS coordinates (Tremble® at 1-m accuracy) were recorded for each sampling site.

Foliar canopy cover of each understory woody plant species (below 2 m) was estimated along 3, 5-m transects that were oriented in random directions from all sampling points during late summer and fall 2000. Canopy intercept (Bonham 1989) was measured to the nearest centimeter along each transect and summed. Overstory canopy cover was estimated at each sampling point with a densiometer (Lemon 1949). Canopy cover of each herbaceous species, litter and bare ground was estimated (Daubenmire 1959) within 3, 1-m^2 quadrats randomly located at a 2-m distance from the 56 sample sites chosen for laboratory soil analyses.

Canonical Correspondence Analysis (CCA) (CANOCO 4.0) was used to relate vegetation data to environmental variables (ter Braak 1998). Biplots were used to provide graphical patterns among the variables.

RESULTS AND DISCUSSION

A total of 113 woody species were encountered in the field; a small portion of these could not be identified to species, so they were given a number until definite identification can be obtained (Table 1). Correspondence Analysis of relative woody plant cover with environmental variables indicated a separation of low (Lo) disturbance sites from moderate (Mod) and severe (Sev) sites, but no marked separation between moderate and severe disturbance sites (Figure 1). Severe disturbance was most closely associated with upland, sandy clay soils. Increased overstory canopy cover as estimated by densiometer measurements were associated with low disturbance sites. These associations have some statistical strength given the signifacance of the first eigenvalue (Table 2), however, the lack of a major decrease of the sequential eigenvalues from Axis 1 through Axis 4 indicates a lack of close association among the variables.

Severe disturbance sites were areas of active heavy military equipment training (tanks and Bradley personnel carriers). Within this classification there was a gradient of disturbance from a condition of virtual absence of woody plants to a condition of scattered larger trees (*Pinus palustris, Quercus arkansana, Pinus taeda*) and remnant shrubs and vines that could withstand, or be spread by, repeated vehicular trampling (*Opuntia* sp., *Ipomea* sp., *Vaccinium* sp., *Viburnum rufidulum*, *Crataegus* sp.). Relative cover of *Rubus* sp. and *Rhus copallina* may be an important indicator of a shift from moderate to severe conditions. These two species are prolific seed producers, enhancing their ability to colonize disturbed sites, and they appear to withstand physical disturbance once established.

We did not sample many bottom sites that had had severe disturbance. The few sites of such classification, however, were downstream and close to road crossing. Erosional fans were evident along with some mortality of overstory trees. Consequently, most bottom sites were classified as low disturbance. Organic soils were rare and were at sites of impounded water. i.e., beaver ponds. It is possible that beaver ponds should be classified as Sev disturbance sites from an ecological viewpoint.

A total of 110 herbaceous species were encountered while sampling (Table 3). Some of these species were not identifiable to species due to immaturity, thus they were given a number. Given that a very limited number of herbaceous species were encountered within the bottom sites, CCA analysis between species cover and environmental variables was conducted for just the slope and upland sites where the vast majority of species occurred. Correspondence Analysis indicated a separation of severe disturbance from moderate and low levels of disturbance along a gradient from high litter cover (low and moderate) to an absence of litter cover (severe) (Figure 2). The separation between low and moderate disturbance categories, however, was not distinctive, hence a possible explanation for a lack of statistical significance for the first eigenvalue of the analysis (Table 2). Inspection of the minimal degree of difference among the third and fouth eigenvalues also indicates a general a lack of structure among the relationships. Therefore, this analysis should be viewed as an indication of a possible trend among the variables. This result possibly is related to the relatively low sample size of the analysis (n= 36).

Litter cover varies with short-term forest management regimens, e.g., burning schedules. Litter cover will be related to basal area of overstory trees and basal area and density of understory plants, both woody and herbaceous. Whole scale removal of plants from an area due to tank and other tracked vehicle training disturbances, however, will subsequently cause long term reduction of litter cover.

Given the limitations of the weak statistical strength of the analysis, there appears to be a relationship between the cover of a subset of the herbaceous species and sites of severe disturbance. Those herbaceous species most closely associated with severely disturbed sites were: *Digitaria ciliaris, Diodia teres, Stylosanthes biflora*, Grass 4, *Aristida purpurescens, Opuntia humifusa*, *Haplopappus dirasicatus*, and *Paspalum notatum*. Solid stands of *Paspalum notatum*, an exotic species of grass, occurred on sites that had been totally denuded in the past, and probably was planted to reduce erosion.

The Ft. Benning Reservation landscape, especially the uplands, has a long history of anthropogenic disturbances ranging from crop cultivation more than a century ago to current military training exercises. The secondary forests that regenerated on this landscape must have originated from a seed stock of very resilient species (Holling 1973, Turner et al. 2001). If so, then it is not surprising to find similar species compositions among the contemporary floras on low and moderately disturbed sites. Repeated disturbance from tracked vehicles, however, will eventually lead to the extirpation of all above ground vegetation. The inherent resilient nature of the existing flora’s response to low to moderate levels of disturbance may impede land managers’ ability of recognize ecological change. Monitoring coverage of bare ground and plant litter may provide the best integrative measure of cumulative plant responses.

REFERENCES

Bailey, R.G., 1995. Description of the ecoregions of the United States. USDA Forestry Service, Ogden, Utah.

Bohnam, C.D. 1989. Measurements for terrestrial vegetation. J. Wiley & Sons. New York.

Cooper, W., 1989. Ecology of the pine-hardwood type. In: Waldrop T.A. (Ed), Proceedings of Pine-Hardwood Mixtures: A Symposium on Management and Ecology of the Type, Atlanta, GA. U.S.D.A. Forest Service Southeastern Forest Experiment Station, Asheville, NC. General Technical Report SE-58, p. 271.

Dale, V.H., Beyeler, S.C., Jackson, B., 2002. Understory vegetation indicators of anthropogenic disturbance in longleaf pine forests at Fort Benning, Georgia, USA. Ecological Indicators **1:**155-170.

Daubenmire, R.F. 1959. Canopy coverage method of vegetation analysis. Northwestern Scientist **33**:43-64.

Holling, C. S. 1973. Resilience and stability of ecological systems. Annual Review of Ecology and Systematics **4**:1-23.

Holling, C.S., and L.H. Gunderson. 2002. Resilience and adaptive cycles. Pages 25-62. in C.S. Holling and L. Gunderson, editors. Panarchy: understanding transformations in human and natural systems. Island Press, Washington, D.C.

Kane, S., Keeton, R., 1998. Fort Benning: the land and the people. Southeast Archaelogical Center, National Park Service, Tallahassee, FL.

Lemon, P.C. 1949. Successional responses of herbs in the longleaf-slash pine forest after fire. Ecology **30:** 135-145.

Ter Braak, C.J.F, Smilauer, P., 1998. CANOCO reference manual and user's guide to Canoco for Windows software for canonical community ordination (Version 4). Microcomputer Power, Ithaca, NY.

Turner, M.G., R.H. Gardner, and R.V. O'Neill. 2001. Landscape ecology in theory and practice: pattern and process. Springer-Verlag. New York..

USDA NRCS, 1997. Soil Survey of Chattahoochee and Marion Counties, Georgia. U.S. Government Printing Office, Washington, D.C.

Westley, F., S.R. Carpenter, W. A. Brock, C. S. Holling, and L. Gunderson. 2002. Why systems of people and nature are not just social and ecological systems. Pages 103-120 in C.S. Holling and L. Gunderson, editors. Panarchy: understanding transformations in human and natural systems. Island Press, Washington, D.C.

Wu, J., and O.L. Loucks. 1995. From balance of nature to hierarchical patch dynamics: a paradigm shift in ecology. Quarterly Review of Biology **70**:439-466.
.

Table 1. Codes for woody species encountered at Ft. Benning Military Reservation, GA.

Codes	Scientific Name	Codes	Scientific Name
AMBE, FAGR	Fagus grandifolia	QUBJ	Quercus marilandica
AMBEA	Callicarpa americana	QUBL	Quercus velutina
ANIS	Illicium floridanum (or parviflorum)	QUHE	Quercus hemisphaerica
ARAR	Aralia spirosa	QULA	Quercus laurifolia
ARARB	Aronia arbutifolia	QUMA	Quercus margaretta
ASHE	Fraxinus pennsylvanica	QUPO	Quercus stellata
AZAL	Rhododendron sp.	QUSE	Quercus sp
BLBE	Carpinus caroliniana	QUSR	Quercus falcata
BLCH	Prunus serotina	QUTU	Quercus laevis
BLGU, NYSY	Nyssa sylvatica	QUWA	Quercus nigra
CAPH	Cephalanthus	QUWH	Quercus alba
CATA	Catalpa bignonioides	REBU	Cercis canadensis
CHIN	Quercus muehlenbergeii	REMA	Acer rubrum
CLAL, CLEAL	Clethra alnifolia	RETI	Cyrilla racemiflora
COBE	Coralbeads	RHCO	Rhus copallina
CRET	Crataegus sp.	RHOD	Symplocos tinctoria
CUGL	Cudwigia glandulosa	RIBI	Betula nigra
CYRA, TYTY	Cyrilla racemiflora	ROCA	Rosa carolina
DEBA	Decumaria barbara	RUBU	Rubus sp.
DESM	Desmodium sp.	SABA	Sabal sp.
DOGW	Cornus florida	SASS	Sassafras albidum
GADU	Gaylussacia dumosa	SBMA	Magnolia virginiana
GAFR	Gaylussacia frondosa	SEFR	Sebastiania fruticosa
HICK	Carya sp.	SH10	Shrub 10
HOBE	Ostrya virginiana	SH11	Shrub 11
HOSU	Lonicera sempervirens or japonica	SH15	Shrub 15
HYHY	Hypericum hypericoides	SH2	Shrub 2
HYQU	Hydrangea quercifolia	SHSS	Shrub seedling
ILCO	Ilex coriacea	SMIL	Smilax sp.
ILDE	Ilex decidua	STAM, STAME	Styrax americanum
ILGL	Ilex glabra	STGR	Styrax grandiflorium
ILOP	Ilex opaca	SUBE	Celtis
IPOM, MOGL	Ipomea sp.	SWGU	Liquidambar styraciflua
ITEA	Itea virginica	TR3	Tree 3
KUDZ	Pueraria lobata	TR6	Tree 6
LISI	Ligustrum sinense	TRCR	Campsis radicans
LOJA	Lonicera japonica	TRSE	Tree seedling
LYON	Lyonia sp.	TUPO	Liriodendron tulipifera
MAGR	Magnolia grandiflora	UN2	Unknown 2
MYCE	Myrica cerifera	UN8	Unknown 8
MYHE	Myrica heterophylla	UNIL	Ilex sp.
MYRI	Myrica sp.	UNSE	Unknown seedling
OABJ	Quercus incana	UNTR	Unknown tree
OPUN	Opuntia sp.	UT1	Unknown tree 1
OXAR	Oxydendrum arboreum	VAAR	Vaccinium arborum
PAPA	Asimina parviflora	VAC2	Vaccinium sp.
PEBO	Persea borbonia	VAEL	Vaccinium elliotii
PERS	Diospyros virginiana	VAMY	Vaccinium myrsinites
PEVI	Ampelopsis arborea	VAST, VASTA	Vaccinium stamineum
PILL	Pinus palustris	VIBRU, VIBU, VIRU	Viburnum rufidulum
PILO, PITA	Pinus taeda	VICR	Parthenocissus quinquefolia
PISH	Pinus echinata	VIN3	Vine 3
POIV	Toxicodendron radicans	VIROT	Vitus rotundifolia
POOA	Toxicodendron pubescens	WIEL	Ulmus alata
PRAN	Prunus angustifolia	WIHA	Hamamelis virginiana
PRUM	Prunus umbellata	YEJE	Gelsemium sempervirens
QUAR	Quercus arkansana	YUCC	Yucca filamentosa

Table 2. Canonical Correlation Analysis eigenvalues for woody plant species relative cover correlated with environmental variables at all sites and herbaceous species absolute cover correlated with environmental variables at upland and slope sites.

	Axis 1[a]	Axis 2	Axis 3	Axis 4
Woody spp	0.52 (*)	0.21	0.18	0.16
Herbaceous spp.	0.56 (ns)	0.48	0.34	0.28

[a] Test of significance of first eigenvalue: *= <0.05, ns= not significant.

Table 3. Codes for herbaceous species encountered at Ft. Benning Military Reservation, GA.

Code	Scientific Name
AGTE	Agalinas tencifolia
ANsC	Andropogon sp. (cover)
ANsD	Andropogon sp. (density)
ANTE	Andropogon ternarius
ANVIc	Andropogon virginicus (cover)
ANVId	Andropogon virginicus (density)
ANRU	Anthaenantia rufa
ARPU	Aristida purpurescens
ARTU	Aristida tuberculosa
ARGA	Arundinaria galgantium
AS1	Aster 1
AS2	Aster 2
AS3	Aster 3
AS4	Aster 4
AS5	Aster 5
AS6	Aster 6
ASDU	Aster dumosus
ASLA	Aster laterifloris
ASTO	Aster tortifolias
BRER	Brachyelytrum erectum
BUCI	Bulbostylis ciliatifolia
CAS	Cassia sp.
CES	Cenchrus sp.
CHLA	Chasmanthium laxum
CIAR	Cinna arundinacea
CRST	Cnidoscolus stimulosus
COER	Commelina erecta
COMA	Coreopsis major
COMS	Coreopsis major var. stellata
COS	Coriopsis sp.
CRGL	Croton glandulosus
DEPA	Desmodium paniculatum
DES	Desmodium sp.
DICI	Digitaria ciliaris
DITE	Diodia teres
ELCA	Elephantopus carolineanus
ELTO	Elephantopus tomentopus
ERS	Erianthus sp.
EGS	Erigonium sp.
ERHI	Erigrostis hirsuta
EUAL	Eupatorium altissimum
EUCA	Eupatorium capillifolium
EUJU	Eupatorium jucundum (Ageratina jucunda)
EUPU	Euphorbia pubentissima
FE1	Fern 1
FO1	Forb 1
FO10	Forb 10
FO2	Forb 2
FO3	Forb 3
FO4	Forb 4
FO5	Forb 5
FO6	Forb 6
FO7	Forb 7
FO8	Forb 8
FO9	Forb 9
GAVO	Galactia volubilis
GACI	Galium circaezans
GAUR, GAFI	Gaura filiper
GR1	Grass 1
GR2	Grass 2
GR3	Grass 3
GR4	Grass 4
HADI	Haplopappus dirasicatus
HEAR	Hexastylis arifolia (Asarum arifolium)
HYGE	Hypericum gentioides
LEVI	Leersia virginica
LECU	Lespedeza cuneata
LEHI	Lespedeza hirta
LIEL	Liatris elegans
LITE	Liatris tencifolia
LIS	Liatris sp.
LISQ	Liatris squarrulosa
ONSE	Onoclea sensibilis
OPHU	Opuntia humifusa
OSCI	Osmunda cinnomomea
OSRE	Osmunda regalis
PACHc	Panicum chamaelanche cover
PACHd	Panicum chamaelanche density
PACLc	Panicum clandestinum cover
PACLd	Panicum clandestinum density
PANO	Paspalum notatum
PASE	Paspalum setaceum
PITY	Pityopsis
POPR	Polypremum procumbens
POS	Potentilla sp.
PTAQ	Pterydium aquilinum var. pseudocaudatum
RAGW	Ragweed
RHMI	Rhynchosia minima
RHMC	Rhynchospora microcephala
SCSCc	Schizacherium scoperium cover
SCSCd	Schizacherium scoperium density
SCMI	Schrankia microphylla
SCL	Scleria bottom
SE1	Sedge 1
SE2	Sedge 2
SE3	Sedge 3
SEPE	Segmaria pectinata
SOOD	Solidago odora
SOS	Solidago sp.
SONU	Sorgastrum nutans
SPMO	Sphagnum moss
SPJUc	Sporobolus junceus cover
SPJUd	Sporobolus junceus density
STBI	Stylosanthes biflora
TEFL	Tephrosia florida
TEVI	Tephrosia virginiana
TRS	Tradescantia sp.
TRFL	Tridens flavus
TRAM	Triplasis americana
UN	Unknown
VEAN	Vernonia angustifolia
WOOB	Woodsia obtusa
WOAR	Woodwardia areolata
XYDI	Xyrus difformis

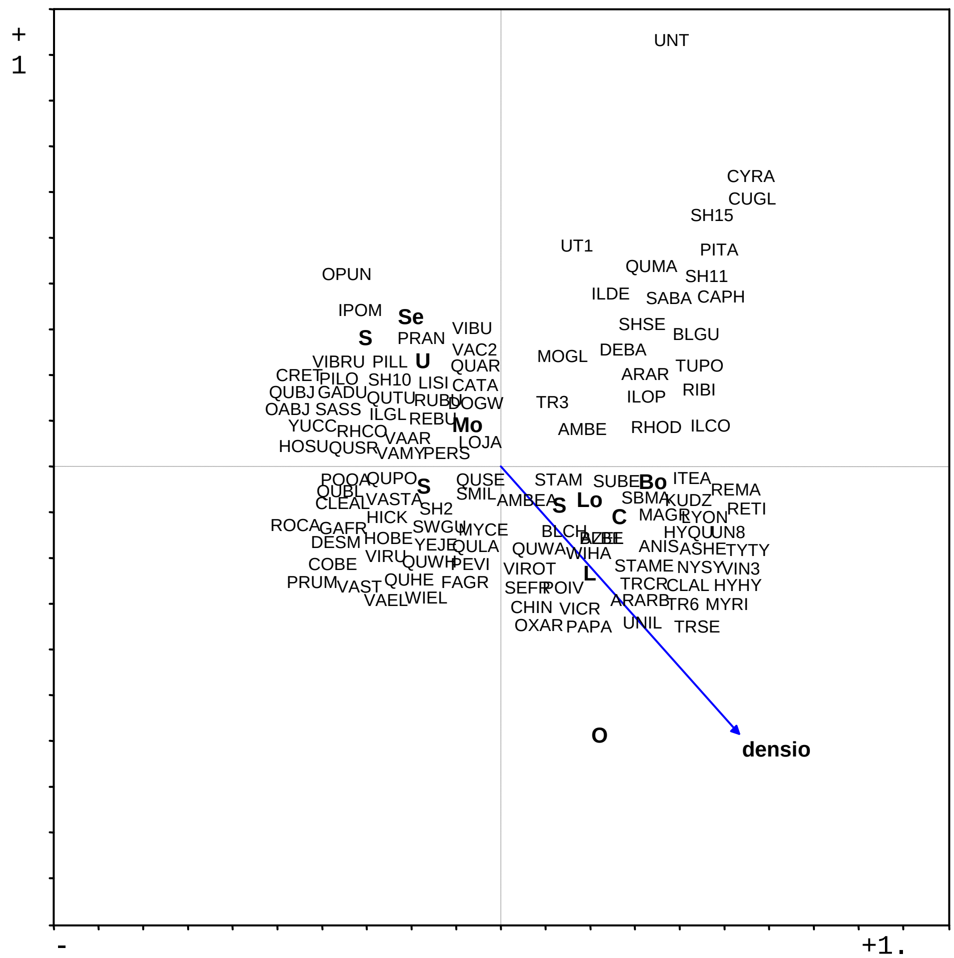


Figure 1. CCA biplot for woody species relative cover and environmental variables.

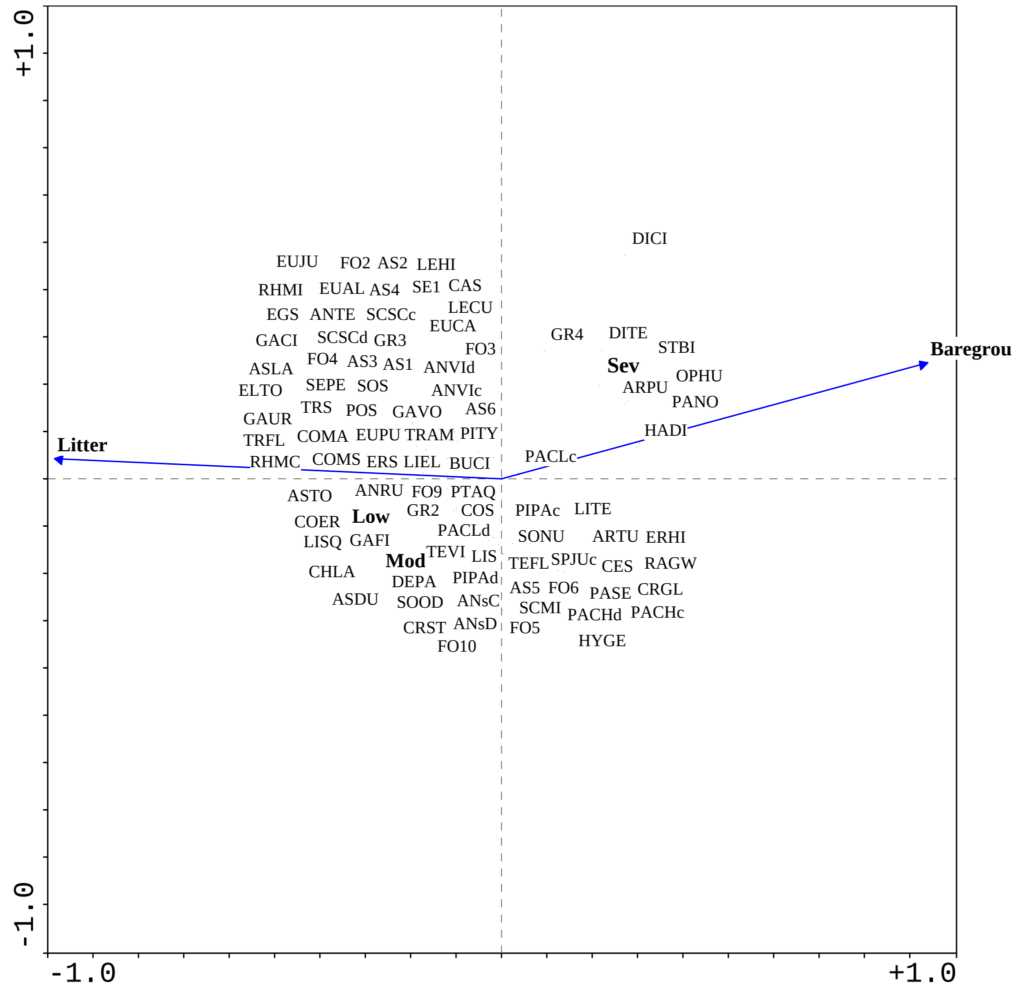


Figure 2. CCA biplot for herbaceous species absolute cover and environmental variables.

3.2.2
Understory Vegetation and Soil response to silvicultural activity in a southeastern mixed pine forest: A chronosequence study. Archer, J and D.L. Miller.

A silvicultural chronosequence was studied in upland pine stands of Fort Benning, Georgia, to assess understory vegetation and soil characteristics following a silvicultural disturbance. Hypotheses regarding patterns of understory vegetation distribution and abundance, and the impact of disturbance on soil properties were evaluated in 32 forest stands. The chronosequence encompassed various times since clear-cut regeneration: stand age (0-3 yrs), (8-10 yrs) (18-20 yrs) and (30-80 yrs). Soil pH, total carbon (C) and nitrogen (N) contents, soil texture, and bulk density were used to characterize soil conditions across the chronosequence. Vegetative conditions across the chronosequence were characterized by quantification of species foliar cover. Canonical correlation analysis (CCA) determined the relationship between understory vegetation pattern and measured soil gradients and stand age. According the CCA, stand age was the most important factor influencing distribution and abundance of understory vegetation. Herbaceous species composition and cover varied more with stand age than understory woody species. Aside from a trend of decreasing bulk density, soil parameters did not vary with recovery time. Indicator analysis identified Gaylussacia mosieri and Carya spp. as the only significant woody indicators of age class. Cyperus croceus and Bulbostylis barbata were identified as herbaceous indicators of the 0-3 age class. Andropogon virginicus, Dichanthelium sp. and Sporobolus junceus were significant indicators of the 8-10 yr age class. Significant indicators of the 18-20 yr class were Pityopsis sp. and Tridens flavus. Andropogon ternarius, Schizachyrium scoparium, Desmodium sp., Hieracium sp., Rhynchosia tomentosa were indicators of 30-80 yr age class. Major changes in understory vegetation cover and composition continued for at least 18-20 years post clear-cut regeneration.
Keywords: Longleaf pine; Silviculture; Groundcover; Disturbance

INTRODUCTION

Longleaf pine forests were and are still considered one of the most species rich in the U.S. and perhaps worldwide (Walker, 1993). At small scales (1-100 m^2), understory herbaceous layers may contain 40-75 species of vascular plants in a single 1-m^2 quadrat and 130 for a 0.1-ha plot (Clewell, 1989). Plant and animal species associated with longleaf pine communities typically exhibit a high incidence of endemism, e.g. the well-known longleaf pine-wire-grass communities includes 191 species of rare plants (Noss et al., 1995).

An estimated 14% of the expansive longleaf pine forests remain, with approximately 3% surviving as old-growth. This loss is comparable with or exceeds that of many of the other unique ecosystems in North America (Noss, 1989). The loss of aerial extent of this ecosystem is due primarily to logging, fire suppression and conversion to plantation forestry (most often of other pine species, e.g. slash pine (P. elliottii) or loblolly pine (P. taeda). Much of these second-growth pine forests have experienced soil erosion and loss of fertility during logging and agricultural use in the last 150 yrs. Of the remaining longleaf pine stands, many are small (<10 ha) and have experienced varied management practices including disturbed fire regimes, grazing, and logging (Provencher et al., 1997).

Historically, fire was an important natural disturbance in maintaining longleaf pine ecosystems. Lightning fires, occurring at intervals of 1-3-yrs, were carried over large areas by fairly continuous cover of perennial grasses, most notable wiregrass (Aristida beryrinchana, A. stricta) and bluestem grasses (Andropogon spp. and Schizachyrium spp,) and pine duff. It is believed that 10%-30% of the southeastern pinelands burned annually (Ferry et al., 1995). These frequent fires reduced litter accumulation and invasion of competing woody species. Pine seedlings and many of the grasses and forbs present in longleaf pine communities are shade-intolerant, and many require bare mineral soil and reduced competition for germination and early growth. With much of the forests being converted to agriculture or used in the Naval stores industry (for turpentine or pitch, used to caulk the seams of wooden ships) and much of the herbaceous understory disturbed and fragmented by logging roads and fields, ground fires can no longer carry for long expanses. In the absence of fire, oak (Quercus spp), hickory (Carya spp.), and other pine species replace longleaf pine on the Coastal Plain (Stout and Marion, 1993) leading to a decline in species associated with pine-dominated ecosystems. Pines and understory grasses exhibit extreme longevity and have a high degree of nutrient and water retention; these attributes maintain their site dominance and increase their resistance following a disturbance such as fire (Landers et al., 1995).

Logging with high levels of soil disturbance impacts stability of certain soil minerals, alters soil structure, increases bulk density and leads to altered vegetation species composition (Congdon and Herbohn, 1993). Heavy machinery in logging operations can have a high degree of impact on soil bulk density and infiltration. However, depending on the amount of compaction and soil type, recovery is possible, especially for surface soils (Gayoso and Iroume, 1991). Reported recovery times vary widely. Following logging, bulk density in a loblolly pine forest was thought to take 18 yrs to recover (Hatchell et al., 1970) while compaction remained significantly elevated 32 yrs after logging in Douglas fir stands (Wert and Thomas, 1981).

Soil disturbance can also result in soil scarification, a process where exposed mineral soil becomes encrusted and compacted by rainfall, thus impeding seedling root penetration (Pierce et al., 1993). High levels of compaction can significantly inhibit root growth and thus shoot growth. Mycorrhizal mycelia growth, seed germination and seedling emergence also maybe restricted by increased compaction (Greacen and Sands, 1980).

Understory plants have been recognized as indicator species of potential timber productivity since the 1920s (Clements 1928, Hunter 1990), yet the majority of the studies on anthropogenic impacts on forests focus on wildlife (Ercelawn 1999; Fimbel et al., 2001) and tree species' (Huth and Ditzer, 2001; Stearns and Likens, 2002) ability to recover. Understory herbaceous vegetation is often left out of the equation even though it is likely to be the most sensitive of the three groups (Duffy and Meier, 1992). However, several recent studies examined the impact of silviculture on herbaceous understory growth, composition and diversity (Halpern and Spies, 1995; Roberts and Gilliam, 1995; Elliott et al., 1997; Gilliam, 2002). Other work reported on the interacting effects of complete deforestation and herbicide on groundcover vegetation recovery (Kochenderfer and Wendel, 1983; Reiners, 1992). Still others looked at impacts of logging on vegetation patterns through comparisons of second-growth and old-growth forests (Qian et al., 1997; Goebel et al., 1996). While these studies focus primarily on herbaceous vegetation diversity, our study focused on composition and abundance of understory vegetation (herbacecous and woody) and soil changes following silvicultural practices.

There were four specific questions in this study: 1) What are the rate and pattern of changes in distribution and abundance of herbaceous ground cover and understory woody species following clear cut regeneration (hereafter, clearcutting)? Certain species known to be particularly sensitive to disturbance will likely increase over recovery time. 2) Do certain soil characteristics change following clear cutting? We hypothesize that bulk density will decrease over time. 3) Can we identify species that are indicators of assemblages of plants characterizing increasing length of time following clear cutting? 4) Are plant assemblages differentially affected by logging depending on soil texture? Since it is known that plant growth can be inhibited with increased bulk density and that the amount of clay content in the soil affects the degree to which the soil can be compacted, we believe that the texture of the soil will affect plant assemblages.

Identification of pattern and rate of understory recovery following clear cutting will aid in identification of sensitivity and rate of return of herbaceous species following low to moderate levels of disturbance and further separate natural variation from variation attributed to anthropogenic disturbance. Recovery time may be reduced through improved management. Silviculture can be managed spatially and temporally so that forest ecosystems do not further depart from natural states, but instead move towards recovery. Indicator species may be the best means for revealing landscapes that are subjected to too much disturbance.

METHODS

Study Area:

The 72,9000-ha, Fort Benning Military Reservation, is located in Muscogee, Marion and Chattahoochee Counties, Georgia and Russell County, Alabama (32^{o}21'N, 84^{o}58'W). The study was conducted on the portion of the reservation found within the Upper Coastal Plain Ecoregion (Bailey, 1995) with nearly level to gently rolling topography and a maximum elevation of 240 m. The semi-tropical climate averages 124cm of rainfall annually with average daily temperature ranging from -1^{o}C to 37^{o}C, (Dale et al., 2002).

The study was restricted to upland, sandhill areas on droughty, infertile entisols and ultisols with loamy sand to sandy loam surface horizons. Major soil associations included Cowarts-Nankin, Troup-Cowarts-Nankin, Troup-Lakeland, Ailey-Troup-Vaucluse, Dothan-Orangeberg-Esto, and Troup-Vaucluse-Pelion (USDA NRCS, 1983; USDA NRCS, 1997). These sandhills are believed to have been longleaf pine forests (Cooper, 1989) but are now dominated by mixed stands of Pinus palustris (longleaf pine), P. taeda (loblolly pine), P. echinata (shortleaf pine), and P. elliottii (slash pine), with understories consisting of Quercus laevis (turkey oak), Q. marilandica (blackjack oak), Q. incana (bluejack oak), Nyssa sylvatica (blackgum), Diospyros virginiana (persimmon), Liquidambar styraciflua (sweetgum), Morella cerifera (wax myrtle), and Cornus florida (dogwood) (King et al., 1998). The groundcover consists mainly of Andropogon virginicus (broomsedge), Andropogon ternarius (splitbeard), Schizachyrium scoparium (little bluestem), Rubus spp. (blackberry), Gaylussacia dumosa (dwarf huckleberry), Vaccinium spp. (blueberry), and Pteridium aquilinum (bracken fern) (Myers, 1990; King et al., 1998).

Most of the reservation area was farmed until 1921 when Ft. Benning was established under military control (Kane and Keeton, 1998). Management practices specific to study sites include clear cutting, selective thinning (on only the oldest sites), use of heavy machinery, the creation of haul roads, the use of log decks and skid trails, drum chopping, planting of pine seedlings (loblolly or longleaf), and prescribed burning in both growing and dormant seasons on a 3-year cycle but no herbicide use. Military activity in the study area was low to moderate. Areas of the reservation with tank and vehicle maneuvers, large-scale troop movements and ordinance activity were not included.

Experimental Design:

A chronosequence of sites was selected based on age since clearcut. Four pine stands with loamy (according to soil survey maps) and 4 with sandy surface soils were randomly selected in each of the following age classes: 0-3 yrs, 8-10 yrs, 18-20 yrs, and 30-80 yrs. Criteria for potential replicates included: minimum size of 5 ha, slopes of 0-6% and similar fire histories. The 0-3 year sites were longleaf plantations, with no overstory and generally high groundcover. The 8-10 year sites were either longleaf or loblolly plantations as most of the plantations were loblolly before 1996. These sites generally had no overstory cover above 4 m. While all sites were clear-cut, an additional thinning was conducted on 30-80 year sites.

Data Collection:

Five subplots were randomly located within each of the 32 sites. Overstory canopy cover was measured with a concave spherical densiometer by averaging the readings of the four cardinal directions from the center point of the subplot (Lemmon, 1956). From the center point, 3-m transects were established at 0^{o}, 120^{o}, and 240^{o}. Along each transect, woody species' (<2 m in height) foliar cover was measured (Bonham 1989), and species cover values for the 3 transect were averaged. Herbaceous vegetation cover by species was measured using foliar ocular observation in 1 m^2 quadrats at the center point and at the terminus of the 240^o transect. Cover values were recorded as midpoints of a modified Daubenmire scale (trace, 1-5%, 6-10%, 11-15%, 16-26%, 27-49%, 50-80%, 81-95%, or 96-100%) (Daubenmire, 1959). Herbaceous species cover values for the two quadrats were averaged.

An undisturbed soil core (upper 20 cm, 8 cm-diameter) was taken adjacent to each herbaceous quadrat for laboratory bulk density(BD) determination of oven-dried soil (Blake, 1965). To analyze for soil texture, pH, total nitrogen, and total carbon, four surface soil samples (upper 20, 4 cm-diameter) were collected at the terminus of each transect and at each center point. These samples were mixed, homogenized and considered as a composite sample for each subplot. Sand, silt and clay percentage were determined using the hydrometer method (Bouyoucos, 1927). Soil pH was measured using a 1:1 DDI water soil slurry on an Orion SA720 pH meter (Fisher Scientific, Fair Lawn, NJ). Total C and N content was determined on dried, ground soil and detritus samples using a Carlo-Erba NA-1500 CNS Analyzer (Haak-Buchler Instruments, Saddlebrook, NJ).

Data Analysis:

Differences in edaphic variables among the four age classes were analyzed using a generalized linear model (PROC GLM) procedure (SAS 8.0) with age since clear-cut and % clay as main factors. Vegetation data were separated into woody and herbaceous species for the purposes of the analysis. Patterns in species composition in relation to the measured environmental and edaphic variables were analyzed with canonical correspondence analysis (CCA) (CANOCO 4.0) (Ter Braak, 1998). For purposes of analysis, the original number of environmental variables was reduced to eight (age since clear cut (0-3yr., 8-10yr., 15-20yr.), pH, total N, % canopy cover, total C, % sand, and bulk density) due to strong autocorrelations (the variables removed from analysis were 30-80yr., bareground, C/N ratio, %clay, and %silt). Also removed from the analyses were the species Pinus taeda and P. palustris as they were artificially planted on the sites and do not represent natural regeneration.

Statistical validity of the resulting environmental axes was evaluated by an unrestricted Monte Carlo permutation test based on 199 random trials (Ter Braak, 1998). The forward-selection option was used to determine the minimal set of environmental variables that could explain the largest amount of variation in the species composition data. At each step, the statistical significance of the environmental variable added in the course of the forward selection was tested by means of a Monte Carlo permutation test. Variables were significant if the permutation test derived $p<0.05$.

The Dufrene and Legendre (1997) method of calculating species indicator values was used to describe the usefulness of individual species for indicating time since clear-cut. This method produces an indicator value between zero (no indication) and 100 (perfect indication). To receive a perfect score, the presence of a species would point to one of the age classes without error (always present and exclusive to that group). A randomization test was used to test for significance of the indicator value.

RESULTS

Soil Characteristics

Texture analysis revealed discrepancies in the original classification of sites as sandy or loamy based on soil survey information. Twelve sites were misclassified. After reclassification based on % sand from laboratory textural analysis, three 15-20 yr. sites and five 30-80 yr. sites were categorized as loamy sand or sandy loam while five 15-20 yr. sites and, three 30-80 yr. sites were classified as sand. This new site classification was used throughout the analysis. Percent sand ranged from 52.6% to 91.3% while clay ranged from 2.3% to 23.7%, with no significant difference among age classes ($p = 0.698$) (Table 1).

Soil bulk density ranged from 0.73 g/cm^3 in a sandy loam >30 year site to 1.31g/cm^3 in a sand 15-20 year site and decreased significantly ($p= 0.0045$) with clay content. While only marginally significant differences were found among age classes ($p = 0.0785$), BD declined by 12.5% with time since clear cut on sandy sites and only 0.8% on loamy sites. No significant differences were found among age classes for pH, total carbon (TC) and total nitrogen (TN). (Table 2). Mean canopy cover of 3, 31, 69, and 70% for most recent to oldest clearcut sites increased significantly ($p<0.001$) with age class until 15 yrs after clear cutting with no significant differences between 15-20 and 30-80 yr sites.

Vegetation Characteristics – Woody Species

A total of 47 woody species were encountered (Table 3). Species richness in age classes from most recently clearcut to the oldest sites was 36, 31, 37, and 32, respectively. The most abundant and frequent species in all age classes was Rubus sp . Indicator analysis identified Gaylussacia mosieri and Carya spp. as the only significant indicators of age class (Table 4). Gaylussacia mosieri occurred most frequently with highest cover in the 30-80 year old age class but occurred infrequently in younger age classes. Carya sp. were indicators of the 15-18 year age class.

Canonical correspondence analysis (CCA) of woody species and environmental variables identify only marginally significant gradients in species abundance and distribution in relation to environmental variables measured (0-3 yr., 8-10 yr., 15-20 yr, % sand, BD, pH, TC, TN) (Table 5). The first axis was most highly correlated with BD, canopy cover (Canopy) and 8-10 yr. age (Table 6; Figure 1 a) while pH and 0-3 yr. were highly correlated with the second axis and are closely associated in ordination space. Length and direction of arrow in the woody environmental biplot illustrate the strength and directions of these relationships (Fig 1a). The most significant of these was the % canopy cover which captured 17% of the total variance explained by the previously subset variables, followed by BD (16%). These two variables captured 33% of the total variance originally explained by all eight environmental variables. The cumulative variation explained by the first three axes of the species-environment relationship in the CCA was 61.5%. Thus, additional factors not measured contribute to woody species distribution patterns.

Relationships among age classes can be inferred from relative position within the CCA diagram. The 0-3 yr. and 8-10 yr. age classes are located in the same quadrant indicating species are not well distinguished between these two age classes. Further, the location of the 15-20 yr age class near the ordination origin suggests a unique woody vegetation community does not dominate this age class. Many woody species are located close to the origin in the species ordination biplot further illustrating the poor differentiation of woody species in association with any of the environmental variables measured (Figure 1b). The 30-80 year age class, although not in the analysis, would be placed in close proximity to the "C30" and "S30" sites found in Figure 1c, which corresponds to the placement of the environmental variable canopy; increases in % canopy cover are associated with the older sites. Also of note is the relative placement of the oldest and youngest sites (Figure 1c). Since they are located in opposite quadrants of the biplot, it can be inferred that species distribution differs most between these sites.

Vegetation Characteristics - Herbaceous Species

One hundred and fifty eight herbaceous ground cover species were encountered. (Table 7). Species richness for age classes from most recently clearcut to oldest sites was 80, 61, 79, and 71/ 80 m^2, respectively. Many species were rare with a total of 57 (approximately 36% of all herbaceous sp.) occurring once in 32 sites, 22 (~ 14%) twice, and 12 (~ 8%) three times. Mean species richness (10 m^2 per site), total cover and total forb cover did not differ significantly among age classes. However, mean total grass cover of 36% for 30 yr- sites was greater than that of 0-3 and 15-20 age classes. Mean grass cover of 22%, 32 %, and 21 % for 0-3, 8-10 and 15-20 yr. classes, respectively, did not differ significantly. Mean percent bare ground decreased significantly 15 yrs. after clear-cut (35%, 24 %, 7% and 7 % for 0-3, 8-10, 15-20 and 30 yr sites, respectively) and was a significant indicator of the 0-3 age class (p=0.05). Cover and frequency of individual herbaceous species also differed across age classes (Table 7). After removal of a single outlier (site with low sand content 52%), indicator analysis identified several species representative of each of the 4 age classes. Analysis based on 31 sites with % sand ranging from 67-91% identified, Cyperus croceus as a significant and Bulbostylis barbata as a marginally significant (p=0.0630) indicator of the 0-3 age class (Table 4). Andropogon virginicus v. virginicus, Dichanthelium sp., unidentified moss and Sporobolus junceus were significant indicators of the 8-10 yr. age class. Andropogon virginicus v. viginicus occurred almost exclusively in the 8-10 yr. age class. Pityopsis species and Tridens flavus were indicators of 15-20 yr class. Andropogon ternarius, Schizachyrium scoparium, Desmodium sp., Hieracium sp., Rhynchosia tomentosa *(*marginally significant*)* were indicators of 30-80 yr sites. Schizachyrium scoparium and Andropogon ternarius are difficult to differentiate in field sampling when floral parts are unavailable. Therefore, values for these two species and those that could not be differentiated as either were summed. Indicator analysis found this complex is a significant indicator of the 30-80 yr. age class.

Because of the large number of rare species, CCA of herbaceous species and environmental variables (age, pH, %canopy cover, total N, total C, % clay, and bulk density) was performed with and without down-weighting. A comparison of the results is shown in Table 8, with both analyses showing overall significance between the environmental variables and species composition. In comparison to the woody ordination biplots (Figs 1a-c), herbaceous species (Figure 3b) show many more species towards the outer edges of the ordination diagram indicating that environmental variables had a greater effect on the distribution of herbaceous species' than they did on the woody species.

For both CCA analyses, the cumulative variation explained by the first three axes of the species-environment relationship was less than 40%. The high correlations between the environmental and species axes do indicate however, that the environmental variables reasonably explained the first two ordination axes (Table 9, Figures 3a and 1a).

In comparison to woody ordination biplots (Figs 1a-c), the length and direction of arrows associated with various age classes (Figure 3a), occurrence of many more herbaceous species towards the outer edges of species biplots (Figure 3b), better clustering of similar age plots and segregation of dissimilar aged plots (Figure 3c) in the herbaceous biplots suggest age since clearcut was more influential in herbaceous species distribution. Forward selection and unrestricted Monte Carlo permutation tests indicated three (8-10 yr., 0-3 yr., and BD) of the nine environmental variables previously selected made statistically significant contributions to explaining the variance in the herbaceous vegetation data. The most significant of these was the 8-10 yr which captured 20% of the total variance explained by the previously subset variables, followed by 0-3 yr (16%) and bulk density (13%). These three variables thus captured 49% of the total variance originally explained by all nine environmental variables. The first CCA axis separated 8-10 yr sites from all other sites while the second axis separated 0-3 yr sites for the most part from older sites. Increased bulk density was closely related to 0-3 yr age class.

DISCUSSION

Sites showed similarities in soil properties regardless of stand age as has been found in other studies. Soil properties were comparable in Appalachian hardwood stands of 20 and 80 yrs (Gilliam and Turrill, 1993) while sugar maple stands in Michigan 50 yrs post clear-cut resembled uncut sites in terms of soil pH, texture, Ca, and Mg (Albert and Barnes, 1987). While edaphic variables measured did not differ significantly among age classes, BD and pH exhibited a decreasing trend from the 0-3 to the 30-80 yr sites. Bulk density was significantly related to soil texture, which has been found in other studies (Will et al., 2002). Koger et al. (1985) found much higher bulk density measurements in sandy loam soils compared to clay soils after multiple pass attempts of skidder tires. Recovery to pre-harvest BD has been shown to differ with respect to soil texture. Clay soils that swell and shrink may partially recover with subsequent wetting and drying (Barzegar et al., 1995), whereas recovery in sandy soils can be slow or even non-existent (Greacen and Sands, 1980). However, a study in Colorado forests reported variation in soil texture, but no significant effects of harvest on bulk density (Whitecotton et al., 2000)

Species richness did not differ among age classes for either woody or herbaceous species, while species distribution and abundance did. Studies on disturbance report increases, decreases, and no change in species richness (Halpern and Spies, 1995; Roberts and Gilliam, 1995; Wender et al., 1999; Ford et al., 2000). At least in Fort Benning, it is clear that individual species rather than diversity will be more informative for revealing landscape recovery following clear cuts.

Of the variables measured, distribution and abundance of woody understory species were most strongly affected by % canopy cover and bulk density, both potentially related to silvicultural activity. However, there was considerable overlap in species composition across this age gradient. An increase in canopy cover was the most important factor in distinguishing between 0-3 yr and 15-20 yr sites. Cowell (1995) found that woody species in Piedmont forests were influenced most by topography and then by the amount of sand fraction in the surface soil. Explanations for this discrepancy in primary gradients may include a larger range of % sand in Cowell's study compared to this study, as well as the potential interacting effect of compaction and soil texture.

Within any given age class, variance in BD was greater in loamy sites than sandy sites, which may reflect differences in soil moisture at time of clearcut. Wet soils are more susceptible to compaction than dry soils (Lovich and Bainbridge, 1999). Finer soil particles such as clay have the ability to hold more moisture due to increased surface area. When medium and fine-textured soils are wet, loads of heavy logs and machinery can cause soil failure, resulting in a partial collapse of soil structural aggregation (Richter, 2000). Under dry conditions, increases in soil bulk density tend to be confined to the surface, whereas in wet conditions significant change can occur to greater depths (Miles, 1978) Also, as time since harvest increased, BD demonstrated a downward trend, potentially contributing to changes in woody species distribution and root growth in the upper soil layers.

Most studies of vegetation recovery after a disturbance do not look at individual species responses but rather use diversity or overall cover and abundance as a measure of the understory species. A study using similar disturbances as this one found that clearcutting and planting slash pine resulted in a decrease of woody species and an increase of herbaceous species, while overall species richness did not change with treatment (Conde et al., 1983). More specifically, of the woody species only Rubus sp. and Hypericum increased in abundance. These findings are similar to those in this study, although recovery was followed for only 2 yrs.

Our results suggest composition and cover of herbaceous species' is more indicative of recovery time than woody species. Herbaceous species may be more sensitive than trees and shrubs to local edaphic variation (Drewa et al., 2002), and thus possibly to disturbances that alter soil characteristics. Generally, compared to herbaceous species, woody species are more broadly distributed, animal dispersed, and have underground root systems that facilitate rapid aboveground regrowth and vegetative spread. This allows greater adaptation to disturbance and thus less responsiveness to change (Olson and Platt, 1995; Gile et al., 1997). The important environmental gradients shaping herbaceous species composition were age class (8-10 yr and 0-3 yr) and BD.

Andropogon spp., Dichanthelium spp., and Aristida spp. have all been found to be more abundant soon after a disturbance, followed by a slow decrease in frequency and abundance over time (Lemon, 1949; Grelen, 1962; Greenberg et al., 1995). Increases in perennial grasses such as Andropogon spp. and Dichanthelium spp. following disturbance may be partially attributable to resprouting (Schmalzer and Hinkle 1992). Schizachyrium scoparium and Andropogon ternarius were associated with 30-80 yr sites. Schizachyrium scoparium is considered a late successional plant throughout it range. While they occurred in all age classes, both increased with recovery time and had higher frequency and cover values on the oldest sites. In a similar chronosequence study, Provencher et al. (1997) found these grasses as potentially successful indicators of varying levels of recovery after disturbance. Provencher et al. grouped Aristida spp., Andropogon spp., and 11 species of Dichanthelium together as concomitant with soil disturbance and decreasing over recovery time. Schizachyrium scoparium and Andropogon ternarius were associated with mid- to late-successional stages, increased with recovery time and peaked 50 yrs after disturbance. Further, Schizachyrium scoparium and Rhynchospora grayi were the only species identified by Provencher et al. (2001) as indicating a recovered condition and perhaps high quality groundcover.

Bulbostylis barbata and Pityopsis spp were identified as indicators of younger sites. Several herbaceous species of Bulbostylis, Pityopsis and Eupatorium genera have been found to be absent from mature forest older than 55 yrs (Greenberg et al., 1995).

To a great extent herbaceous species composition was similar in the two oldest age classes. In fact, some studies found herbaceous species composition undergoes very little change after 20 yrs of recovery from disturbance (Davison and Forman, 1982; Kochenderfer and Wendel, 1983; Gilliam and Turrill, 1993). This has been attributed to quick recolonization by groundcover species. Percent bare ground was similar between the oldest two age classes and youngest two age classes and may serve as an indicator to quickly and easily differentiate between disturbed and recovered sites. Provencher et al. (2001) also identified bare ground as an indicator of soil disturbance. However, highly sensitive species may take longer to reestablish after disturbance. For example, wiregrass (Aristida beryrinchiana) an important indicator species in more southern longleaf pine forests, did not recover on clearcut sites for 90 yrs (Provencher et al., 2001).

Generally, only a few species stand out as possible indicators of recovery after silvicultural disturbance. Several studies have found successful herbaceous understory indicators of pine tree establishment and growth (Strong et al., 1991; Dibble et al., 1999), although there is the question of whether successful pine growth can be a proxy for overall landscape health. One of the criteria for a successful indicator is for the species to have low variability in response to change in environmental conditions (Dale and Beyeler, 2001). While we found several herbaceous species to serve well for indicators in this study, further studies incorporating more sites and a wider range of soil textures and age classes would help to test the validity of these species as indicators in a broader geographic context.

It is critical to note however, that indicator species found to be useful in the Fort Benning landscape may not be generalized. A regional synthesis of longleaf pine vegetation revealed high variation in understory vegetation structure and composition due to fire regime, soil moisture, soil texture, geographic location, and anthropogenic disturbances (Rodgers and Provencher, 1999). Further, sandhills may have highly heterogeneous soil depth and moisture, possibly contributing to plant species richness and community composition (Provencher et al., 1997). Thus, further studies are necessary in order to verify the success of potential indicators outside of this localized area. Rodgers and Provencher (1999) suggest the use of controlled studies to adequately assess historical species distributions and recovery pathways following disturbance.

The importance of this study lies in the fact that it looks for patterns in understory species composition – which are often overlooked in managed southeastern pine forests. The understory vegetation, especially in longleaf pine habitat, is critical in maintaining faunal diversity and ecosystem function. Thus it is imperative for land managers to realize that complete recovery for soil and understory vegetation may be longer than the normal rotation length for longleaf and loblolly pine plantations on Fort Benning (Zabowski et al., 1996).

REFERENCES

Albert, D.A., Barnes, B.V., 1987. Effects of clear-cutting on the vegetation and soil of a sugar maple-dominated ecosystem, western Upper Michigan. For. Ecol. Mgmt. 18, 283-298.

Bailey, R.G., 1995. Description of the ecoregions of the United States. USDA Forestry Service, Ogden, Utah.

Barzegar, A.R., Rengasamy, P., Oades, J.M., 1995. Effects of clay type and rate of wetting on the mellowing of compacted soils. Geoderma 68, 39-49.

Blake, G.R., 1965. Bulk density. In: Black, C.A. (Ed.), Methods of soil analysis. American Society of Agronomy. Madison, WI, pp. 374-390.

Bonham, C.D., 1989. Measurements for Terrestrial Vegetation. Wiley, New York.

Bouyoucos, G., 1927. The hydrometer as a new method for the mechanical analysis of soils. Soil Sci. 23, 343-352.

Clements, F.E., 1928. Plant succession and indicators. H.W. Wilson, New York.

Clewell, A.F., 1989. Natural history of wiregrass (Aristida stricta Michx., Gramineae). Nat. Areas J. 9, 223-233.

Conde, L.F., Swindel, B.F., Smith, J.E. 1983. Plant species cover, frequency, and biomass: early responses to clearcutting, burning, windrowning, discing, and bedding in Pinus elliottii flatwoods. For. Ecol. Mgmt. 6, 319-331.

Congdon, R.A., Herbohn, J.L., 1993. Ecosystem dynamics of disturbed and undisturbed sites in north Queensland wet tropical rain forest. 1. Pluristic composition, climate and soil chemistry. J. Trop. Ecol. 9, 349-363.

Cooper, W., 1989. Ecology of the pine-hardwood type. In: Waldrop T.A. (Ed), Proceedings of Pine-Hardwood Mixtures: A Symposium on Management and Ecology of the Type, Atlanta, GA. U.S.D.A. Forest Service Southeastern Forest Experiment Station, Asheville, NC. General Technical Report SE-58, p. 271.

Cowell, C.M., 1995. Presettlement piedmont forests: patterns of composition and disturbance in central Georgia. Ann. Assoc. Am. Geogr. 85, 65-83.

Dale, V.H., Beyeler, S.C., 2001. Challenges in the development and use of ecological indicators. Ecological Indicators 1, 3-10.

Dale, V.H., Beyeler, S.C., Jackson, B., 2002. Understory vegetation indicators of anthropogenic disturbance in longleaf pine forests at Fort Benning, Georgia, USA. Ecological Indicators 1, 155-170.

Daubenmire, R.F., 1959. Canopy coverage method of vegetation analysis. Northwest Sci. 3, 43-64.

Davison, S.E., Forman, R.T.T., 1982. Herb and shrub dynamics in a mature oak forest: a thirty-year study. Bull. Torrey Bot. Club 109, 64-73.

Dibble, A.C., Brissette, J.C., Hunter Jr., M.L., 1999. Putting community data to work: some under-story plants indicate red spruce regeneration habitat. For. Ecol. Mgmt. 114, 275–291.

Drewa, P.B., Platt, W.J., Moser, E.B., 2002. Community structure along elevation gradients in headwater regions of longleaf pine savannas. Plant Ecol. 160, 61-78.

Duffy, D.C., Meier, A.J., 1992. Do Appalachian herbaceous understories ever recover from clearcutting? Conserv. Biol. 6, 196-201.

Dufrene, M., Legendre, P., 1997. Species assemblages and indicator species: the need for a flexible asymmetrical approach. Ecol. Mongr. 67, 345-366.

Echternacht, A.C., Harris, L.D., 1993. The fauna and wildlife of the Southeastern United States. In: Martin, W.H., Boiyce, S.G., Echternacht, A.C., (eds.), Biodiversity of the Southeastern United States: lowland terrestrial communities. John Wiley & Sons, Inc. pp. 81-116.

Elliott, K.J., Boring, L.R., Swank, W.T., Haines, B.R., 1997. Successional changes in plant species diversity and composition after clearcutting a Southern Appalachian watershed. For. Ecol. Mgmt. 92, 67–85.

Ercelawn, A. 1999. End of the road - the adverse ecological impacts of roads and logging: a compilation of independently reviewed research. Natural Resources Defense Council, New York.

Ferry, G.W., Clark, R.G., Montgomery, R.E., Mutch, R.W., Leenhouts, W.P., Zimmerman, G.T., 1995. Altered fire regimes within fire-adapted ecosystems. In: LaRoe, E.T., Farris, G.S., Puckett, C.E., Doran, P.D., Mac, M.J., (eds.), Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals, and ecosystems. U.S. Department of the Interior, National Biological Service, Washington, D.C., pp. 222-224.

Fimbel, R.A., Robinson, J.G., Grajal, A., 2001. The cutting edge: conserving wildlife in logged tropical forests. Columbia University Press, New York.

Ford, W.M., Odom, R.H., Hale, P.E., Chapman, B.R., 2000. Stand-age, stand characteristics, and landform effects on understory herbaceous communities in southern Appalachian cove-hardwoods. Biol. Conserv. 93, 237-246.

Gayoso, J., Iroume, A., 1991. Compaction and soil disturbances from logging in Southern Chile. Annales des Sciences Forestieres 48, 63-71.

Gile, L.H., Gibbens, R.P., Lenz, J.M., 1997. The near-ubiquitous pedogenic world of mesquite roots in an arid basin floor. J. Arid Environ. 35, 39-58.

Gilliam, F.S., Turril, 1993. Herbaceous layer cover and biomass in a young versus a mature stand of a central Appalachian hardwood forest. Bull. Torrey Bot. Club 120, 445–450.

Gilliam, F.S., 2002. Effects of harvesting on herbaceous layer diversity of a central Appalachian hardwood forest in West Virginia, USA. For. Ecol. Mgmt. 155, 33-43.

Goebel, P.C., Hix, D.M., Olivero, A.M., 1996. Seasonal ground-flora patterns and site factor relationships of second-growth and old-growth south-facing forest ecosystems, southeastern Ohio, USA. Nat. Areas J. 19, 12–19.

Greacen, E.L. Sands, R., 1980. Compaction of forest soils. A review. Australian Journal of Soil Research 18, 163-189.

Greenberg, C.H., Neary, D.G.,Harris, L.D., Linda, S.P., 1995. Vegetation recovery following high-intensity wildfire and silvicultural treatments in sand pine scrub. Am. Midl. Nat. 133, 149-163.

Grelen, H.E. 1962. Plant succession on cleared sandhills in northwest Florida. Am. Midl. Nat. 67, 36-44.

Halpern, C.B. Spies, T.A., 1995. Plant species diversity in natural and managed forests of the Pacific Northwest. Ecol. Appl. 5, 913–934.

Hatchell, G.E., Ralston, G.W.,Foil, R.R., 1970. Soil disturbance in logging. J. For. 68, 772-775.

Hunter, M.L., Jr. 1990. Wildlife, Forests, and Forestry. Prentice Hall, Englewood Cliffs, NJ.

Huth, A., Ditzer, T., 2001. Long-term impacts of logging in a tropical rain forest - a simulation study. For. Ecol. Mgmt. 142, 35-51.

Kane, S., Keeton, R., 1998. Fort Benning: the land and the people. Southeast Archaelogical Center, National Park Service, Tallahassee, FL.
King, T.G., Howell, M.A., Chapman, B.R., 1998. Comparisons of wintering bird communities in mature pine stands managed by prescribed burning. Wilson Bull. 110, 570-574.
Kochenderfer, J.N. Wendel, G.W., 1983. Plant succession and hydrologic recovery on a deforested and herbicided watershed. For. Sci. 29, 545–558.
Koger, J.L., Burt, E.C., Trouse Jr., A.C., 1985. Multiple pass effects of skidder tires on soil compaction. Trans. ASAE 28, 11-16.
Landers, J.L., Lear, D.H.V., Boyer, W.D., 1995. The longleaf pine forests of the southeast: requiem or renaissance? J. For. 93, 39-44.
Lemmon, P.E. 1956. A spherical densiometer for estimating forest overstory density. For. Sci. 2, 314-320.
Lemon, P.C. 1949. Successional responses of herbs in the longleaf-slash pine forest after fire. Ecology 30, 135-145.
Lovich, J.E. Bainbridge, D., 1999. Anthropogenic degradation of the southern California desert ecosystem and prospects for natural recovery and restoration. Environ. Manage. 24, 309-326.
Miles, J.A. 1978. Soil compaction produced by logging and residue treatment. Trans. ASAE 21, 60-62.
Myers, R.L. 1990. Scrub and high pine. In: Myers, R.L., Ewel, J.J., (Eds.), Ecosystems of Florida. University of Central Florida Press, Orlando, FL, pp. 150-193.
Noss, R.F. 1989. Longleaf pine and wiregrass: keystone components of an endangered ecosystem. Nat. Areas J. 9, 211-213.
Noss, R.F., LaRoe III, E.T., Scott, J.M., 1995. Endangered ecosystems of the United States: a preliminary assessment of loss and degradation. National Biological Service, Biological Report 28.
Olson, M.S., Platt, W.J., 1995. Effects of growing season firest on resprouting of shrubs in longleaf pine savannas. Vegetatio 119, 101-118.
Pierce, R.S., Hornbeck, J.W., Martin, C.W., Tritton, L.M., Smith, C.T., Federer, C.A.,Yaeney, H.W. Yaeney., 1993. Whole-tree clearcutting in New England: manager's guide to impacts on soils, streams, and regeneration. USDA Forest Service General Technical Report NE-172.
Provencher, L., Rodgers, H.L., Galley, K.E.M., Hardesty, J.L., Tanner, G.W., Gordon, D.R., McAdoo, J.P., Sheehan, J., Brennan, L.A., 1997. Initial post-treatment analysis of restoration effects on plants, invertebrates, and birds in sandhill systems at Eglin Air Force Base, Florida. Annual Report to Natural Resources Division, Eglin Air Force Base, Niceville, FL. Public Lands Program, The Nature Conservancy, Gainesville, FL.
Provencher, L., Litt, R., Galley, K.E.M., Gordon, D. R., Tanner, G.W., Brennan, L.A., Gobris, N. M., McAdoo, S.J., McAdoo, J.P., Herring, B.J., 2001. Restoration of fire-suppressed longleaf pine sandhills at Eglin Air Force Base, Florida. Final report to the Natural Resources Management Division, Eglin Air Force Base, Niceville, FL. Science Division, The Nature Conservancy, Gainseville, FL.
Qian, H., Klinka, K., Sivak, B., 1997. Diversity of the under story vascular vegetation in 40 year-old and old-growth forest stands on Vancouver Island, British Columbia. J. Veg. Sci. 8, 773–780.

Reiners, W.A., 1992. Twenty yrs of ecosystem reorganization following experimental deforestation and re-growth suppression. Ecol Mongr* 62, 503–523.
Richter, D.D. 2000. Soil and water effects of modern forest harvest practices in North Carolina. Individual White Paper associated with the "Economic and Ecologic Impacts Associated with Wood Chip Production in North Carolina Study," Durham, NC: Southern Center for Sustainable Forests, Duke University. http://www.env.duke.edu/scsf/. Accessed February 2, 2003.
Roberts, M.R., Gilliam, F.S., 1995. Disturbance effects on herbaceous layer vegetation and soil nutrients in Populus forests of northern lower Michigan. J. Veg. Sci. 6, 903–912.
Rodgers, H.L., Provencher, L., 1999. Analysis of longleaf pine sandhill vegetation in northwest Florida. Castanea 64, 138-162.
SAS Institute Inc., 1989. SAS for Windows, Version 8.0, Cary, North Carolina.
Schmalzer, P.A. Hinkle, C.R., 1992. Recovery of oak-saw palmetto scrub after fire. Castanea 57, 158-173.
Stearns, F., Likens, G.E., 2002. One hundred yrs of recovery of a pine forest in northern Wisconsin. Am. Midl. Nat. 148, 2-19.
Stout, I.J., Marion, W.R., 1993. Pine flatwoods and xeric pine forests of the southern (lower) Coastal Plain. In: Martin, W.H., Boyce, S.G., Echternacht, A.C., (Eds.), Biodiversity of the southeastern United States: lowland terrestrial communities. John Wiley & Sons, NY, pp. 373-446.
Strong, W.L., Bluth, D.J., LaRoi, G.H., Corns, I.G.W., 1991. Forest understory plants as predictors of lodgepole pine and white spruce site quality in west-central Alberta. C J For Res 21, 1675–1683.
Ter Braak, C.J.F, Smilauer, P., 1998. CANOCO reference manual and user’s guide to Canoco for Windows software for canonical community ordination (Version 4). Microcomputer Power, Ithaca, NY.
USDA NRCS, 1983. Soil Survey of Muscogee County, Georgia. U.S. Government Printing Office, Washington, D.C.
USDA NRCS, 1997. Soil Survey of Chattahoochee and Marion Counties, Georgia. U.S. Government Printing Office, Washington, D.C.
Walker, J. 1993. Rare vascular plant taxa associated with the longleaf pine ecosystems: patterns in taxonomy and ecology. In: Hermann, S.M. (Ed.), The Longleaf Pine Ecosystem: Ecology, Restoration, and Management. Proceedings of the Tall Timbers Fire Ecology Conference, No. 18. Tall Timbers Research Station, Tallahassee, FL, pp. 105-127.
Wender, B.W., Hood, S.M., Smith, D.W., Zedaker, S.M., Loftis, D.L., 1999. Response of vascular plant communities to harvest in southern Appalachian mixed-oak forests: two-year results. In: Proceedings of the Tenth Southern Biennial Silvicultural Research Conference, Shreveport, LA, February 16-18, 1999, USDA Forest Service General Technical Report SRS-30, pp. 34–38.
Wert, S., Thomas, B.R., 1981. Effects of skid roads on diameter, height, ad volume growth in Douglas fir. Soil Sci. Soc. Am. J. 45, 629-632.
Whitecotton, R.C.A., David, M.B., Darmody, R.G., Price, D.L., 2000. Impact of foot traffic from military training on soil and vegetation properties. Environ. Manage. 26, 697-706.

Will R.E., Wheeler, M.J., Markewitz, D., Jacobson, M.A., Shirley, A.M., 2002. Early loblolly pine stand response to tillage on the Piedmont and Upper Coastal Plain of Georgia: tree allometry, foliar nitrogen concentration, soil bulk density, soil moisture, and soil nitrogen status. South. J. Appl. For. 26, 190-196.

Zabowski, D., Rygiewicz, P.J., Skinner, M.F., 1996. Site disturbance effects on a clay soil under radiata pine. 1. Soil solutions and clay mineral stability. Plant Soil 186, 343-351.

Table 1: Mean percent sand, silt and clay surface soil collected for four age representing yrs post clear cut.

Age (yr)	Sand (%)	Silt (%)	Clay (%)
0-3	85.36	9.75	4.86
8-10	79.89	12.35	7.77
15-20	83.78	10.26	5.96
30-80	81.08	11.25	7.67

Table 2: Mean and variance for bulk density (BD), pH, total carbon (TC) and total nitrogen (TN) for four age classes representing yrs post a clear cut, divided into sandy and loamy surface soils groups.

Age Class (yr)	**BD (g/cm^3)**				**PH**			
	Sandy		Loamy		Sandy		Loamy	
	Mean	Var	Mean	Var	Mean	Var	Mean	Var
0-3	1.16	0.003	1.04	0.0083	5.48	0.00541	5.75	0.0673
8-10	1.10	0.002	1.05	0.0049	5.44	0.0363	5.54	0.0325
18-20	1.06	6E-05	1.03	0.0038	5.28	0.0704	5.46	0.229
30-80	1.02	0.002	1.03	0.006	5.42	0.00488	5.24	0.0477

Age Class (yr)	**TC (g/Kg)**				**TN (g/Kg)**			
	Sandy		Loamy		Sandy		Loamy	
	Mean	Var	Mean	Var	Mean	Var	Mean	Var
0-3	9.59	1.87	9.5	3.60	0.358	0.00257	0.390	0.0228
8-10	8.79	1.86	9.82	8.67	0.336	0.00117	0.426	0.0263
18-20	9.21	10.97	9.55	5.63	0.342	0.00757	0.381	0.0158
30-80	10.16	4.72	7.74	1.75	0.289	0.00699	0.302	0.00382

Table 3: Mean percent cover and frequency (based on 80 subplots per age class) for woody species in four age classes indicating yrs post clear cut.

Woody Species	**Code**	**0-3**		**8-10**		**15-18**		**30-80**	
		Cover (%)	Freq. (%)	Cover (%)	Freq. (%)	Cover (%)	Freq. (%)	Cover (%)	Freq. (%)
Acer rubrum	acru	0	0	0	0	t	1	0	0
Callicarpa americana	caam	0.4	3	0	0	0.4	2	0	0
Campsis radicans	cara	t *	1	0	0	0	0	0	0
Carya sp.	cas	t	1	t	1	0.5	4	0.1	1
Cornus florida	cofl	0.1	1	0	0	0.1	1	0	0
Corylus americana	haz	0	0	0	0	0.1	1	0	0
Crataegus sp.	crs	1.	11	2.7	9	0.4	5	0.1	5
Diospyros virginiana	divi	0.4	5	0.7	3	0.6	9	0.1	3
Gaylussacia mosieri	gamo	t	2	0.1	3	0.2	2	0.7	9
Gelsemium sempervirens	gese	0.2	3	0.3	3	0.5	9	0.7	6
Hypericum gentianoides	hyge	0.3	7	0.1	3	0	0	0.2	3
Hypericum hypericoides	hyhy	0	0	0.1	2	0.1	1	t	1
Ilex decidua	ilde	0	0	0	0	0	0	0.1	1
Ilex glabra	ilgl	0.2	1	0	0	1.3	5	0.3	2
Liquidambar styraciflua	list	1.8	11	2.0	12	9.0	17	3.0	16
Lonicera sp.	los	0.1	1	0	0	0	0	0	0
Morella cerifera	myce	0.1	1	0.8	2	1.1	6	1.6	8
Parthenocissus quinquefolia	paqu	0	0	t	1	t	1	0	0
Pinus glabra	pigl	0.1	1	0	0	0	0	0	0
Pinus palustris	pipa	1.2	7	7.0	16	0.1	1	0.1	1
Pinus taeda	pita	0.1	1	6.2	17	0.4	5	0.1	2
Prunus caroliniana	prca	t	1	0	0	0	0	0	0
Prunus serotina	prse	0.1	2	0	0	0	0	0.1	1
Quercus alba	qual	0.1	1	0.2	1	0.5	3	0	0
Quercus falcata	qufa	0.5	6	0.6	3	1.6	9	0.4	4
Quercus incana	quin	0.6	8	1.3	5	0.1	1	0.1	1
Quercus laevis	qula	0.1	1	1.3	5	1.3	4	0.2	1
Quercus laurifolia	qula	0.4	7	1.0	8	0.4	6	0.6	10
Quercus marilandica	quma	0	0	0.2	1	0.9	2	0	0
Quercus minima	qumi	0.4	2	0	0	0	0	0	0
Quercus nigra	quni	0.2	2	0.6	6	t	1	0.1	2
Quercus sp. seedling	qus	t	2	0	0	0	0	0	0
Quercus stellata	qust	0.6	6	0.2	1	0.6	5	0.1	1
Rhus copallinum	rhco	2.0	21	2.4	14	4.5	28	1.7	18
Rubus sp.	rus	8.3	34	6.7	31	10.6	36	3.7	21

Sassafras albidum	saal	0.1	2	0	0	0.2	1	0.9	5
Smilax sp.	sms	1.3	11	0.3	8	2.5	15	0.9	14
Toxicodendron pubescens	topu	t	2	0.4	3	0.2	2	t	2
Toxicodendron radicans	tora	0	0	0.1	2	0.1	3	t	1
Ulmus alata	ulal	0.3	4	0	0	0.2	2	0	0
Ulmus americana	ulam	0	0	0	0	0	0	t	1
Vaccinium arboreum	vaar	0.8	10	1.2	11	2.1	10	2.8	18
Vaccinium elliottii	vael	0.1	3	0.3	5	0.2	2	0.2	6
Vaccinium myrsinites	vamy	0	0	0.3	1	0.8	7	0.6	3
Vaccinium stamineum	vast	0	0	t	1	0.2	2	0.5	3
Vitis rotundifolia	viro	0.3	3	t	2	0.7	8	t	2
Yucca sp.	yus	0	0	0.2	1	0.3	1	0	0

* t = trace = <.1

Table 4. Post clear cut age class, indicator value and significance for species identified as indicators.

Species	Age Group (yrs)	Indicator Value	p-value
Bulbostylis barbata	0-3	36.2	0.063
Cyperus croceus	0-3	43.7	0.034
Andropogon virginicus	8-10	62.1	0.002
Dichanthelium species	8-10	41.1	0.012
Unknown moss like	8-10	50.3	0.008
Sporobolus junceus	8-10	28.6	0.044
Pityopsis species	15-20	44.5	0.020
Tridens flavus	15-20	40.2	0.048
Desmodium species	15-20	39.8	0.034

Carya species	15-20	31.1	0.072
Andropogon ternarius	30-80	36.3	0.028
Schizachyrium scoparium	30-80	49.0	0.011
Schizachyrium/Andropogon ternarius Complex	30-80	47.1	0.019
Hieracium species	30-80	36.7	0.024
Rhynchosia tomentosa	30-80	32.9	0.088
Gaylussacia mosieri	30-80	43.7	0.028

Table 5: Significance of eigenvalues of the woody species relationship to the environmental variables as determined by the canonical correspondence analysis (CCA).

	Eigenvalue	F-ratio	p-value
1st Canonical Axis	0.307	2.537	0.070
All Canonical Axes	0.984	1.209	0.090

Table 6: Weighted correlation matrix for species axes, environmental axes, and environmental variables for woody species, using average measures for the 32 sites.

	SP AX1	SP AX2	SP AX3	ENV AX1	ENV AX2	ENV AX3
SP AX1	1.000					
SP AX2	0.022	1.000				
SP AX3	0.092	-0.054	1.000			
ENV AX1	0.808	0.000	0.000	1.000		
ENV AX2	0.000	0.860	0.000	0.000	1.000	
ENV AX3	0.000	0.000	0.834	0.000	0.000	1.000
BD	0.460	0.353	-0.047	0.569	0.411	-0.056
%Clay	-0.384	-0.340	0.079	-0.475	-0.395	0.094
TC	-0.120	0.151	-0.531	-0.149	0.176	-0.637
TN	-0.186	-0.142	-0.505	-0.230	-0.165	-0.605
pH	0.045	-0.583	0.218	0.056	-0.677	0.261
0-3	0.017	-0.510	0.000	0.021	-0.593	0.000
8-10	0.407	-0.094	0.257	0.504	-0.109	0.308
15-20	-0.187	0.107	-0.592	-0.232	0.125	-0.710
Canopy	-0.426	0.478	-0.037	-0.527	0.556	-0.044
Eigenvalue	0.307	0.178	0.120			

Table 7: Mean aerial cover (%) and frequency (%) (based on 80 subplots per age class) for herbaceous species in four age classes indicating yrs post clear cut.

Herbaceous Species		**0-3**		**8-10**		**15-20**		**30-80**	
		Cover (%)	Freq. (%)	Cover (%)	Freq. (%)	Cover (%)	Freq. (%)	Cover (%)	Freq. (%)
Acalypha gracilens	acgr	0.5	12	1.3	26	0.4	14	1.3	23
Acanthaceae fam	aca	0	0	0	0	0	0	t*	1
Agalinis setacea	agse	0.3	2	t	1	0	0	0.2	2
Agrimonia microcarpa	agmi	0	0	0	0	0	0	0.2	1
Andropogon gerardii	ange	0.1	2	t	1	t	1	0	0
Andropogon gyrans	angy	0	0	0	0	0	0	0.2	1
Andropogon virginicus	anvi	0.2	1	7.0	27	0	0	1.0	4
Andropogon ternarius	scte	1.0	5	0.7	5	0.8	7	2.7	30
Apiaceae fam	api	0.1	2	t	2	0.2	7	0.3	3
Aristida purpurascens	arpu	0.1	1	t	1	0	0	0	0
Aristida sp.	ars	0.7	6	5.3	39	1.4	14	2.2	27
Arundinaria gigantea	argi	0	0	0	0	0.2	2	0	0
Symphyotrichum concolor	asco	0.4	5	0.5	4	t	1	0.3	4
Symphyotrichum dumosum var. dumosum	asdu	1.9	14	0.3	14	1.8	20	2.3	23
Ionactis linariifolius	asli	0	0	0	0	0	0	0.3	2
Symphyotrichum patens var. patens	aspa	0	0	0	0	0.2	3	0.1	1
Sericocarpus asteroides	aspt	0	0	0.1	1	0.2	2	0	0
Sericocarpus linifolius	asso	0.2	3	0	0	0	0	0.1	2
Aster sp.	ass	t	1	0.1	3	0.1	3	0.5	9
Sericocarpus tortifolius	asto	0.4	4	0.3	7	0.2	6	0.6	10
Bulbostylis barbata	buba	1.2	7	t	1	0	0	0	0
Centrosema virginianum	cevi	0.2	2	0	0	0.3	5	0.3	5
Cercis canadensis	ceca	0	0	0	0	t	1	0	0
Chamaecrista fasciculata	chfa	0.6	13	0.2	6	1.2	21	1.1	17
Chasmanthium sessiliflorum	chla	0.6	6	0	0	0.2	2	0	0
Chrysopsis mariana	chma	0	0	0	0	t	1	0.1	1
Cirsium sp.	cis	0	0	0	0	0	0	0.1	1
Conyza Canadensis	coca	1.2	21	0.9	10	0.2	3	0	0
Coreopsis sp.	cos	1.0	23	3.5	42	2.7	35	3	30
Crotalaria rotundifolia	crro	0	0	0.2	3	0	0	0	0
Croton michauxii	crli	0	0	t	1	0	0	0	0
Cyperus croceus	cycr	0.4	11	0	0	t	1	t	2
Dalea sp.	das	0	0	0	0	t	1	0	0
Desmodium rotundifolium	dero	0	0	0	0	t	3	0.1	1
Desmodium sp.	des	0.4	6	0	0	0.4	4	1.2	10
Dichanthelium sp.	dis	7.7	65	12.7	74	5.8	59	4.9	46
Digitaria cognata	dico	0.1	2	0.1	3	t	1	0	0
Digitaria filiformis var. filiformis	difi	1.1	4	0.5	5	0	0	0	0
Diodia teres	dite	0.8	3	0	0	0	0	0	0
Elephantopus elatus	elel	0.2	1	0	0	0	0	0	0

Eragrostis hirsuta	erhi	0.1	3	0.6	10	0.9	6	0.1	2
Ageratina aromatica var. aromatica	euar	0.1	2	0	0	0.3	3	0.2	4
Eupatorium capillifolium	euca	3.7	24	3.9	28	2.2	17	0.7	7
Eupatorium mohrii	eumo	0.3	1	0	0	0	0	0	0
Eupatorium rotundifolium	euro	t	1	0	0	0	0	t	1
Euthamia tenuifolia var. tenuifolia	eucr	0	0	0	0	t	1	0.9	4
Fabaceae fam	fab	0.5	4	0	0	0.8	16	0.5	5
Florichia floridana	flfl	0.1	1	0	0	0	0	0	0
Galactia microphylla	gami	0.1	2	0	0	0	0	0	0
Galactia sp.	gas	0.2	5	0	0	t	2	t	1
Galium pilosum	gapi	1.1	6	0.2	5	0.2	6	0.1	3
Pseudognaphalium obtusifolium ssp. obtusifolium	gnob	t	1	0	0	0	0	0	0
Gnaphalium sp.	gns	0.2	7	0.2	9	0.1	5	0.2	5
Gymnopogon ambiguus	gyam	0.5	8	0.2	6	0.5	7	0.9	7
Croptilon divaricatum	hadi	0	0	0	0	0.1	3	0.1	1
Houstonia procumbens	hepr	0	0	0	0	t	1	t	1
Helianthemum corymbosum	heco	t	1	0	0	0	0	t	1
Helianthus floridanus	hefl	0	0	0.1	1	0	0	0	0
Heterotheca subaxillaris	hesu	t	1	0	0	0	0	0	0
Hieracium sp.	his	0.1	2	0	0	0	0	0.1	5
Hypericum gentianoides	hyge	0	0	t	1	0	0	0	0
Ipomoea sp.	ips	t	1	0	0	0	0	0	0
Juncus dichotomus	judi	0	0	0	0	t	1	0	0
Kummerowia striata	kust	0	0	t	2	t	1	0.1	1
Lechea minor	lemi	0.1	3	0.2	3	0	0	0.3	4
Lechea mucronata	lemu	0	0	0.1	2	0	0	0	0
Lechea sp.	les	0.4	7	0.3	8	t	1	0.1	3
Lespedeza hirta	lehi	0.1	1	t	2	0.1	1	0	0
Lespedeza stuevei	lest	2.3	22	1.0	9	0.7	7	1.8	23
Liatris elegans	liel	0.2	5	0.1	2	0.1	1	t	1
Liatris tenuifolia	lite	0.4	4	0	0	0	0	0.1	2
Liatris sp.	lis	t	1	0	0	0	0	0	0
Lobelia puberula	lopu	0	0	0	0	t	2	0	0
Ludwigia sp.	lus	0	0	0	0	t	1	0	0
Mollugo verticillata	move	0.1	2	0	0	0	0	0	0
Opuntia humifusa	ophu	0	0	0.1	2	0.3	6	0	0
Oxalis corniculata	oxco	0.1	3	0.1	4	0	0	0	0
Panicum anceps	paan	0	0	0	0	t	1	0	0
Panicum rigidulum	pari	0	0	0	0	0.2	3	0	0
Panicum verrucosum	pave	0	0	0	0	0.2	3	0	0
Panicum virgatum	pavi	0	0	t	1	0.1	1	0.1	2
Paspalum notatum	pano	1.1	8	0.4	8	0	0	0.1	2
Paspalum setaceum	pase	0	0	t	1	0	0	0	0
Phlox nivalis	phni	0.3	1	0	0	0.2	1	0	0
Piriqueta cistoides ssp. caroliniana	pica	0	0	0.1	2	0	0	t	1

Pityopsis sp.	pis	4.4	29	4.6	39	13.2	52	7.5	44
Poaceae fam	poa	0	0	t	1	0	0	2.0	16
Polygala grandiflora	pogr	t	1	0	0	0	0	0	0
Polypremum procumbens	popr	0.8	10	0.4	5	0.1	1	0.1	2
Pteridium aquilinum	ptaq	1.1	8	0	0	0.5	4	0.4	3
Rhexia mariana	rhma	0	0	0	0	0.1	3	0	0
Rhus copallinum	rhre	0	0	t	1	0	0	0	0
Rhynchosia reniformis	rhto	0.4	4	t	1	0.1	2	t	1
Rhynchosia tomentosa	rufu	0.3	4	0	0	0.2	4	0.5	13
Rudbeckia fulgida	ruca	0	0	0	0	0.1	2	0	0
Ruellia caroliniensis	saal	0	0	0	0	0	0	t	1
Saccharum alopecuroidum	sch	0	0	0	0	0	0	0.3	2
Andropogon ternarius/Schizachyrium scoparium		0.5	7	0.7	10	0.6	5	3.8	29
Schizachyrium scoparium	scsc	4.0	24	3.0	27	7.5	36	14.0	57
Scleria sp.	scs	1.0	12	0.1	5	0.2	4	1.3	10
Seymeria pectinata	sepe	0	0	0	0	0.1	1	0	0
Silphium compositum	sico	0	0	0	0	0.3	2	0.1	1
Solidago fistulosa	sofi	0	0	0	0	0.1	2	0.4	1
Solidago latissimifolia	sola	0	0	0	0	0	0	0.3	2
Solidago nemoralis	sone	2.9	24	0.4	12	1.9	22	3.0	30
Solidago odora	sood	0	0	0	0	0.6	4	0	0
Solidago sp.	sos	0	0	0	0	0.1	2	0	0
Sorghastrum secundum	sose	0	0	0.5	6	0.3	2	0	0
Unknown moss	sph	0	0	1.6	24	0.1	5	0.1	2
Sporobolus junceus	spju	0	0	1.5	9	0	0	0	0
Stylisma patens	stpa	t	1	0.1	2	0	0	0	0
Stylodon carneus	stca	0	0	0	0	0.2	1	0.1	1
Tephrosia florida	tefl	0.2	2	0	0	0	0	0	0
Tephrosia sp.	tes	0	0	0	0	t	1	0	0
Tephrosia virginiana	tevi	t	1	0	0	0	0	0	0
Tragia urens	trur	t	1	t	1	0	0	t	1
Trichostema dichotomum	trdi	0	0	0.5	3	0	0	0	0
Trichostema setaceum	trse	0.3	1	0.3	4	0.1	5	0	0
Tridens carolinianus	trca	0	0	t	1	0	0	0	0
Tridens flavus	trfl	0.2	6	0	0	1.0	11	0	0
16 Unknown herbaceous		t-1	t-2	t	0	t	t-1	t	t
Urtica sp.	urs	0.1	3	0	0	t	1	0	0
Vicia sp.	vis	t	1	0	0	0	0	0	0
Viola palmata	vipa	t	1	0	0	0	0	0	0
Viola primulifolia	vipr	0	0	t	1	t	1	0	0
Wahlenbergia marginata	wama	0	0	0.3	1	0	0	t	1

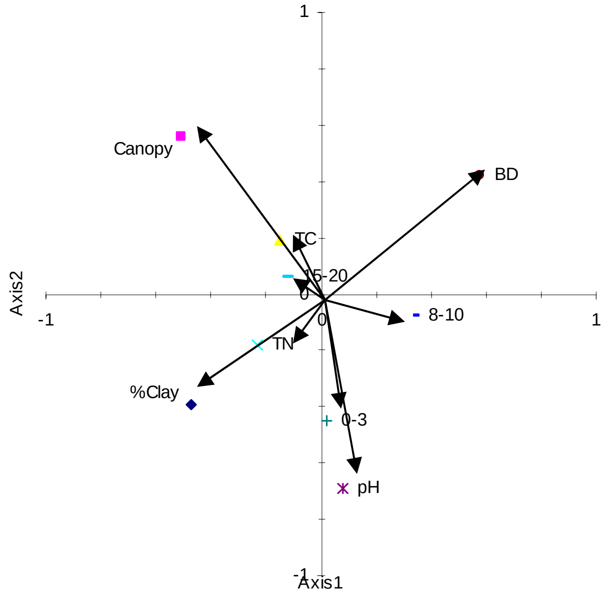
* t = trace = <0.1

Table 8: Results from four different manipulations of the CCA of the herbaceous species. Values in the columns for axes 1, 2, and 3 refer to the eigenvalues for each ordination axis in each analysis. The p-values refer to the Monte Carlo permutation test run on the 1st ordination axis and all of the axes combined.

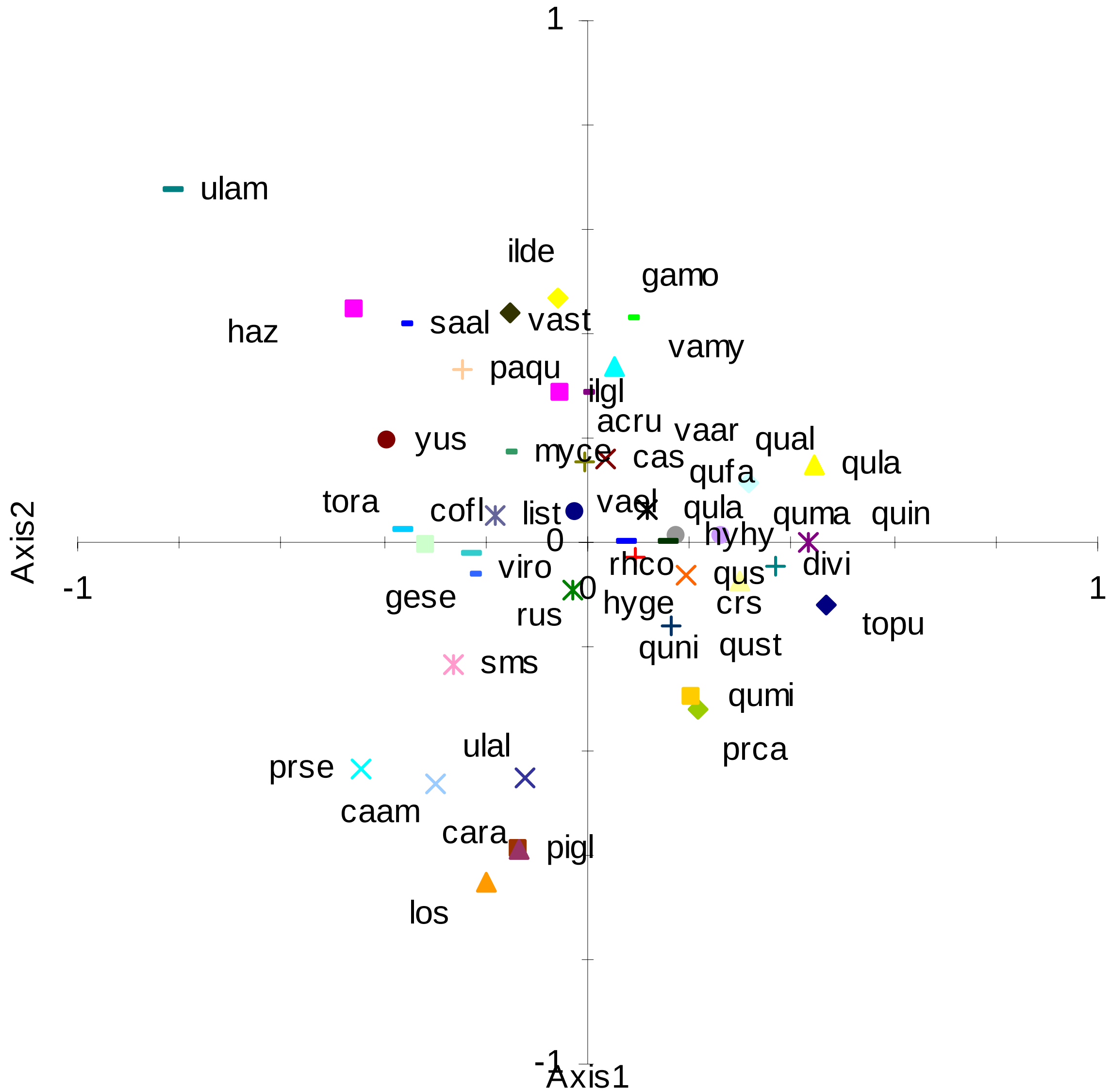
CCA Analysis	Axis 1	Axis 2	Axis 3	p-value 1st	p-value all
All species	0.291	0.220	0.177	0.035	0.045
down-weight	0.259	0.171	0.115	0.010	0.010

Table 9: Weighted correlation matrix for species axes, environmental axes, and environmental variables for herbaceous species, with down-weighting performed by CANOCO.

	SP AX1	SP AX2	SP AX3	ENV AX1	ENV AX2	ENV AX3
SP AX1	1.000					
SP AX2	0.035	1.000				
SP AX3	-0.023	0.060	1.000			
ENV AX1	0.926	0.000	0.000	1.000		
ENV AX2	0.000	0.938	0.000	0.000	1.000	
ENV AX3	0.000	0.000	0.777	0.000	0.000	1.000
BD	0.394	-0.135	0.565	0.425	-0.144	0.727
%Clay	-0.348	-0.063	-0.424	-0.375	-0.067	-0.545
TC	0.091	0.299	-0.222	0.098	0.319	-0.285
TN	0.081	0.244	-0.045	0.087	0.260	-0.058
pH	-0.072	0.313	0.052	-0.078	0.333	0.067
0-3	-0.096	0.695	0.444	-0.104	0.741	0.571
8-10	0.739	-0.037	-0.270	0.798	-0.039	-0.347
15-20	-0.266	-0.034	-0.274	-0.287	-0.037	-0.353
Canopy	-0.405	-0.414	-0.271	-0.438	-0.442	-0.348
Eigenvalue	0.259	0.171	0.115			



1
Canopy
BD
TC
15-20
0
-1
0
1
8-10
TN
%Clay
0-3
pH
-1
Axis1
Axis2



1
-1
0
1
-1
Axis1
Axis2
ulam
ilde
gamo
haz
saal
vast
vamy
paqu
ilgl
acru
vaar
qual
yus
myce
cas
qula
qufa
tora
cofl
list
vael
qula
quma
quin
hyhy
rhco
viro
qus
divi
gese
hyge
crs
rus
topu
quni
qust
sms
qumi
prca
ulal
prse
caam
cara
pigl
los

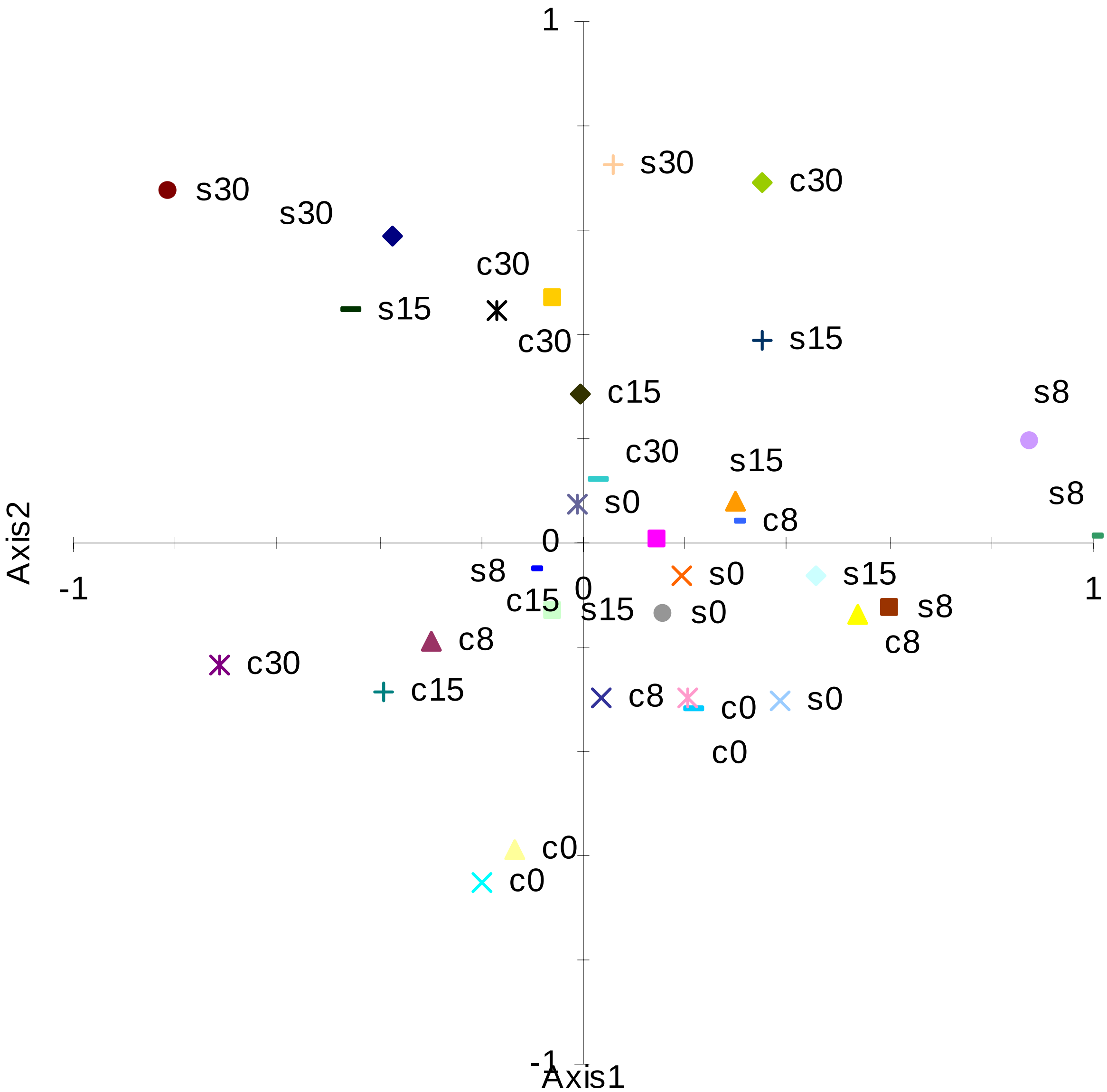
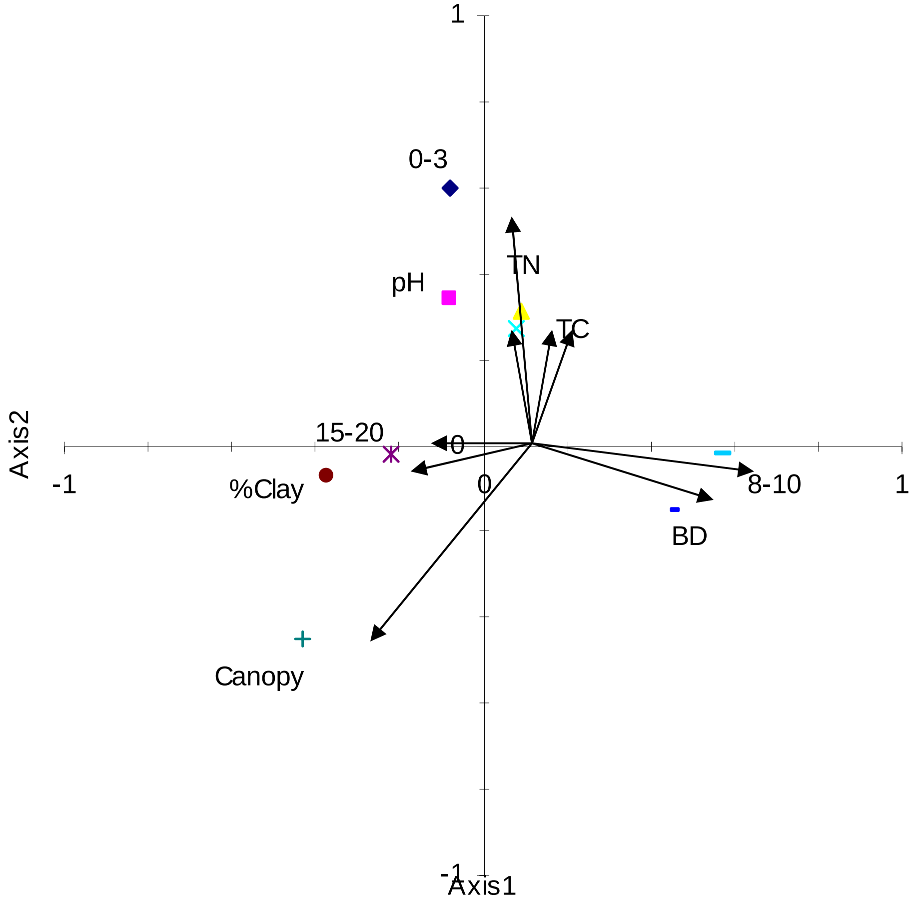
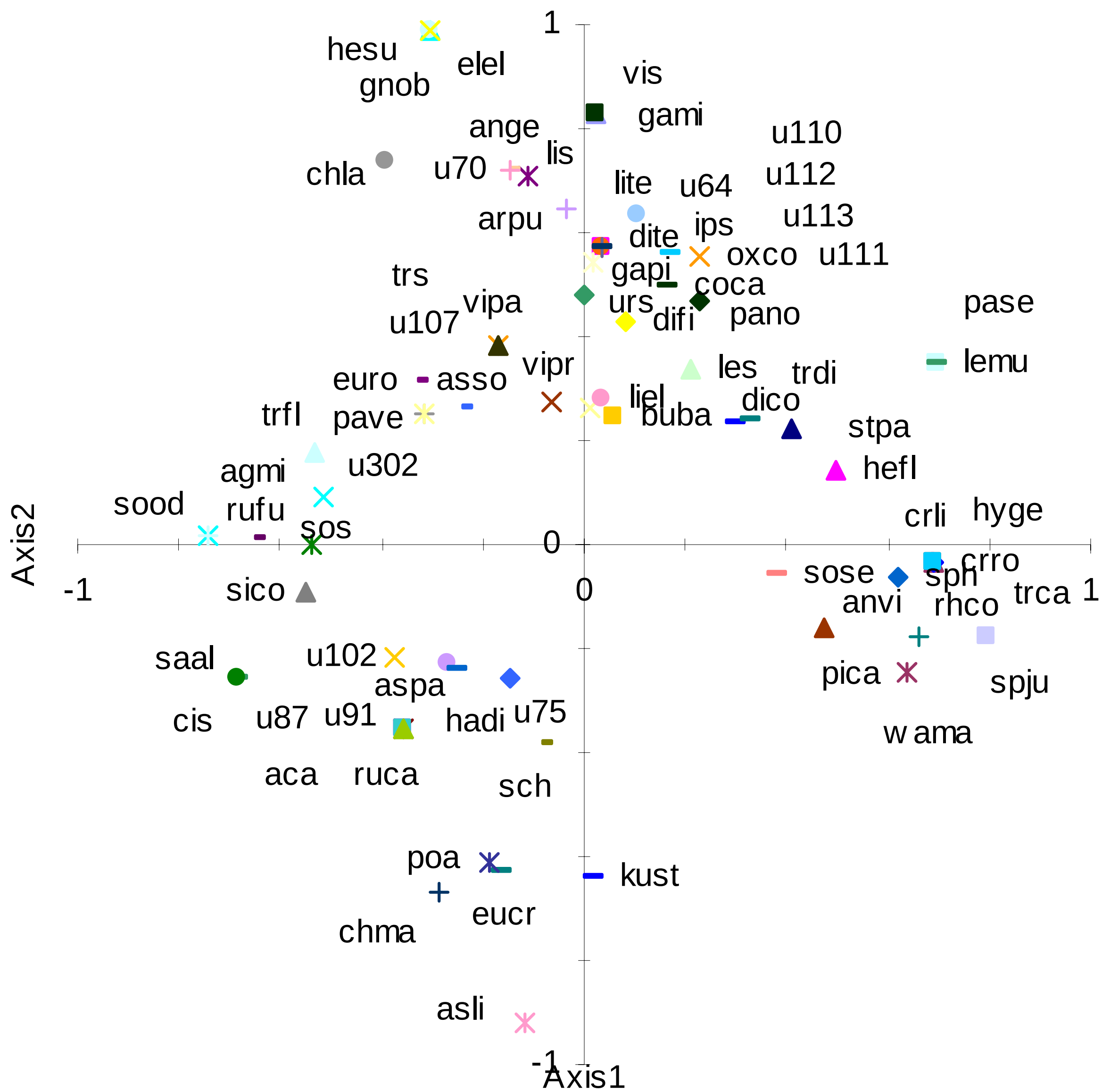


Figure 1.
Woody Species: (a) Canonical correspondence analysis (CCA) ordination plot showing the location, length and direction of edaphic variables. (b) CCA ordination plot showing the species occurrence in relation to edaphic variables. Code names for the species are located in Table 3 A. (c) CCA ordination plot showing the approximate locations of sample sites. Sites are coded with c=loamy, s=sandy, 0=0-3 yr., 8=8-10 yr., 15=15-20 yr. and 30=30-80 yr.


1
0-3
TN
pH
TC
15-20
-10
-1
%Clay
0
8-10
1
BD
Canopy
-1
Axis1
Axis2



Axis2
Axis1
1
0
-1
-1
0
1
hesu
elel
gnob
vis
gami
ange
lis
chla
u70
u110
u112
u113
u111
lite
u64
arpu
ips
dite
oxco
gapi
coca
trs
vipa
urs
difi
pano
pase
u107
lemu
vipr
les
trdi
euro
asso
liel
dico
buba
trfl
pave
stpa
hefl
u302
agmi
rufu
sood
sos
crli
hyge
crro
sph
sose
anvi
rhco
trca
sico
pica
spju
wama
saal
u102
aspa
cis
u87
u91
hadi
u75
aca
ruca
sch
poa
kust
eucr
chma
asli

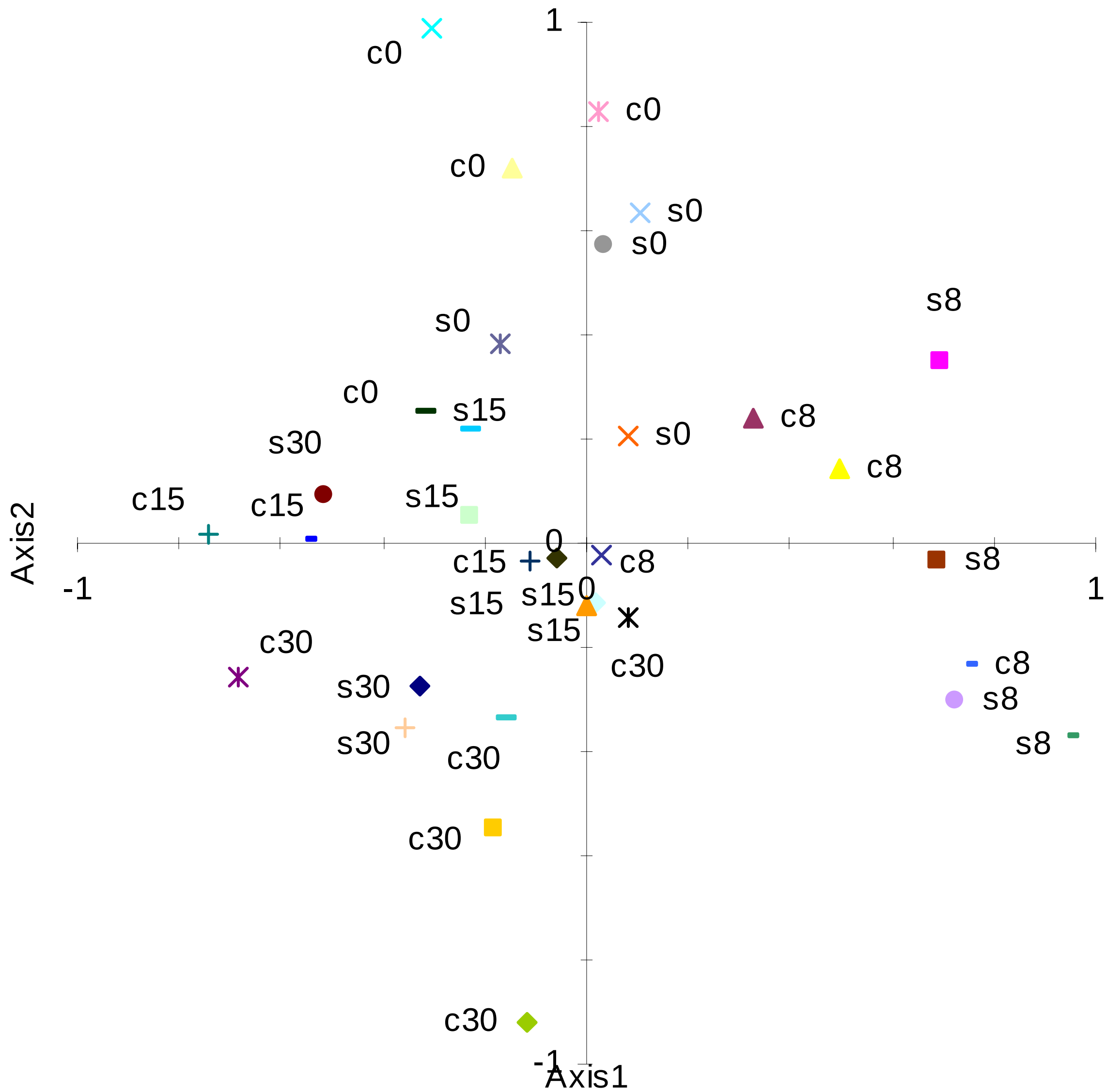


Figure 2.
Herbaceous species: (a) Canonical correspondence analysis (CCA) ordination plot showing the location, length and direction of edaphic variables. (b) CCA ordination plot showing the species occurrence in relation to edaphic variables. Note that some species located near the origin of the plot were left out for ease of viewing. Code names for the species are located in Table 7 (c) CCA ordination plot showing the approximate locations of sample sites. Sites are coded with c=loamy, s=sandy, 0=0-3 yr., 8=8-10 yr., 15=15-20 yr., and 30=30-80 yr.

3.3 Hydrology

Hydrologic indicators are of significant value for analysis of disturbance or recovery on a watershed scale. Information from hydrologic studies that may contribute to an increased understanding of ecological impacts include the following:

- **3.3.2 Correlation and regression analyses were performed to determine relationships among the watershed physical characteristics and the storm-based hydrologic indices.**

- **3.3.3 Analysis of hydrographs clearly reflect hydrologic imbalances resulting from soil and vegetation disturbance in uplands.**

- **3.3.4 Soil physical parameters (bulk density, porosity, texture, grain-size distribution, and saturated hydraulic conductivity) are potentially useful at small spatial scale.**

3.3.1
Characterization and Modeling of Throughfall Temporal Variability for Forest Communities in the Southeastern U.S. Bryant, M.L., S. Bhat, and J.M. Jacobs.

ABSTRACT

The temporal variability of interception losses was characterized for contrasting forest communities in the southeastern United States. Throughfall was measured simultaneously at Fort Benning in western GA for the five forest communities that are characteristic of the region: mature pine, 13 year-old pine plantation, lowland hardwood, upland hardwood, and mixed upland hardwood/pine. The measured interception losses over the study period were 22.3, 18.6, 17.7, 17.6, and 17.4% of the total precipitation in the pine, mixed forest, lowland hardwood, pine plantation, and upland hardwood forests, respectively. The Gash et al. (1995) model, using annual average canopy cover values, predicted interception losses with an agreement of -8.1 to 10.5%. Application of seasonal canopy cover values to the model improved accuracy in all cases with an overall range from -7.3 to 4.5%. The Gash model's assumption of a constant canopy storage capacity was examined for pine and lowland hardwood plots with varying densities and found to have good agreement for mature pine forests having 64, 44, and 29% canopy cover. However, reasonable interception predictions for the riparian wetland forests required corrections for species composition and understory vegetation.

Key Words: Throughfall; Interception; Gash Model; Canopy Cover; Fort Benning

INTRODUCTION

Canopy cover intercepts a significant fraction of precipitation on forests. Such intercepted precipitation evaporates without reaching the soil (Klaassen, 2001). Main factors influencing the amount of precipitation interception are canopy storage capacity and evaporation rate (Loustau et al., 1992). Canopy interception is an important component of the hydrological budgets (Whelen and Anderson, 1996) and nutrient cycles (Loescher et al., 2002) of the terrestrial ecosystems. Interception of precipitation and its subsequent evaporation constitute a considerable net loss to the system under certain conditions (Schellekens et al., 2000). For example, Cantu-Silva and Gonzalez-Rodriguez (2001) reported that in pine and oak forests in northeastern Mexico the total precipitation during the experimental period was 974 mm and estimated interception loss was 19.2, 13.6 and 23% for the pine, oak and pine-oak canopies, respectively. Llorens et al. (1997) reported 15-49% interception by conifers in a Mediterranean mountainous area under different climatic conditions. Studies in tropical and subtropical rainforests showed that the interception losses varied from 6-42% (e.g., Asdak et al., 1998; Hutjes et al., 1990; Sinun et al., 1992; Scatena 1990).

For patchy landscapes, the distribution of these forest communities can influence the spatial variability of interception losses at scales larger than the characteristic patch size (Crockford and Richardson, 2000). Teklehaimanot and Jarvis (1991) examined within community variations of tree density and their effects on canopy storage capacity for a Sitka spruce plantation. They concluded that canopy storage capacity is a property of individual trees and is unaffected by tree density. This suggests that canopy storage capacity is consistent among tree species and spatial variations of interception losses for a single community are a function of canopy cover only.

Rutter et al. (1971) developed a physically based numerical, point interception model that has been the basis of numerous subsequent analytical or semi-analytical models (Zeng et al., 2000). Gash (1979) introduced a variation of the Rutter model that demonstrated the evaporation of precipitation intercepted by forest canopies can be estimated from the forest structure, the mean precipitation and evaporation rates, and the precipitation pattern (Gash et al., 1995). The Rutter and Gash models are extremely sensitive to the formulation of evaporation (Lankreijer et al., 1993). In both models, evaporation is calculated per unit area of ground. This formulation is problematic for sparsely vegetated forests. Valente et al. (1997) reported that original Gash model (Gash, 1979) overestimated interception losses by as much as 44% in a sparse pine forest in Portugal. Various studies (e.g., van Dijk and Bruinjnzeel, 2001b; Jackson, 2000; Carlyle-Moses and Price, 1999; Návar et al., 1999; Valente et al., 1997; Dykes, 1997) reported that the revised Gash model (Gash et al., 1995), that calculates evaporation per unit area of canopy rather than per unit area of ground, greatly improved the accuracy of the interception loss predictions in sparse forests.

Numerous experimental studies have identified measured parameters that are required for interception models such as the Gash and Rutter models (see reviews by Liu, 2001; Llorens and Gallart, 2000). The critical forest parameters and data necessary to simulate interception losses are fairly consistent among models and include canopy storage capacity, forest cover (one minus gap fraction), total precipitation, average precipitation intensity, and average evaporation during the canopy wetting and drying cycles (Zeng et al., 2000; Liu, 1997; Gash et al., 1995; Gash, 1979). While most research considers a single forest plot, several studies have reported parameters for forests with spatial variations in cover or density (e.g., Huber and Iroume, 2001; Jackson, 2000; Llorens and Gallart, 2000). Few studies simultaneously characterized parameters for more than one forest (e.g., Huber and Iroume, 2001; Tobón Marin et al., 2000). As parameters (e.g., canopy and trunk storage capacity) can vary depending on the analysis approach (Lankreijer et al., 1999; Klaassen et al., 1998; Bunnell and Vales, 1990), the transferability of canopy parameters determined during different experiments may be of limited value for site comparisons and modeling applications.

The goal of this research is to characterize the canopy specific parameters, climatic variables, and interception components by forest communities as necessary to determine the relative net water input across those forest communities distinctive of the southeastern United States. In support of this effort, the effects of seasonal variability of canopy cover and landscape heterogeneity on interception losses are examined. The revised Gash model (1995), referred to as the revised model hereafter, was chosen for this study since it has been validated in many different landscapes and allows the explicit parameterization of canopy cover in its formulation. The objectives of this study are to: 1) determine the parameters necessary to apply the revised model for use in five forest biomes, 2) use the model to predict precipitation interception and compare the measured and modeled results across forest types and canopy densities, 3) explore the influence of seasonal changes in canopy characteristics on interception losses, and 4) consider the relationship between interception predictions for individual forest communities to watershed scale aggregated values at a range of temporal resolutions.

STUDY AREA

The study was conducted at the Fort Benning military reservation, located in southwest Georgia. Long, hot summers and mild winters characterize the region's climate. Average annual precipitation is about 830 mm with a monthly average of 69 mm. 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. The soils in the area are dominated by loamy sand with some sandy loam.

Two second-order watersheds, Bonham-1 and Bonham-2, were selected for this study. These watersheds were selected because they represent most of the region's predominant forest types. The Bonham-1 watershed has an area of 0.76 km^2, a minimum elevation of 87.8 m and maximum elevation of 144.2 m, and an average slope of 8.42%. The Bonham-2 watershed has an area of 2.21 km^2, a minimum elevation of 90.5 m, a maximum elevation of 159.1 m, and an average slope of 8.04%.

Patchy land cover, formed from a mosaic of open or forested areas, characterizes the two watersheds (Fig. 1a). The open areas, comprising 6.8% of the total areas may be further categorized as military use, brush, or wildlife openings. The military openings are non-forest 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 *Crateagus*. The wildlife openings are openings in the forests that are maintained in grass and forbs. The forested areas include five forest types – mature pine, pine plantation, lowland hardwoods, upland hardwood, and mixed hardwood/pine. The mature pine, which consists of loblolly (*Pinus taeda*) and shortleaf pine (*Pinus echinata*), is the dominant forest type of each watershed. The pine plantation is a 13-year old longleaf pine (*Pinus palustris*) stand planted in rows. The lowland hardwoods dominate the riparian wetland corridor and hereafter are referred to as wetland forests. They consist mostly of various hardwood trees along with a range of wetland understory vegetation. The upland hardwood stands (hereafter hardwood) are generally mature scrub oak (*Quercus berberidifolia*). White oak (*Quercus alba*), shortleaf pine, and loblolly pine are the dominant species in the mixed pine/hardwood stands. The five forest communities considered in this study covered approximately 93% of the watersheds (Table 1). The forests are managed for the endangered red-cockaded woodpecker (*Picoides borealis*) using prescribed burning on a three-year cycle. A majority of the land area included in this study was burned within one year of the study period.

METHODS AND INSTRUMENTATION

The study was conducted from April 4, 2001 through June 11, 2002. Canopy parameters, climatic variables, and interception components were measured throughout this period. The measured data include precipitation, throughfall, stemflow, atmospheric conditions, and canopy cover. The following sections describe the instrumentation and methods applied to this study.

Study Plots

A rectangular plot was established in each forest community. The plots were randomly selected within areas having vegetation that is consistent with the average vegetation density and distribution for the respective forest community (Fig. 1b). The dimensions of each plot ranged from 10 x 40 m in wetland area in the riparian corridor to 30 x 30 m in upland areas. The plot in the riparian corridor was designated as Wetland (WET). Similarly, the plots in the upland area were designated as Pine (PIN), Pine Plantation (PIP), Hardwood (HRD), and Mixed (MXD). Each plot was subdivided into four sampling grids of equal size. Although these sampling grids consist of similar tree species, there are differences in number of trees per grid, percent of total number by tree types, average tree height and diameter, and trees per hectare. Each grid was outfitted with four throughfall collectors and one tipping bucket rain gauge for a total of 20 sampling points per plot. The throughfall collectors and tipping buckets were randomly placed on the ground within the confines of the grid. As the average separation distance necessary to ensure independent measurements likely would have extended plots beyond a single forest type and density (Loescher et al., 2002), each instrument was relocated randomly within the grid after one set of data were collected to reduce the standard error of estimation (Lloyd and Marques, 1988). The throughfall data were collected using a 203.2 mm diameter tipping bucket rain gauges (model RG-100a, RainWise®) and 152.4 mm diameter throughfall collectors on a bi-weekly basis. The tipping bucket measurements were aggregated to the same biweekly periods as the throughfall collectors.

To test the hypothesis that canopy storage capacity is consistent among tree species and that temporal variations of interception losses may be characterized by canopy cover, additional wetland and mature pine plots were established with varying canopy density. Two additional plots for both the wetland and mature pine communities were monitored from February 1 to April 29, 2002 (Fig. 1b). The additional plots, referred to as Wetland B (WET_B), Wetland C (WET_C), Pine B (PIN_B), and Pine C (PIN_C), were selected based on a visual inspection and a spherical densiometer survey. The instrumentation, methods, and data collection used for these plots were identical to those described above.

Stemflow was measured in the wetland, pine, pine plantation, and hardwood plots. For these plots, the dominant tree species were further sub-divided into three classes of diameter at breast height (DBH) and projected crown radius. Stemflow gauges were installed in each plot such that the dominant tree species were sampled and that a representative tree from each DBH and projected crown radius category was included. The mixed plot stemflow was determined by averaging the pine and hardwood measurements.

Canopy Cover

Canopy cover was determined by direct measurement with a Model-A spherical densiometer using the method outlined by Lemmon (1956). This method provides a simple and reliable measure of canopy cover (Englund et al., 2000; Bunell and Vales, 1990). Readings were taken in the center of each study plot on a bi-weekly basis throughout the study period. The measurements were taken in each cardinal direction and averaged to estimate the plot mean.

Climate Data

Precipitation data were measured by two tipping bucket rain gauges (model RG-100a, RainWise®). The tipping buckets were located in a 50 x 150 m clearing on the boundary between the Bonham-1 and Bonham-2 watersheds. The distance from the precipitation gauges to the throughfall plots ranged from 315 to 900 m. An additional gauge was deployed at the northern boundary of the Bonham-1 watershed for a subperiod of the study. t-tests showed no significant differences ($p<0.05$) among storm event totals measured by the three gages. Storm totals were calculated as the average recording of the two rain gauges deployed for the entire study period. Data were collected from the instruments on a bi-weekly basis.

The Ecosystem Characterization and Monitoring Initiative (ECMI) meteorological monitoring station at McKenna Mout within the military installation continuously records the atmospheric data. The station is located at approximately 7 km south of the study watersheds. The station's data include air temperature, relative humidity, solar radiation, wind speed, and precipitation. Data were collected in 1-minute intervals and averaged over 30-minute periods. Wind speed was corrected for roughness differences between the measurement site and study plots using a power law scaling wind function.

MODELING APPROACH

The revised model uses a canopy water balance approach. Precipitation reaching the canopy either evaporates, runs down the trunk, or falls to the ground as canopy drip. The model considers each precipitation event as an individual event with enough time between events to allow the trunk and canopy to completely dry. The total interception loss is the summation of the interception losses from a series of individual events over a period of time. Each precipitation event consists of a wetting up period, a saturation period, and a drying out period. During each event, intercepted precipitation is lost through evaporation. The revised model assumes the amount of precipitation lost due to evaporation is a function of the unit area of canopy. This approach requires the determination of specific canopy and trunk parameters and the measurement of several atmospheric parameters. The necessary canopy parameters include the canopy cover expressed as a percent of canopy per unit area, the canopy storage capacity, and the trunk storage capacity. The required atmospheric parameters are the gross precipitation per storm event, the average evaporation rate from a saturated canopy, and the mean precipitation rate per storm event.

Interception losses in the revised model are calculated using the following equation:

$$\sum_{j=1}^{n+m} I_j = c\sum_{j=1}^{m} P_{G\,j} + \left(c\overline{E}/\overline{R}\right)\sum_{j=1}^{n}\left(P_{G\,j} - P'_G\right) + c\sum_{j=1}^{n} P'_{G\,j} + qS_t + p_t\sum_{j=1}^{n-q} P_{G\,j} \qquad (1)$$

where I is the interception loss, n is the number of saturation events, m is the number of non-saturation events, c is the mean canopy cover, P_G is the total precipitation during the event, $\overline{E}$ is the mean evaporation rate from a saturated canopy scaled in proportion to canopy cover, $\overline{R}$ is the mean precipitation rate, P'_G is the amount of precipitation necessary to fill the canopy storage capacity, q is the number of events that saturate the trunk storage capacity, S_t is the trunk storage capacity, and p_t is the incident precipitation reaching the trunks.

The experimental data were used to determine the canopy specific parameters, climatic variables, and interception components. The total precipitation during the event is the average of the two tipping bucket rain gauges. An event is any period where precipitation was recorded without a break of more than three hours between successive half hour recordings. The mean precipitation rate is the total precipitation during the event divided by the storm duration. Average annual canopy cover is calculated by averaging all measured values during the experiment. The mean evaporation rate from a saturated canopy scaled in proportion to canopy cover is equal to the mean evaporation rate multiplied by the canopy cover. The mean evaporation rate $\overline{E}$ was determined with the Penman-Monteith equation with the surface resistance set to zero. The hourly average atmospheric measurements taken from the meteorological monitoring station were used to calculate evaporation. Net radiation was determined from the measured solar radiation using by using an albedo of 0.15 and an emissivity of 0.96 for the forest communities (Tables 6.4 and 6.5 in Brutsaert, 1982). Wind speed was corrected for difference between the measured surface roughness and the forest surface roughness using a power law approximation and an estimated surface roughness of 1 cm and 50 cm for the short grass and forest canopy, respectively.

The amount of precipitation necessary to fill the canopy storage capacity P'_G is given by the following equation (Carlyle-Moses and Price, 1999):

$$P'_G = -\left(\overline{R}/E_c\right)S_c * \ln\left[1-\left(E_c/\overline{R}\right)\right] \qquad (2)$$

where $\overline{R}$ is the mean precipitation rate falling on a saturated canopy, E_c is the mean evaporation rate scaled in proportion to canopy cover where $E_c = \overline{E} * c$, and S_c is the canopy storage capacity per unit area of cover where $S_c = S / c$. The canopy storage capacity S was determined from the negative regression line intercept of the precipitation versus throughfall plot for saturation events. Half-hourly precipitation data were used to extract the storm events. Each half-hourly precipitation was accumulated until the precipitation stopped. An event was defined for the time frame within which 2.8 mm or more precipitation was accumulated and the same events were used to determine the canopy storage capacity. The trunk storage capacity and the incident precipitation reaching the trunks were determined by the method used by Gash and Morton (1978) and Carlyle-Moses and Price (1999) where S_t is the slope and p_t is the regression line intercept of the stemflow versus incident precipitation graph.

The revised model was applied to the additional wetland plots using physically based corrections to adjust the canopy cover and canopy storage capacity for wetland community composition differences among plots. Plot statistics were used to determine the percentage of total canopy area contributed by the overstory for WET. The canopy cover contributed by the overstory vegetation in WET may be described as

$$C_{Wadj} = A_O / A_T * C_W \qquad (3)$$

where C_{Wadj} is the adjusted canopy cover, A_o is the projected canopy area for all trees greater than 12 m tall, A_T is the projected canopy area for all trees, and C_W is canopy cover.

The canopy storage capacity for WET was adjusted to account for the difference in species composition among plots. Weighted averaging was used to account for the difference in the pine contribution. The adjusted canopy storage capacity S_{adj} was determined by

$$S_{adj} = (S_W * R_B + S_P * R_P) / C_{Wadj} \qquad (4)$$

where S_W and S_P are the canopy storage capacities for the WET and PIN, respectively, R_p is the difference in percent pine composition between the WET and WET_B or WET_C, R_B (where $R_B = 1 – R_P$) is that portion of plots WET_B or WET_C sharing the same species composition as WET.

RESULTS

Climate

During the study period, 140 discrete storm events generated 752.8 mm of precipitation. The events ranged in intensity from 0.3 to 14.4 mm hr^{-1} with an average intensity of 1.8 mm hr^{-1}. Total precipitation accumulation for each event ranged from 0.3 to 73.2 mm with an average of 5.4 mm. Approximately 46% of all storms deposited less than 1 mm. The duration of each event ranged from 0.5 to 34 hours with 50% of all events being one hour or less. The analysis of precipitation data collected from the Bonham-1, Bonham-2 and McKenna Mout meteorological station showed no systematic spatial trend. The rainfall rate and evaporation rates were found to vary seasonally (Fig. 2). The relative rate E/R is observed to exceed the annual average rate by 34% in the summer and to fall below that average by up to 42% during the remainder of the year.

Canopy Cover

The wetland, mixed, and hardwood plots exhibit a distinct seasonal variation in canopy cover (Fig. 2). The hardwood plot experiences the most pronounced seasonal variation. Its canopy cover decreases from an average of 60% during the spring and summer to 37% during the winter. The wetland canopy cover drops from an average of

93% during the spring and summer (typically mid April to late November) to 75% during the winter (typically late November to mid April). The mixed plot experiences a smaller decrease, from 77% to 70%, during the same time period. Neither the mature pines nor the pine plantation showed distinct seasonal variations in canopy cover. The pine plantation canopy cover increased over the study period due to tree growth.

Throughfall

The forest stands exhibited a range of measured throughfall and derived interception values (Table 2). The total throughfall measurements ranged from 553.8 mm in the mixed plot to 614.5 mm in the wetland plot. Over the study period, sampling variability was relatively low with coefficients of variation (CV) ranging from 0.11 in the hardwood plots to 0.17 in the wetland plot. Throughfall plus stemflow accounted for 77.7 to 82.5% of incident precipitation for mature pine and hardwood forests, respectively. Interception losses were largest in the mature pine forest (22.3%) and smallest in the hardwood forest (17.4%). A comparison of average canopy cover and actual interception losses in different forest stands showed that interception losses were very consistent, within 2%, for all forest communities except the mature pine. The pine losses were approximately 5% greater than the other communities. The relative ratio of throughfall to precipitation was observed to vary seasonally with the largest ratio observed in the fall and winter (Fig. 3).

Stemflow

Stemflow was measured for individual rainfall events and problems with the instrumentation preclude the use of this data to quantify the total amount of stemflow over the study period. Approximately 12% of the total stemflow data were excluded from the analysis. However, the collected data, ranging from 12 to 29 storms per plot, were sufficient to develop a linear regression model to predict stemflow on an event basis. The cumulative calculated stemflow during the study period ranged from 3.7 mm for the mixed plot to 14.2 mm for the pine plantation plot (Table 2). The percent of incident precipitation for calculated stemflow ranged from 0.54% for the pine, hardwood, and mixed plots to 1.96% for the pine plantation plots.

Revised Model Parameters and Results

Revised model parameters derived from the experimental measurements, canopy parameters, climatic variables, and interception components are summarized by forest community (Table 3). Despite the similarity in the interception percentage, the canopy parameters exhibit a significant range of variability. The precipitation required to saturate the canopy was determined using (2). The P'_G values ranged from 1.14 mm for the wetland plot to 4.00 mm for the pine plantation plot. The canopy storage capacity was determined for each plot by scaling the precipitation versus throughfall regression line intercept by the annual average canopy cover. The canopy storage capacity values for the sparse canopies ranged from 0.98 mm for the wetland plot to 1.97 mm for the mature pine plot.

The revised model was applied using data for the period April 19, 2001 through April 29, 2002 for the wetlands, mature pine, pine plantation, and hardwood plots. The mixed plot was modeled for the period June 12, 2001 through April 29, 2002. During this period, the relative number of saturation precipitation events to non-saturation events ranged between 0.91 for the wetland plot and 2.0 for the pine plantation. This variation results from differences in P_G' values by community.

Two approaches for estimating canopy cover, a single annual average value and a seasonally distributed values, were modeled. Table 4 summarizes the model results for both approaches. For the average annual canopy cover values, the revised model performed well with little error and relatively high Nash-Sutcliffe model efficiency for the mature pine, wetland, and mixed plots. The model overestimated interception by 8.1% in the pine plantation plot. This error is likely due to significant growth of the planted pines during the study period resulting in changes to the tree stem, branch and needle distribution and size as well as undergrowth changes. The largest modeling error occurred for the hardwood plot with an annual difference between the modeled and the measured interception of approximately 13.3 mm. The use of seasonal canopy cover values improves the revised model interception predictions for all communities (Table 4). The improvement is most evident for the hardwood forest where predicted interception increases by over 10 mm, the error decreases from 10.5 to 2.4%, and the Nash-Sutcliffe efficiency increases from 73% to 80%.

Canopy Density Intercomparison

The canopy cover averaged 64, 46, and 29% for the PIN, PIN_B, and PIN_C, respectively. The canopy cover averaged 88, 78, and 66% for WET, WET_B, and WET_C, respectively. The canopy density comparison experiment was conducted from February 1 to April 29, 2002. During this period, 24 individual storm events generated 243.0 mm of precipitation. The event intensities were comparable to the yearlong study while the total precipitation accumulation for each event covered the entire range observed during the year. The net values of throughfall, stemflow, and interception loss are summarized in Table 5. The ratios of measured interception to modeled interception were 0.99, 1.10, and 0.98 for mature pine forests having 64, 44, and 29% canopy cover, respectively. Application of (3) and (4) with R_B and R_P of 15 and 85%, respectively, results in a C_{Wadj} of 66% for WET and an adjusted wetland canopy storage capacity of 1.50 mm for plots WET_B and WET_C. This canopy storage capacity is applicable for the wetland communities with sparse understories. Application of the revised model using the adjusted canopy storage capacity and the seasonal canopy cover significantly reduced the error between the measured and the predicted interception to 6.3 and 8.2% for plots WET_B and WET_C, respectively.

WATERSHED SCALE APPLICATION

The revised model was applied at a watershed scale by first assuming a constant canopy cover value and second by capturing the seasonal dynamics of leaf fall and growth with the canopy cover. Both approaches were used to calculate seasonal and annual interception for each forest community. Land use maps were used to determine watershed seasonal and annual interception values by an area-weighting method. Table 6 lists the modeled interception results using both approaches as well as results for the individual forest communities from April 29, 2001 to April 29, 2002. Recorded precipitation was 835.6 mm during this period. Using the first approach, the area weighted average by forest type with the annual average canopy cover, the revised model predicts the net watershed interception to be 148.4 mm or 18% of gross precipitation. The second approach, using a seasonal canopy cover, the predicted interception is 155.8 mm or 19% of gross precipitation. There is less than 8 mm difference between methods for the watershed scale prediction of annual interception and a maximum of 6.7 mm difference on a seasonal basis.

DISCUSSION

Overall, the observed interception values in this study compare well with published results (e.g., Liu, 2001; Crockford and Richardson, 2000). The pine losses (22.3%), the relatively moderate as compared to previous studies, reflect the fairly short duration, low intensity precipitation events under moderately dry conditions that are characteristic of the study region. The current study supports earlier findings that pine forests have larger interception losses than many other forest biomes. The within plot sampling coefficient of variation, ranging from a study average of 0.11 in the hardwood plot to 0.15 in the wetland plot, was comparable to the 13.6% value found by Carlyle-Moses et al. (2004) for two red oak stands. Assuming the point fluxes follow the normal distribution observed by Carlyle-Moses et al. (2004), the 95% confidence intervals this experiment's 20 gauges ranged from ±6.4 to ±10.6% of the mean throughfall for hardwood and wetland communities, respectively. The results of this study show that sampling requirements vary by community and lowland hardwoods require the greatest number of gages to achieve acceptable errors.

The agreement of the stemflow values with published values (e.g., Liu, 1998; Valente et al., 1997; Hanchi and Rapp, 1997) provides validation for the parameter values determined from the regression line i.e. trunk storage capacity and incident precipitation reaching the trunks. The physical characteristics of the young longleaf pine species, slightly higher branch angle and improved flow pathways, may have caused the slightly higher stemflow values. Overall, stemflow provides only relatively small net water input for the study's forests.

The canopy storage capacity parameters are in good agreement with the available literature. The mature pine, pine plantation, mixed and hardwood values fall within the range reported in reviews by van Dijk and Bruinjnzeel (2001b). Little experimental data exist for wetland forests. However, Liu's (1998) 0.94 mm canopy storage capacity for a cypress wetland in Florida compares favorably with this study's 0.98 mm. The two largest storage capacities were found for pine and the planted pine forests. The storage capacity for the mixed forest (1.58 mm) was intermediate between the pine and the oak hardwood (1.40 mm) storage capacities. The wetland forest's low canopy storage capacity suggests a very low storage capacity for the predominant tree species, sweet gum, but precise determination for the species is not possible as the stand includes a variety of tree species.

The ability of the revised model to provide good predictions under varying canopy cover conditions is important as numerous biomes experience dynamic plant growth and seasonal variations in canopy density (van Dijk and Bruijnzeel, 2001a). Dolman (1987) found a distinction between the measured winter and summer throughfall for an oak forest. During the two years when pests mostly destroyed the oak canopy, the author found that interception was considerably lower (20% and 13%) than the measured other periods (34%, 44%, and 23%). Huber and Iroumé (2001) found a seasonal dependence on interception losses in broadleaved forests. In a tropical forest, Jackson (2000) found that the revised model was strongly dependent on the extent of canopy cover variations caused by pruning. In the current study, the revised model performed well for all forest types with measured interception to modeled interception ratios of 0.95 in wetlands, 0.99 in pine, 1.07 in pine plantation, 0.98 in hardwood, and 1.05 in mixed forests when seasonal canopy cover values were used. These values compare favorably with earlier studies reviewed by Aboal et al. (1999) including Gash et al.'s (1980) results for pine that range from 0.81 to 1.18. While the model results of this study suggest that significant improvements are only found under highly variable canopy cover, the inclusion of additional canopy cover information, to some extent, improved interception for all forests types. As such, the current results support van Dijk and Bruijnzeel's (2001b) theory and resultant model that integrate a continuous canopy cover function with the revised model.

In addition to canopy cover, considered in this study, the revised model results are sensitive to other parameters. To identify the relative importance of the model parameters, a sensitivity analysis was conducted. Three model parameters, canopy cover, the ratio of evaporation to rainfall intensity (E/R), and the canopy storage capacity, were varied from values defined in Table 3 by fixed percentages up to +/- 40%. Typical results are shown in Figures 4 and 5 for the pine and hardwood plots, respectively. The results indicate that the relative impact of canopy cover and canopy storage capacity on interception was approximately 10 times that for E/R for comparable parameter variations. Although the relative response is small, the E/R variability exceeds that of canopy cover in fall and spring (Fig. 2) and inclusion of this variability may improve future modeling efforts.

Zeng et al. (2000) also considered the role of climate parameters. Using an analytical model of interception, they found that the three time scales controlling interception are the mean storm inter-arrival rate, the mean storm duration and the time to evaporate the canopy. For the current study, the mean storm inter-arrival rate and the mean storm duration were 64.5 hrs and 2.65 hrs, respectively, with an average storm intensity of 1.8 mm h^{-1}. The time to evaporate the canopy, a function of the canopy water holding capacity and the wet canopy evaporation (0.1 mm h^{-1}), varied from 7.3 hr for the hardwood to 12.1 hrs for the pine communities. The observed time scales indicate that typical storms at the study site will quickly fill the canopy and that the canopy has sufficient time to dry out between storms as assumed in the revised model. These results, coupled with study's finding that the revised model is a reasonable approach, support the application of critical times scales for apriori determination of the revised model's applicability.

Variations in tree species and understory composition among forest communities may also have a significant impact on model parameters and subsequent interception prediction. Successful application of the revised model to wetlands with a range of densities required additional characterization of understory and species composition. The model corrections to reduce the importance of understory are supported by Calder's (1996) finding that the primary impact of raindrops is the wetting of the upper portion of the canopy. The methods introduced here to correct canopy cover measurements and canopy storage capacity provide a preliminary approach to characterize canopy specific parameters on the basis of site characteristics. While the applied methods draw from a physically-based approach, the corrections were based on a limited dataset and require additional study.

Spatial variation of interception thus needs to reflect variations within a community as well as differences among forest communities. For watersheds in the southeastern region that may be characterized as a mosaic of forest communities, this study shows that pine forests play a distinct role in interception. The pine communities mediate seasonal canopy cover variations and corresponding seasonal differences in the throughfall to precipitation ratio. Thus, the application of average annual canopy cover is more appropriate for watersheds having a large proportion of pine forests. Pine forests, having high canopy storage capacity and a relatively high canopy cover, also consistently intercept larger portions of precipitation as compared to other communities in the landscape. For example, in the study site, application of watershed average annual interception values to individual forest communities would underpredict interception by 16% in the pine forests and overpredict interception in the hardwood forests and riparian wetland forests by 10%. Consideration of patch size is recommended when applying interception results to individual communities.

CONCLUSION

This study derived a set of parameters, coefficients, and physical properties for wetland, mature pine, a 13-year old pine plantation, mixed, and hardwood forest parcels that are appropriate for studying diverse forested communities. The results show that the interception losses are greatest from the mature pine forest, but similar across the other forest biomes. Application of seasonal canopy cover values in lieu of annual average values improved the agreement of the modeled and the actual interception losses for all five forests included in this study. However, significant improvements were found only under the highly variable canopy cover in the hardwood forest. Furthermore, the model was found to be transferable to comparable forest communities of varying canopy cover as long as the canopy cover is adjusted for the area of interest. A new approach was proposed to correct derived parameters for site-specific vegetation in riparian wetlands.

The results demonstrate that attempts to disaggregate a single interception value determined at a relatively large scale should consider the mosaic of forest communities that exist at smaller scales. In particular, observed dissimilarities between mature pine forests and deciduous communities result in differences in the magnitude and timing of water balance components between these communities. To capture the variability of water dynamics for a study region, models that predict interception losses should consider the patch scale of landscape communities and the seasonal canopy changes when formulating the appropriate landscape aggregation scheme.

REFERENCES

Aboal, J. R., Morales, D., Hernández, M., Jiménez, M. S., 1999. The measurement and modeling of the variation of stemflow in a laurel forest in Tenerife, Canary Islands. J. Hydrol. 221, 161-175.

Asdak, C., Jarvis, P. G., van Gardingen, P., Fraser, A., 1998. Rainfall interception loss in unlogged and logged forest areas of Central Kalimantan, Indonesia. J. Hydrol. 206, 237-244.

Brutsaert, W., 1982. Evaporation Into the Atmosphere: theory, history, and applications, D. Reidel, Hingham, MA.

Bunnell, F. L., Vales, D. J., 1990. Comparison of methods for estimating forest overstory cover: differences among techniques. Can. J. For. Res. 20, 101- 107.

Cantu Silva, I., Gonzalez Rodríguez, H., 2001. Interception loss, throughfall and stemflow chemistry in pine and oak forests in northeastern Mexico. Tree Physiol. 21, 1009–1013

Calder, I. R., 1996. Dependence of rainfall interception on drop size: 1. Development of the two-layer stochastic model. J. Hydrol. 185, 363-378.

Carlyle-Moses, D. E., Price, A. G., 1999. An evaluation of the Gash interception model in a northern hardwood stand. J. Hydrol. 214, 103-110.

Carlyle-Moses, D. E., Flores Laureano, J.S., Price, A. G., 2004. Throughfall and throughfall spatial variability in Madrean oak forest communities of northeastern Mexico. J. Hydrol. (in Press).

Crockford, R. H., Richardson, D. P., 2000. Partitioning of rainfall into throughfall, stemflow and interception: effect of forest type, ground cover and climate. Hydrol. Processes 14, 2903-2920.

Dolman, A. J., 1987. Summer and winter rainfall interception in an oak forest. Predictions with an analytical and a numerical simulation model. J. Hydrol. 90, 1-9.

Dykes, A. P., 1997. Rainfall interception from a lowland tropical rainforest in Brunei. J. Hydrol. 200, 260-279.

Englund, S. R., O'Brien, J. J., Clark, D. B., 2000. Evaluation of digital and film hemispherical photography and spherical densiometry for measuring forest light environments. Can. J. For. Res. 30, 1999-2005.

Gash, J. H. C., 1979. An analytical model of rainfall interception by forests. Q. J. R. Meteorol. Soc. 105, 43-55.

Gash, J. H. C., Lloyd, C. R., Lachaud, G., 1995. Estimating sparse forest rainfall interception with an analytical model. J. Hydrol. 170, 79-86.

Gash, J. H. C., Morton, A. J., 1978. An application of the Rutter model to the estimation of the interception loss from Thetford forest. J. Hydrol. 38, 49-58.

Gash, J.H.C., Wright, I.R., Lloyd, C.R., 1980. Comparative estimates of interception loss from three coniferous forests in Great Britian. J. Hydrol. 48, 89-105.

Hanchi, A., Rapp, M., 1997. Stemflow determination in forest stands. For. Ecol. Manage. 97, 231-235.

Huber, A., Iroumé, A., 2001. Variability of annual rainfall partitioning for different sites and forest covers in Chile. J. Hydrol. 248, 78-92.

Hutjes, R.W.A., Wierda, A., Veen, A.W.L., 1990. Rainfall interception in the Tai forest, Ivory Coast: Application of two simulation models to a humid tropical system. J. Hydrol. 114, 259-275.
Jackson, N. A., 2000. Measured and modeled rainfall interception loss from an agroforestry system in Kenya. Agric. For. Meteorol. 100, 323-336.
Klaassen, W., Bosveld, F., de Water, E., 1998. Water storage and evaporation as constituents of rainfall interception. J. Hydrol. 212/213, 36-50.
Klaassen, W., 2001. Evaporation from rain-wetted forest in relation to canopy wetness, canopy cover, and net radiation. Water Resour. Res., 37, 3227-3236.
Lankreijer, H., Lundberg, A., Grelle, A., Lindroth, A., Seibert, J., 1999. Evaporation and storage of intercepted rain analysed by comparing two models applied to a boreal forest. Agric. For. Meteorol. 98/99, 595-604.
Lankreijer, H.J.M., Hendriks, M.J., Klaassen, W., 1993. A comparison of models simulating rainfall interception of forests. Agric. For. Meteorol. 64, 187-199.
Lemmon, P. E., 1956. A spherical densiometer for estimating forest overstory density. For. Science 2, 314-320.
Liu, S., 1997. A new model for the prediction of rainfall interception in forest canopies. Ecol. Modell. 99, 151-159.
Liu, S., 1998. Estimation of rainfall storage capacity in the canopies of cypress wetlands and slash pine uplands in North-Central Florida. J. Hydrol. 207, 32-41.
Liu, S., 2001. Evaluation of the Liu model for prediction rainfall interception in forests world-wide. Hydrol. Processes 15, 2341-2360.
Llorens, P., Gallart, F., 2000. A simplified method for forest water storage capacity measurement. J. Hydrol. 240, 131-144.
Llorens, P., Poch, R., Latron, J., Gallart, F., 1997. Rainfall interception by a Pinus sylvestris forest patch overgrown in a Mediterranean mountainous abandoned area I. Monitoring design and results down to the event scale. . J. Hydrol. 199, 331-345.
Lloyd, C.R., Marques, F.A.de.O., 1988. Spatial variability of throughfall and stemflow measurements in Amazonian rain forests. Agric. For. Meteorol. 42, 63-73.
Loescher, H. W., Powers, J. S., Oberbauer, S. F., 2002. Spatial variation of throughfall volume in an old-growth tropical wet forest, Costa Rica. J. Trop. Ecol. 18, 397-407.
Loustau, D., Berbigier, P., Granier, A., 1992. Interception loss, throughfall and stemflow in a maritime pine stand, II, An application of Gash's analytical model of interception. J. Hydrol. 138, 469-485.
Návar, J., Charles, F., Jurando, E., 1999. Spatial variations of interception loss components by Tamaulipan thornscrub in northeastern Mexico. For. Ecol. Manage. 124, 231-239.
Rutter, A. J., Dershaw, K. A., Robins, P. C., Morton, A. J., 1971. A predictive model of rainfall interception in forests. I. Derivation of the model from observations in a plantation of Corsican pine. Agric. Meteorol. 9, 367-384.
Scatena, F.N., 1990. Watershed scale rainfall interception on two forested watersheds in the Luquillo mountains of Puerto Rico. J. Hydrol. 113, 89-102.

Schellekens, J., Bruijnzeel, L.A., Scatena, F.N., Bink, N.J., Holwerda, F., 2000. Evaporation from a tropical rainforest, Luquillo Experimental Forest, eastern Puerto Rico. Water Resour. Res. 36, 2183-2196.

Sinun, W., Meng, W.M., Douglas, I., Spencer, T., 1992. Throughfall, stemflow, overland flow and throughflow in the Ulu Segama rain forest, Sabah, Malasia. Phil. Trans. R. Soc. Lond. B 335, 389-395.

Teklehaimanot, Z., Jarvis, P. G., 1991. Direct Measurement of Evaporation of Intercepted Water from Forest Canopies. J. Applied Ecol. 28, 603-618.

Tobón Marin, C., Bouten, W., Sevink, J., 2000. Gross rainfall and its partitioning into throughfall, stemflow and evaporation of intercepted water in four forest ecosystems in western Amazonia. J. Hydrol. 237, 40-57.

Valente, F., David, J. S., Gash, J. H. C., 1997. Modeling interception loss for two sparse eucalypt and pine forests in central Portugal using reformulated Rutter and Gash analytical models. J. Hydrol. 190, 141-162.

van Dijk, A. I. J. M., Bruijnzeel, L. A., 2001a. Modeling rainfall interception by vegetation of variable density using an adapted analytical model. Part 1. Model description. J. Hydrol. 247, 230-238.

van Dijk, A. I. J. M., Bruijnzeel, L. A., 2001b. Modeling rainfall interception by vegetation of variable density using an adapted analytical model. Part 2. Model validation for a tropical upland mixed cropping system. J. Hydrol. 247, 239-262.

Whelan, M.J., Anderson. J.M., 1996. Modelling spatial patterns of throughfall and interception loss in a Norway spurce (*Picea abies*) plantation at the plot scale. J. Hydrol. 186, 335-354.

Zeng, N., Shuttleworth, J. W., Gash, J. H. C., 2000. Influence of temporal variability of rainfall on interception loss. Part I. Point analysis. J. Hydrol. 228, 228-241.

Table 1. Land use distribution in the Bonham-1 and Bonham-2 watersheds and study plot tree distribution and characteristics. Plots are acronymed as follows: Wetland (WET), Wetland B (WET_B), Wetland C (WET_C), Pine (PIN), Pine B (PIN_B), Pine C (PIN_C), Pine Plantation (PIP), Mixed (MXD), and Hardwood (HRD).

Forest Community	Watershed Area (m^2)	Watershed Area (%)	Plot	Tree Type	Number of Trees	Percent of Total	Minimum Height (meters)	Maximum Height (meters)	Average Height (meters)	Average Projected Canopy Radius (meters)	Average Diameter (meters)	Trees per Hectare
Bottomland Hardwoods	299,000	10.1	WET	Sweet Gum	29	60	5.0	31.0	12.88	1.62	0.14	1200
				Bay	8	17	5.0	16.0	9.75	1.31	0.07	
				Long L. Pine	4	8	4.0	8.0	6.00	0.90	0.05	
				Water Oak	4	8	4.0	28.0	11.13	1.23	0.12	
				Dog Wood	3	6	6.0	14.5	10.50	2.33	0.11	
			WET_B	Sweet Gum	21	46	6.0	28.0	17.60	1.70	0.23	1150
				Long L. Pine	9	20	14.0	21.0	18.89	1.80	0.23	
				Dog Wood	6	13	4.0	14.0	7.83	1.82	0.08	
				Water Oak	6	13	9.0	28.0	16.58	2.33	0.19	
				Bay	4	9	5.0	13.0	9.13	1.60	0.07	
			WET_C	Sweet Gum	21	54	3.0	25.0	14.98	2.18	0.20	975
				Long L. Pine	8	21	3.8	5.0	4.25	0.78	0.05	
				Dog Wood	8	21	3.0	6.5	4.44	1.19	0.06	
				Water Oak	2	5	9.5	23.0	16.25	1.80	0.18	
Pine	1,293,800	43.7	PIN	Loblolly Pine	41	82	4.0	22.0	13.76	2.08	0.21	556
				Oak	5	10	6.0	15.0	9.80	2.34	0.14	
				Crateagus	4	8	5.0	9.0	6.50	2.38	0.11	
			PIN_B	Loblolly Pine	33	100	4.0	25.5	12.86	2.14	0.20	367

Forest Community	Watershed Area (m^2)	Watershed Area (%)	Plot	Tree Type	Number of Trees	Percent of Total	Minimum Height (meters)	Maximum Height (meters)	Average Height (meters)	Average Projected Canopy Radius (meters)	Average Diameter (meters)	Trees per Hectare
			PIN_C	Long Leaf Pine	13	76	2.5	19.0	11.31	1.21	0.14	189
				Loblolly Pine	3	18	16.0	22.0	19.33	3.30	0.33	
				Dogwood	1	6	7.0	7.0	7.00	1.30	0.20	
Pine Plantation	118,700	4.0	PIP	Long Leaf Pine	80	98	5.0	9.0	8.00	1.00	0.10	2050
				Oak	2	2	8.0	8.0	8.00	0.50	0.07	
Upland Mixed	506,700	17.1	MXD	Oak	30	47	3.0	17.0	11.77	2.30	0.16	711
				Loblolly Pine	23	36	5.0	18.0	13.17	1.47	0.18	
				Cherry	3	5	3.0	8.0	5.67	1.50	0.07	
				Plum	3	5	3.0	4.0	3.50	1.00	0.08	
				Dogwood	3	5	6.0	7.0	6.33	2.17	0.07	
				Crateagus	1	2	5.0	5.0	5.00	1.50	0.07	
				Sassafras	1	2	15.0	15.0	15.00	1.00	0.13	
Upland Hardwoods	542,600	18.3	HRD	Oak	125	98	5.0	10.0	9.00	1.20	0.14	1411
				Long Leaf Pine	2	2	8.0	8.0	8.00	0.70	0.60	

Table 2. Measured precipitation, throughfall, and derived stemflow for 4/04/01 through 6/11/02. Plots are acronymed as follows: Wetland (WET), Pine (PIN), Pine Plantation (PIP), Hardwood (HRD), and Mixed (MXD).

	WET	PIN	PIP	HRD	MXD
Measured precipitation (mm)	752.8	752.8	724.8	724.8	684.9
Measured throughfall (mm)	614.5	580.8	583.3	594.5	553.8
Coefficient of variation of measured throughfall	0.17	0.15	0.14	0.17	0.14
Stemflow (mm)	4.9	4.1	14.2	3.9	3.7
Actual interception (mm)	133.4	167.9	127.3	126.4	127.4
Throughfall % of precipitation	81.6	77.2	80.5	82.0	80.9
Stemflow % of precipitation	0.65	0.54	1.96	0.54	0.54
Interception % of precipitation	17.7	22.3	17.6	17.4	18.6

Table 3. Derived canopy specific parameters, climatic variables, and interception components. Plots are acronymed as follows: Wetland (WET), Pine (PIN), Pine Plantation (PIP), Hardwood (HRD), and Mixed (MXD).

	WET	PIN	PIP	HRD	MXD
Total precipitation during the event, P_G (mm)	752.8	752.8	724.8	724.8	684.9
Number of saturation events, n	71	53	45	52	51
Number of non-saturation events, m	69	87	90	83	76
Mean precipitation rate, $\overline{R}$ (mm hr^{-1})	2.03	2.03	2.02	2.02	1.95
Canopy storage capacity, S (mm)	0.98	1.97	1.70	1.40	1.58
Mean evaporation rate, $\overline{E}$ (mm hr^{-1})	0.10	0.10	0.10	0.10	0.10
Mean canopy cover, c	0.88	0.64	0.43	0.52	0.74
Incident precipitation reaching trunk, p_t (mm)	0.02	0.01	0.05	0.01	0.01
Trunk storage capacity, S_t (mm)	0.16	0.13	0.46	0.08	0.10
Precipitation to fill S_t, P'_t (mm)	9.41	9.29	9.20	6.82	8.20
$\overline{E}$ scaled to c, E_c (mm hr^{-1})	0.09	0.06	0.04	0.05	0.07
Precipitation to fill S, P'_G (mm)	1.14	3.13	4.00	2.73	2.18

Table 4. Measured precipitation, throughfall, and interception data as compared to model results using average annual and seasonal distributed canopy cover values for April 19, 2001 through April 29, 2002. Values in parentheses are the error between measured and modeled. Plots are acronymed as follows: Wetland (WET), Pine (PIN), Pine Plantation (PIP), Hardwood (HRD), and Mixed (MXD).

	WET	PIN	PIP	HRD	MXD
Measured precipitation (mm)	752.8	752.8	724.8	724.8	684.9
Measured interception (mm)	133.4	167.9	127.3	126.4	127.4
Average canopy cover					
Predicted interception (mm) and	126.7	161.7	137.6	113.1	135.2
Error from measured (%)	(5.0)	(3.7)	(-8.1)	(10.5)	(-6.1)
Nash-Sutcliffe model efficiency	0.81	0.91	0.44	0.73	0.86
Seasonal canopy cover					
Predicted interception (mm) and	127.4	166.7	136.6	123.4	134.1
Error from measured (%)	(4.5)	(0.7)	(-7.3)	(2.4)	(-5.3)
Nash-Sutcliffe model efficiency	0.82	0.94	0.44	0.80	0.85

Table 5. Measured precipitation, throughfall, and interception data as compared to model results for density variations within pine and wetland plots using seasonal distributed canopy cover values. Density comparison study occurred from 2/01/02 to 4/29/02. Plots are acronymed as follows: Pine (PIN), Pine B (PIN_B), Pine C (PIN_C), Wetland (WET), Wetland B (WET_B), and Wetland C (WET_C).

	PIN	PIN_B	PIN_C	WET	WET_B	WET_C
Canopy cover range (%)	48 - 80	41 - 55	17 - 34	67 - 87	69 - 85	49 - 82
Average canopy cover (%)	64	44	29	88	78	66
Measured precipitation (mm)	243.0	243.0	243.0	243.0	243.0	243.0
Measured throughfall (mm)	204.0	211.9	211.9	215.3	205.8	208.1
Stemflow (mm)	1.9	1.9	1.9	2.3	2.3	2.3
Actual interception (mm)	37.1	29.2	29.2	25.4	34.9	32.6
Predicted interception (mm)	37.3	32.2	28.7	26.8	26.1[1] 32.8[2]	23.3[1] 30.0[2]
Nash-Sutcliffe model efficiency	0.99	0.69	0.59	0.42	0.60[1] 0.76[2]	0.73[1] 0.98[2]
Percent difference (actual and predicted interception)	-0.5	-10.2	1.7	-5.7	25.2[1] 6.3[2]	28.5[1] 8.2[2]

[1]Calculated using unadjusted canopy cover and canopy storage capacity.
[2]Calculated using adjusted canopy cover and canopy storage capacity.

Table 6. Watershed scale analysis for cumulative interception by land-use and scaled to overall watershed using an area weighting approach for April 19, 2001 through April 29, 2002 in Columbus, GA. Plots are acronymed as follows: Wetland (WET), Pine (PIN), Pine Plantation (PIP), Mixed (MXD), and Hardwood (HRD).

		Precipitation (mm)	Watershed Total (mm)	WET (mm)	PIN (mm)	PIP (mm)	MXD (mm)	HRD (mm)
Annual average canopy cover	Spring	250.5	40.9	37.9 (1.08)	48.7 (0.84)	42.9 (0.95)	44.5 (0.92)	35.2 (1.16)
	Summer	231.3	48.3	42.9 (1.12)	57.6 (0.84)	51.3 (0.94)	52.7 (0.92)	42.1 (1.15)
	Fall	138.2	23.6	22.3 (1.06)	27.9 (0.85)	25.4 (0.93)	25.9 (0.91)	20.3 (1.16)
	Winter	215.7	35.7	33.5 (1.07)	42.3 (0.84)	37.2 (0.96)	39.1 (0.91)	30.8 (1.16)
	Total	835.6	148.4	136.6 (1.09)	176.5 (0.84)	156.7 (0.95)	162.2 (0.92)	128.4 (1.16)
Seasonal canopy cover	Spring	250.5	44.8	39.2 (1.14)	54.0 (0.83)	46.1 (0.97)	46.8 (0.96)	40.2 (1.11)
	Summer	231.3	55.0	47.0 (1.17)	64.6 (0.85)	43.2 (1.27)	57.9 (0.95)	56.5 (0.97)
	Fall	138.2	22.7	24.5 (0.92)	25.2 (0.90)	23.1 (0.98)	27.8 (0.82)	19.0 (1.19)
	Winter	215.7	33.4	30.6 (1.09)	41.2 (0.81)	42.5 (0.79)	34.6 (0.97)	25.8 (1.30)
	Total	835.6	155.8	141.4 (1.10)	185.0 (0.84)	154.9 (1.01)	167.1 (0.93)	141.5 (1.10)

Numbers in parenthesis represent the ratio of the watershed total to that predicted by land-use

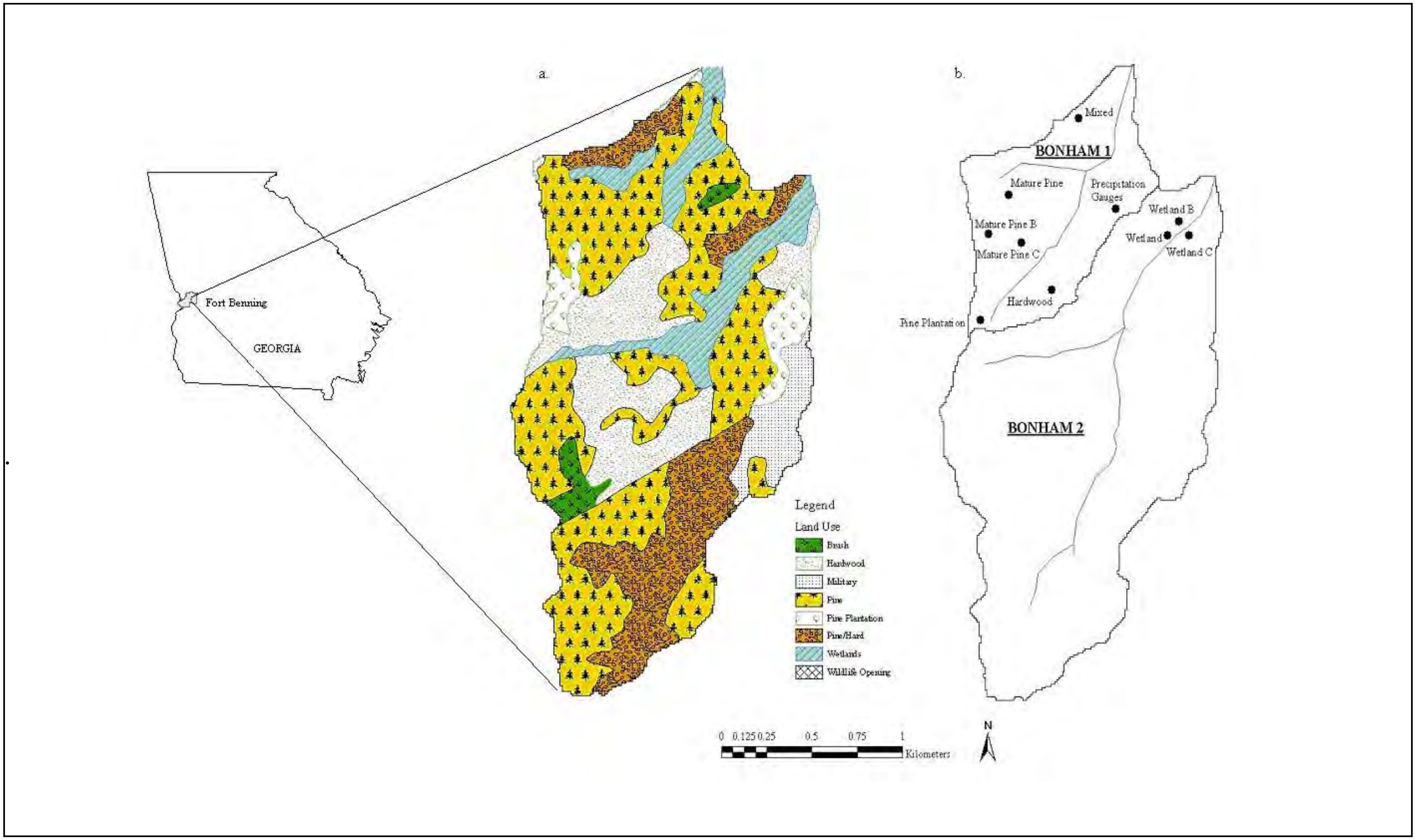


Figure 1. Bonham-1 and Bonham-2 Study watersheds in Fort Benning, GA with (a) land use distribution, and (b) plot locations for interception determination indicated.

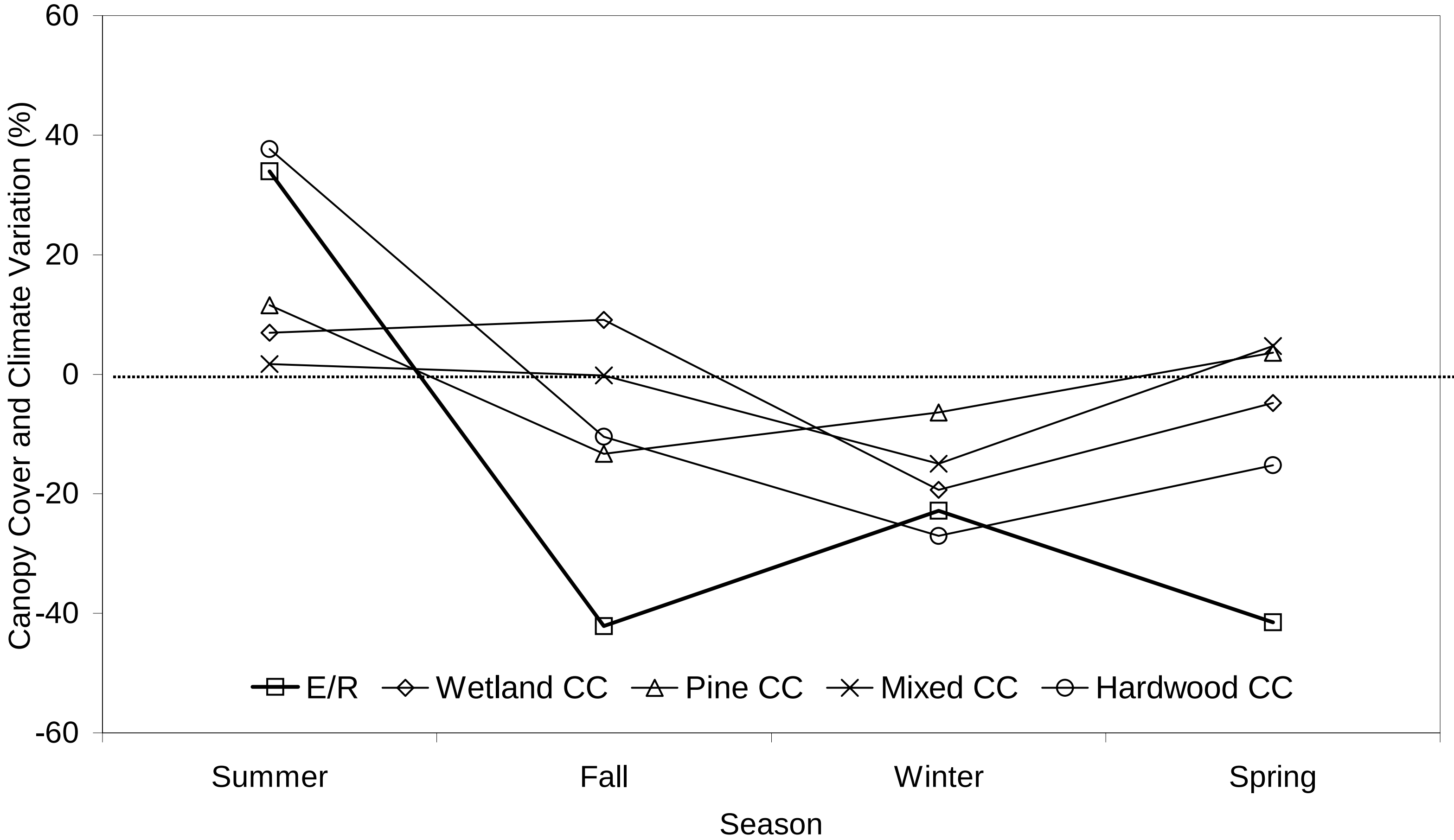


Figure 2. Seasonal variation of canopy cover by land cover and evaporation to rainfall intensity rates relative to annual average values.

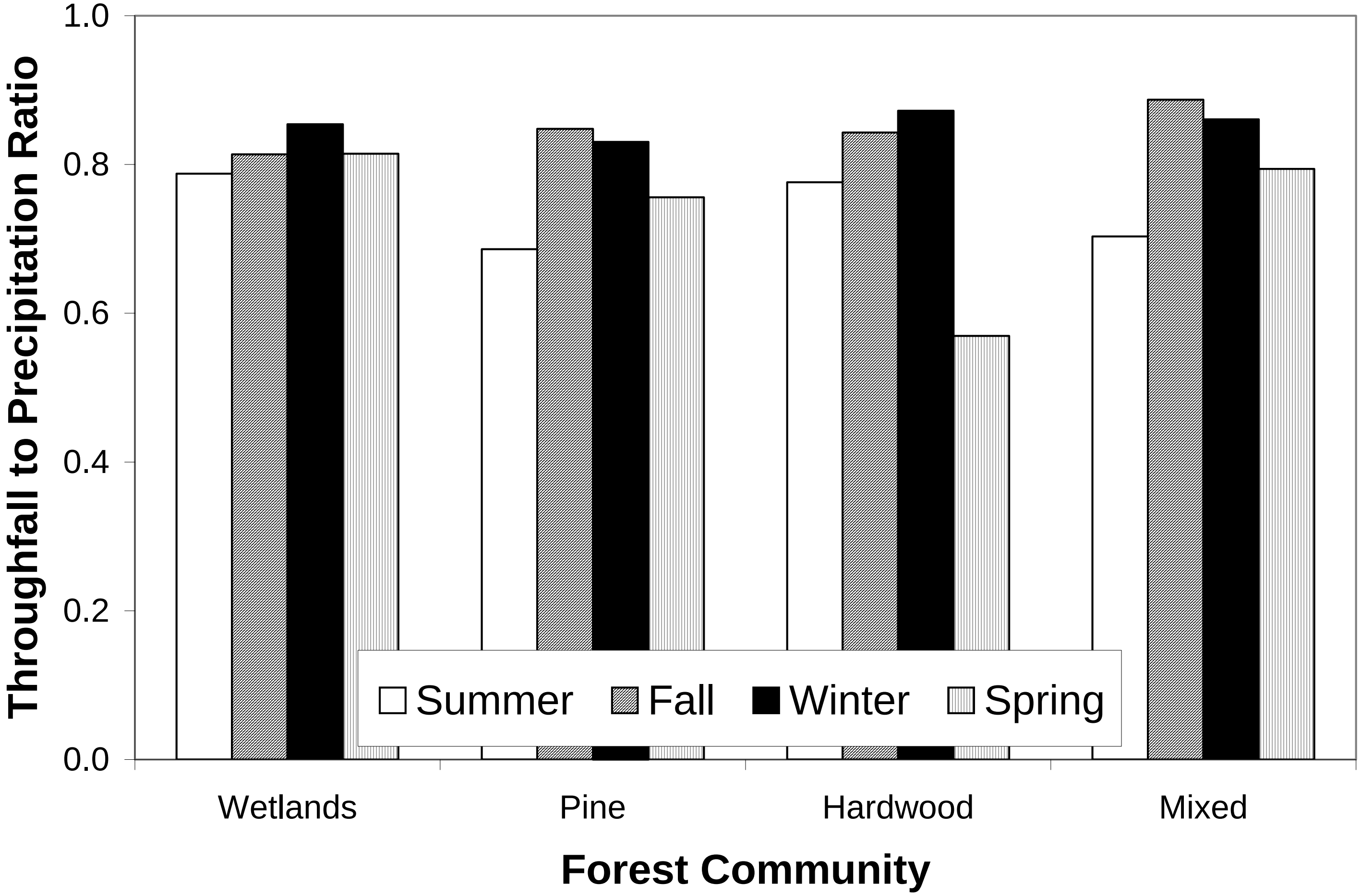


Figure 3. Seasonal variation of throughfall to precipitation ratio by land cover.

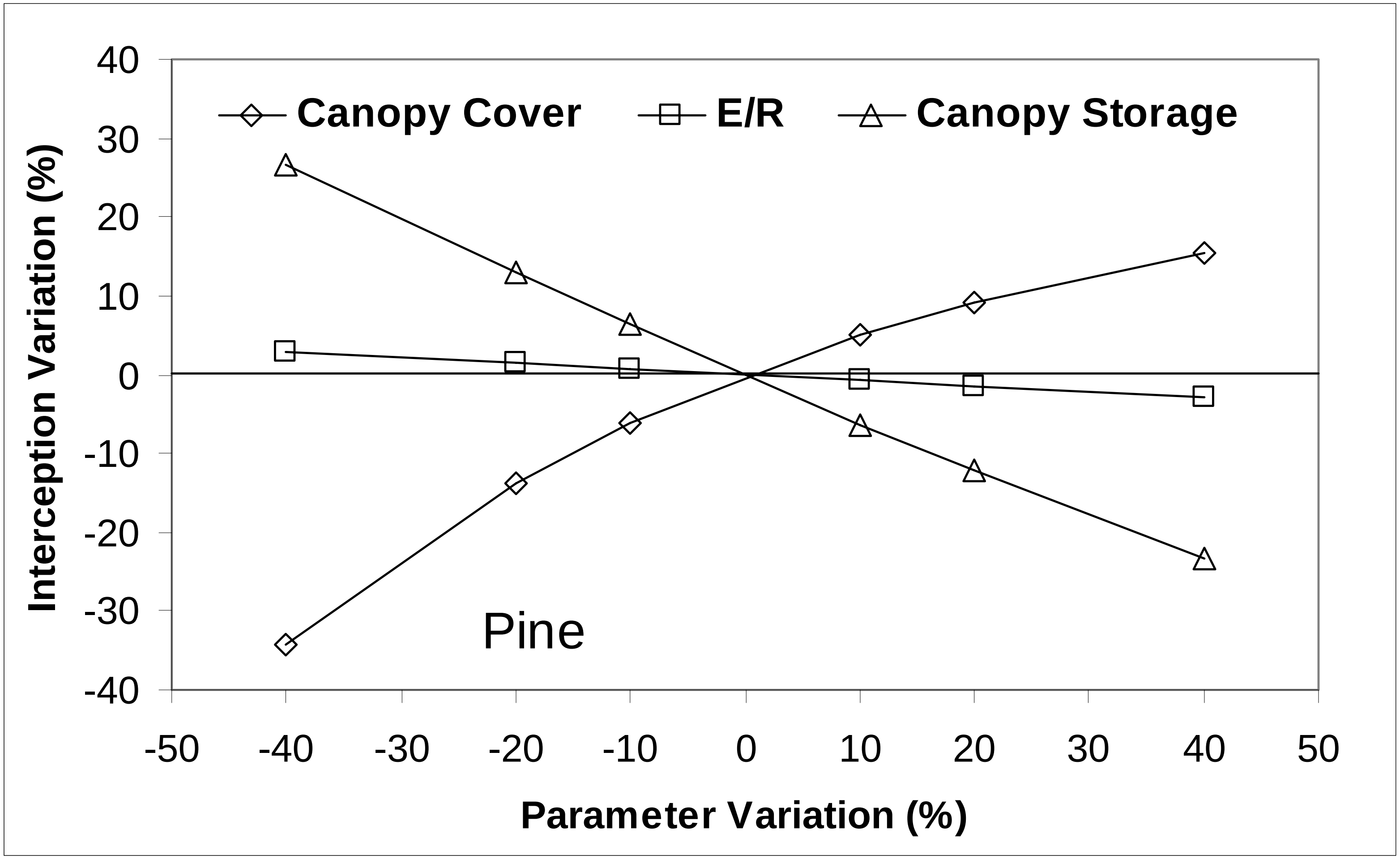


Figure 4. Sensitivity of modeled interception to canopy cover, ratio of evaporation to rainfall intensity rates (E/R) and canopy storage capacity for pine plot. Parameter sensitivity is relative to mean observed value. Interception variation is relative to model results using observed values.

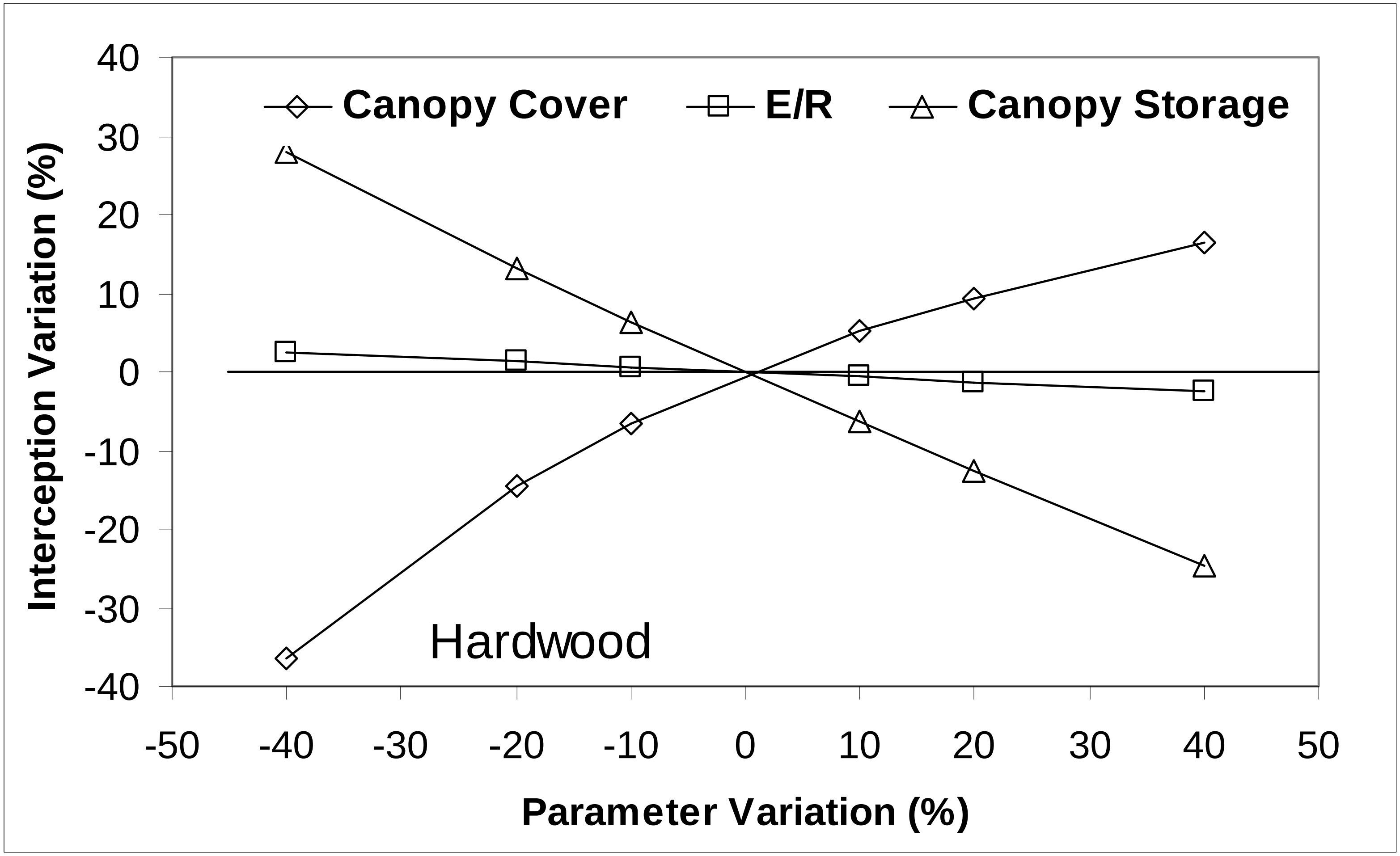


Figure 5. Sensitivity of modeled interception to canopy cover, ratio of evaporation to rainfall intensity rates (E/R) and canopy storage capacity for hardwood plot. Parameter sensitivity is relative to mean observed value. Interception variation is relative to model results using observed values.

3.3.2
Hydrologic Indices of Watershed Scale Military Impacts in Fort Benning, GA
Shirish Bhat, Jennifer M. Jacobs, Kirk Hatfield, and Wendy D. Graham

Abstract
The magnitude, frequency, duration, timing, and rate of change of hydrologic conditions regulate ecological processes in aquatic ecosystems. These components characterize the entire range of flows and specific hydrologic phenomena, such as floods or low flows that are critical to the integrity of such ecosystems. By defining flow regimes in these terms, the ecological consequences of particular human activities that modify one or more components of the flow regime can be considered explicitly. We compare a reference watershed with an impacted watershed at Fort Benning, Georgia, in terms of hydrologic indices that are derived from long-term daily flow data. The comparison of the results between a reference and impacted watershed show a clear distinction between the watersheds in terms of hydrological flow regimes. The reference watershed showed more vulnerability to stream ecology as it produced higher magnitude flows, more frequent low- and high-flows. The constancy of the daily flows, and groundwater input, however, were higher in impacted watershed. Eight out of 26 non-redundant hydrologic indices are recommended for Fort Benning region for ecological analysis.

We also proposed 18 storm-based indices, grouped into four flow components, to statistically characterize hydrologic variation among different watersheds. These 18 indices provide information on ecologically significant flow regimes that influence aquatic, wetland, and riparian ecosystems. 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. Correlation and regression analyses were performed to determine the relationship among the watershed physical characteristics and the storm-based hydrologic indices. A number of significant relationships were found. The correlation results show that the increase in road density increased the variability in the peak discharges and the slopes of the rising limb. The increase in the military land increased the time of rise as well as the variability in the time base. The number of roads crossing streams is positively correlated with the response lag, whereas it is negatively correlated with the time base and the variability in the slopes of the falling limb. Increase in the bare land and the disturbance index increased the time of rise as well as the variability in the time base. Stepwise multiple correlations identified the relationships between the event indices and the management related watershed physical characteristics that are susceptible to the disturbances. Military land, road density, and the number of roads crossing streams predicted storm-based baseflow index, bankfull discharge, response lag, and time of rise well.

KEY WORDS: Military reservation, ecohydrology, watershed disturbance, hydrologic indices, flow regimes

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 no 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 hydrological 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 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 (IHAs: 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 (U.S.A). Towards this end, we investigate both storm and annual hydrographs in this study. We hypothesize 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 contrast impacted watersheds with a reference watershed. Additionally, indices are related to specific military land management practices.

Background

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 different stream types that may differ in major aspects of ecological organization. Poff (1996) provides a comprehensive catalog of the stream type 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 1. The numerous time characteristics of the hydrograph and its relationship to the precipitation event are detailed in Table 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 watersheds is proposed. The proposed indices include 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 (Table 2). 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 groups of hydrologic regimes, magnitude, frequency, duration, and rate of change, to statistically characterize hydrologic variation.

Magnitude

The magnitude 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).

Duration

The 6 parameters in the duration 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 (Williams and Hynes, 1977), and fishes (Closs and Lake, 1996), prolonged flows of particular levels can also be damaging. 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).

Frequency

The two parameters in the frequency 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 discharge above a threshold of two times the bankfull discharge 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).

Rate of change

Four parameters, mean rate of change in peak discharge in rising limb and falling limb of the storm hydrograph and their variabilities, in the rate of change group measure how rapidly water attenuates or leaves the watershed. 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 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). In addition to standard watershed characteristics, a dimensionless military disturbance parameter was calculated. The parameter, disturbance index (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., 2004). Additional details regarding the study area are provided elsewhere (Bhat et al., Ecological indicators in forested watersheds in Fort Benning, GA: Relationship between land use and stream water quality, submitted to *Ecological Indicators*, 2004; hereinafter referred to as Bhat et al., submitted manuscript, 2004).

According to variations in ecologically relevant hydrological characteristics that are based on flow variability and predictability, and low- and high-flow extremes, the streams in the study watersheds are classified as 'perennial runoff' (Poff, 1996). As the present study focuses on the impacts of military land use 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 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, the hydrologic indices are adapted from the procedure outlined in the Indicators of Hydrologic Alteration (IHA) method (Richter et al., 1996). The adaptation uses only nonredundant yet biologically significant hydrologic indices as per the recommendations of Olden and Poff (2003). Table 4 lists these indices for the perennial runoff type of streams found in the study region. The definitions and the methods used to calculate these indices are listed in Appendix 1. In this study, the annual-based indices are used to 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 2). Appendix 2 lists the indices, definitions and methods of calculation.

Statistical Analyses

The ANOVA and the Tukey's multiple comparison tests were performed to test for determining the differences between reference watershed and the impacted watersheds' mean values of indices. The percent of forest extent, military land fraction, 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. Most of the watersheds are highly vegetated (70% or more) 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 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 extent (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 Ft. Benning installation. While road density is relatively comparable among watersheds, the number of roads crossing streams varies from 1 to 21. Watershed characteristics strongly correlated at p-values of 0.05 or lower are military land with percent bare land and disturbance index, and percent bare land with disturbance index.

Annual-Based Hydrologic Indices

Table 4 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 as compared to the reference. 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) in the impacted watershed. 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) was 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) for about 2-3 days a year. 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 higher than the impacted watershed. However, on a few occasions, the flow crossed a lower threshold of 3 times median (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) is 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 2. ANOVA 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 indicates that the baseflow and peak discharge in the further confirmed that the mean values of 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.

The storm flows consistently lasted longer and responded faster to rain events in the reference watershed (D_MTB and D_VTB) as compared to the impacted ones. 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.

5.4 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 5 indicates that 7 key storm-based hydrologic indices are significantly related to military land management. Increased military land extent, bare land, and 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. The results also show that the 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, however, identified for forestry management practices.

Stepwise multiple correlations characterized the response of storm-based indices to military impacts (Table 6). Military land extent, road density, and the number of road crossing streams were the three management variables that impacted storm responses. The greatest impact of land management is found with statistically significant predictive models for indices of time base, response lag, and time of rise.

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). While this study's watersheds are relatively small, the general changes to the energy base of stream ecosystems, invertebrate community composition, and metabolism due to reduced overall flow may be guided by earlier studies' findings.

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. 2004). 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. This statement is true as the variability in the duration during low flow condition (D_L10, D_L17, and D_L6) is higher in the reference watershed. 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) as compared to the impacted one. 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, Additionally, 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 land extent, 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 results from 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 few examples of roads on terrestrial and aquatic ecosystems. Moreover, road crossings commonly act as barriers to the movement of fishes and other aquatic animals (Furniss et al., 1991). Roads 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, we compare a reference watershed 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. We recommend 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 land extent, road density, and the number of road crossing streams, were found to significantly affect these indices.

Reference

Auble, G.T., Friedman, J.M., Scott, M.L., 1994. Relating riparian vegetation to present and future streamflows. Ecological Applications. 4, 544-554.

Beschta, R.L., Jackson, W.L., 1979. The intrusion of fine sediments into a stable gravel bed. J. Fish. Res. Board Can. 36, 207–210.

Biggs, B.J.F., 1996. Hydraulic habitat of plants in streams. Regulated Rivers: Research and Management 12, 131-144.

Biggs, B.J.F., 2000. Eutrophication of streams and rivers: dissolved nutrient-chlorophyll relationships for benthic algae. Journal of the North American Benthological Society. 19, 17-31.

Biggs, B.J.F., Close, M.E., 1989. Periphyton biomass dynamics in gravel bed rivers: the relative effects of flows and nutrients. Freshwater Biology 22, 209-231.

Biggs, B.J.F., Smith, R.A., Duncan, M.J., 1999. Velocity and sediment disturbance of periphyton in headwater streams: biomass and metabolism. Journal of the North American Benthological Society 18, 144-163.

Biggs, B.J.F., Stokseth, S., 1996. Hydraulic habitat preferences for periphyton in rivers. Regulated Rivers: Research and Management 12, 251-261.

Bhat S., Jacobs, J.M., Hatfield, K., Prenger, J., 2004. Ecological indicators in forested watersheds in Fort Benning, GA: Relationship between land use and stream water quality. Submitted to Ecol. Indicators.

Bren, L.J., 1992. Tree invasion of an intermittent wetland in relation to changes in the flooding frequency of the River Murray, Australia. Australian Journal of Ecology. 17, 395-408.

Bryant, M.L., Bhat, S., Jacobs, J.M., 2004. Characterization and modeling of throughfall temporal variability for forest communities in the southeastern U.S. J. of Hydrology. Resubmitted.

Cairns, Jr., J., McCormick, P.V., Neiderlehner, B.R., 1993. A proposed framework for developing indicators for ecosystem health. Hydrobiologia, 263, 1-44.

Chapman, R.J., Hinckley, T.M., Lee, L.C., Teskey, R.O., 1982. *Impact of water level changes on woody riparian and wetland communities. Vol. 10. Kearneysville (WV),* US Fish & Wildlife Service. Publication no. OBS-82/83.

Clausen, B., Biggs, B.J.F., 1997. Relationships between benthic biota and hydrological indices in New Zealand streams. Fresh. Biol. 38, 327–342.

Clausen, B., Biggs, B.J.F., 2000. Flow indices for ecological studies in temperate streams: groupings based on covariance. J. Hydrol. 237, 184–197.

Close, M.E., Davies-Colley, R.J., 1990. Baseflow water chemistry in New Zealand rivers. 2. Influence of environmental factors. New Zealand Journal of Marine and Freshwater Research 24, 343-356.

Closs, G.P., Lake, P.S., 1996. Drought, differential mortality and the coexistence of a native and an introduced fish species in a south east Australian intermittent stream. Env. Biol. Fishes. 47, 17–26.

Cobb, D.G., Galoway, T.D., Flannagan, J.F., 1992. Effects of discharge and substrate stability on density and species composition of stream insects. Canadian Journal of Fisheries and Aquatic Sciences 49, 1788-1795.

Colwell, R.K., 1974. Predictability, constancy, and contingency of periodic phenomena. Ecology. 55, 1148–1153.

Davies-Colley, R.J., Quinn, J.M., 1998. Stream lighting in Æve regions of North Island, New Zealand: control by channel size and riparian vegetation. New Zealand Journal of Marine and Freshwater Research 32, 591-606.

Fisher, S.G., 1983. Succession in streams. In Barnes, J.R., Minshall, G.W. (eds.) *Stream ecology: application and testing of general ecological theory*. New York: Plenum Press.

Grimm, N.B., Fisher, S.G., 1992. Responses of arid-land streams to changing climate. In: Firth, P., Fisher, S.G. (Eds.). Global Climate Change and Freshwater Ecosystems, Springer, New York, pp. 211-233.

Furniss, M.J., Roeloffs, T.D., Yee, C.S., 1991. Road construction and maintenance. In

Meehan, W.R. (ed.) Influences of forest and rangeland management on salmonid fishes and their habitats. Special publication 19. American fisheries Society, Bethesda, Maryland. pp. 297-323.

Gjessing, E., Lygren, E., Berglind, L., Gulbrandsen, T., Skanne, R., 1984. Effect of highway runoff on lake water quality. Science of the Total Environment. 33, 247-257.

Hanson, D.L., Waters, T.F., 1974. Recovery of standing crop and production rate of a brook trout population in a flood-damaged stream. Trans. Am. Fish. Soc. 103, 431-439.

Helvey, J.D., Kochenderfer, J.N., 1990. Soil density and moisture content on two unused forest roads during first 30 months after construction. Research paper NE-629. U.S. Forest Service. Northeast Forest Experiment Station, Broomhall, Pennsylvania.

Hemphill, N., Cooper, S.D., 1983. The effect of physical disturbance on the relative abundances of two filter-feeding insects in a small stream. Oecologia. 58, 378-383.

Horwitz, R.J., 1978. Temporal variability patterns and the distributional patterns of stream fishes. Ecol. Monogr. 48, 307-321.

Jones, J.A., Grant, G.E., 1996. Cumulative effects of forest harvest on peak streamflow in the western cascades of Oregon. Water Resources Research. 32, 959-974.

Jowett, I.G., Duncan, M.J., 1990. Flow variability in New Zealand rivers and its relationship to in-stream habitat and biota. New Zealand Journal of Marine and Freshwater Research. 24, 305–317.

Junk, W.J., Bayley, P.B., Sparks, R.E., 1989. The flood pulse concept in river-floodplain systems. Canadian Special Publication of Fisheries and Aquatic Sciences. 106, 110–127.

Keller, E.A., Swanson, F.J., 1979. Effects of large organic material on channel form and fluvial processes. Earth Surface Processes and Landforms. 4, 351–380.

Leopold, L.B., Wolman, M.G., Miller, J.P., 1964. *Fluvial processes in geomorphology*. San Francisco, W. H. Freeman and Sons.

Maloney, K.O., Mulholland, P.J., Feminella, J.W., 2004. Influence of catchment-scale military land use on physicochemical conditions in small Southeastern Plains streams (USA). Submitted to Land Use and Stream Physicochemistry.

Meffe, G.K., 1984. Effects of abiotic disturbance on coexistence of predator and prey fish species. Ecology. 65, 1525–1534.

Meffe, G.K., Minckley, W.L., 1987. Persistence and stability of fish and invertebrate assemblages in a repeatedly disturbed Sonoran Desert stream. Am. Midl. Nat. 117, 177-191.

Milchunas, D.G., Schultz, K.A., Shaw, R.B. 1999. Plant community response to disturbance by mechanized military maneuvers. J. Env. Qual. 28, 1533-1547.

Minckley, W.L., Deacon, J.E., 1991. *Battle against extinction: native fish management in the American West.* University of Arizona Press, Tucson, AZ.

Minckley, W.L., Meffe, G.K., 1987. Differential selection by flooding in stream-fish communities of the arid American Southwest. In Matthews, W.J., Heins, D.C. (eds.) *Community and evolutionary ecology of North American stream fishes.* University of Oklahoma Press, Norman, OK.

Minshall, G.W., 1984. Aquatic insect-substratum relationships. In: Resh, V.H., Rosenberg, D.M. (Eds.). The Ecology of Aquatic Insects, Praeger ScientiÆc, pp. 358-400.

Moore, K.M.S., Gregory, S.V., 1988. Response of young-of-the-year cutthroat trout to manipulations of habitat structure in a small stream. Transactions of the American Fisheries Society. 117, 162–170.

Moss, D., Furse, M.T., Wright, J.F., Armitage, P.D., 1987. The prediction of the macroinvertebrate fauna of unplotted running-water sites in Great Britain using environmental data. Freshwater Biol. 17, 41-52.

Olden, J.D., Poff, N.L., 2003. Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. River Res. Applic. 19, 101-121.

Pettit, N.E., Froend, R.H., Davies, P.M., 2001. Identifying the natural flow regime and the relationship with riparian vegetation for two contrasting western Australian rivers. Regulated Rivers: Research and Management. 17, 201-215.

Piegay, H., Landon, N., 1997. Promoting ecological management of riparian forest on the Drome River, France. Aquatic Conservation: Marine and freshwater Ecosystems. 7, 287-304.

Poff, N.L., 1996. A hydrogeography of unregulated streams in the United States and an examination of scale-dependence in some hydrological descriptors. Freshwater Biology,
36, 101-121.

Poff, N.L., Allan, J.D., 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology. 76, 606-627.

Poff, N.L., Allan, J.D., Bain, M.B. , Karr, J.R., Prestegaard, K.L., Richter, B.D., Sparks, R.E., Stromberg, J.C., 1997. The natural flow regime: A paradigm for river conservation and restoration. BioScience. 47, 769-784.

Poff, N.L., Ward, J.V., 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patterns. Canadian Journal of Fisheries and Aquatic Sciences. 46, 1805–1818.

Power, M.E., Stewart, A.J., 1987. Disturbance and recovery of an algal assemblage following flooding in an Oklahoma stream. Am. Midl. Nat. 117, 333-345.

Quist, M.C., Fay, P.A., Guy, C.S., Knapp, A.K., Rubenstein, B.N., 2003. Military training effects on terrestrial and aquatic communities on a grassland military installation. Ecol. Appl. 13, 432-442.

Resh, V.H., Brown, A.V., Covich, A.P., Gurtz, M.E., Li, H.W., Minshall, G.W., Reice, S.R., Sheldon, A.L., Wallace, J.B., Wissmar, R., 1988. The role of disturbance in stream ecology. Journal of the North American Benthological Society. 7, 433–455.

Richter, B.D., Baumgartner, J.V., Powell, J., Braun, D.P., 1996. A method for assessing hydrologic alteration within ecosystems. Conservation Biology. 10, 1163–1174.

Schlosser, I.J., 1985. Flow regime, juvenile abundance, and the assemblage structure of stream fishes. Ecology. 66, 1484–1490.

Scott, M.L., Friedman, J.M., Auble, G.T., 1996. Fluvial processes and the establishment of bottomland trees. Geomorphology. 14, 327-339.

Stromberg, J.C., Fry, J., Patten, D.T., 1997. Marsh development after large floods in an alluvial, arid-land river. Wetlands. 17, 292–300.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., Cushing, C.E., 1980. The river continuum concept. Canadian Journal of Fisheries and Aquatic Sciences 37, 130-137.

Wallace, J.B., Benke, A.C., 1984. Quantification of wood habitat in subtropical coastal plains streams. Canadian Journal of Fisheries and Aquatic Sciences. 41, 1643–1652.

Welcomme, R.L., 1992. River conservation - future prospects. In Boon, P.J., Calow, P., Petts, G.E. (eds.) *River conservation and management*. John Wiley and Sons. New York.

Wemple, B.C., Jones, J.A., Grant, G.E., 1996. Channel network extension by logging roads in two basins, western Cascades, Oregon. Water Res. Bul. 32, 1195-1207.

Williams, D.D., Hynes, H.B.N., 1977. The ecology of temporary streams. II. General remarks on temporary streams. Internationale Revue des gesampten Hydrobiologie. 62, 53–61.

Wilson, S.D., 1988. The effects of military tank traffic on prairie: a management model. Envir. Manage. 12, 397-403.

Wood, P.J., Agnew, M.D., Petts, G.E., 2000. Flow variations and macroinvertebrate community responses in a small groundwater-dominated stream in south-east England. Hydrological Processes. 14, 3133–3147.

Table 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

Table 2. 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. * represent 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 2

Flow components and name of the hydrologic index	Symbol	Units	Reference BON-1	Impacted BON-2	Impacted BON	Impacted LPK	Impacted 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	2.00	2.30	1.20	1.00	1.40
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

Table 3. 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 Land, %	0	6	5	2	2
NDVI	0.36	0.30	0.32	0.34	0.36
Soil					
Sand, %	78	69	69	72	49
Loam, %	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

Table 4. 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. 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 1

Flow components and name of the hydrologic index	Symbol	Units	BON-1 (Reference)	BON-2 (Impacted)	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
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		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

Table 5. Pearson correlation coefficients between watershed physical characteristics and event based hydrologic indices. Characteristics are acronymed as follows: Forest (FOR), Military land (MIL), Road Density (RDN), No. of Roads Crossing Streams (NRC), % Bare Land (PBL), and Disturbance Index (DIN). For details of Hydrologic Indices, refer to Appendix 2. * 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

Table 6. Stepwise multiple regression models for event based hydrologic indices. Refer to Table 5 for the definitions of the indices. Military 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)**
$M_M BF$	0.729 + 1.91 MIL + 0.00968 NRC	0.94*
$M_V PD$	- 1.21 + 0.666 RDN	0.88**
$F_1 FD$	0.929 - 6.97 MIL - 0.039 NRC	0.94*
$D_M TB$	1.8 - 0.0352 NRC	0.77*
$D_V TB$	0.418 + 3.05 MIL	0.95**
$D_M RL$	0.483 + 0.0393 NRC	0.92**
$D_M TR$	0.454 + 0.837 MIL	0.97**
$R_{NV} PD$	2.20 – 0.0563 NRC	0.70*

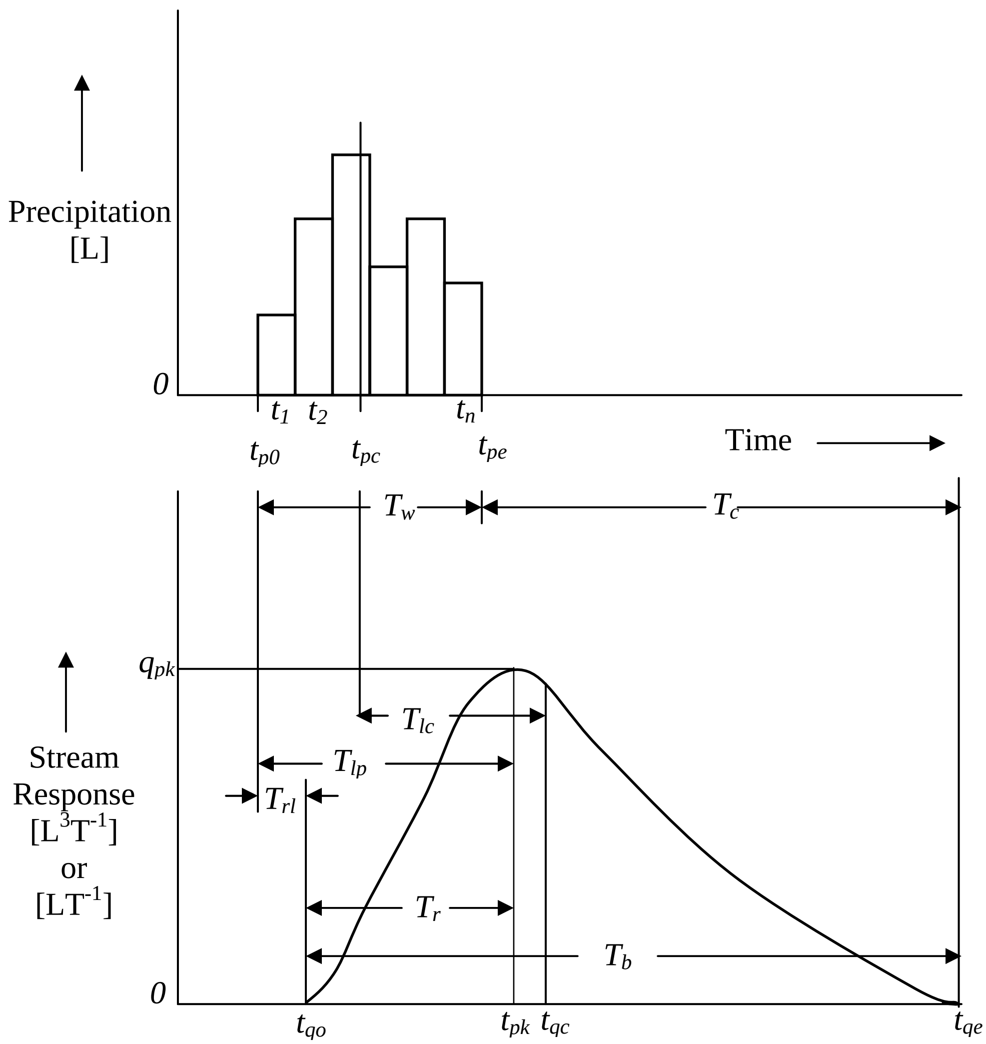


Figure 1. Terms used to describe hyetograph and response hydrograph. Refer to Table 1 for the definitions of the terms.

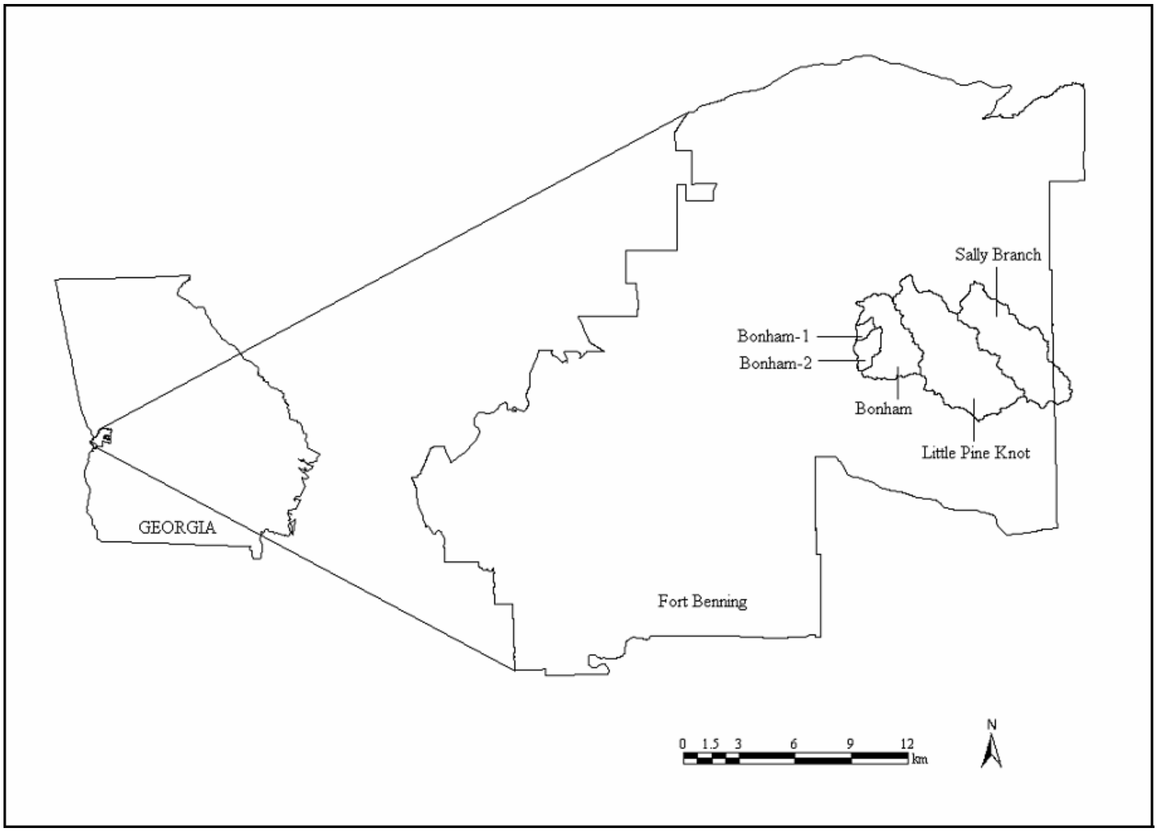


Figure 2. Study watersheds, Bonham-1, Bonham-2, Bonham, Little Pine Knot, Randall, Sally Branch, and Oswichee, in the Fort Benning military installation.

Appendix 1: Definitions and calculation procedures for the indices reported in Table 4.

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	• Extract lowest yearly flow ($q_{i,\min}$), i = 1,…, n; , n = no. of years

	averaged across all 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 3 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	• 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})

	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 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 Fh4
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, $i = 1,\ldots, n$; , 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, $i = 1,\ldots, n$; , n = no. of years • Calculate $\bar{Q}_{30\max} = \sum_{i=1}^{n} Q_{30\max i} / n$ • Calculate median flow (Q_{mi}),$i = 1,\ldots, n$; , 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}), $i = 1,\ldots, n$; , 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 ($Q_{1.67yri}$), $i = 1,\ldots, n$; , 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_j), 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	• Determine flow of magnitude exceeding a return interval of 1.67 years based on log-normal distribution ($Q_{1.67yr}$) over the period of

	one of six 60-day 'seasonal' windows	record • Divide every year into six 60-day 'seasonal' windows • Determine the maximum number of $Q_{1.67yr}$ exceedance days ($D_{1.67yr\max}$) over period of record in any one of six 60-day 'seasonal' windows • Determine the total number of $Q_{1.67yr}$ exceedance days over all years in one of six 60-day seasonal window ($D_{1.67yrtot}$) • Calculate the ratio ($D_{1.67yr\max}/D_{1.67yrtot}$)
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 ($Q_{1.67yr}$) over the period of record • Determine the maximum number of $Q_{1.67yr}$ non-exceedance days ($D_{1.67yrnon}$) over period of record • Determine the total number of $Q_{1.67yr}$ non-exceedance days over period of record ($D_{1.67yrtot}$) • Calculate the ratio ($D_{1.67yrnon}/D_{1.67yrtot}$)
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	• 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

	flow	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 2. Definitions and calculation procedures for the indices reported in Table 2.

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 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, $\left(\bar{q_{pk} / A}\right) = \sum_{i=1}^{n} \left(\frac{q_{pk}}{A}\right)_i / n$
$M_V PD$	Coefficient of variation in peak	• Calculate $\left(\bar{q_{pk} / A}\right)$

	discharges	• 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}), $i = 1,\ldots,n$; 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	• Calculate mean of R_PPD_i • Calculate (*SD*) of R_PPD_i • Calculate $CV = SD$/mean of R_PPD_i

	peak discharge in rising limb	
$R_N PD$	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 $R_N PD_i$ • Calculate (SD) of $R_N PD_i$ • Calculate $CV = SD$/mean of $R_N PD_i$

3.3.3 Impacts on Soil Saturated Hydraulic Conductivity: Mechanized Military Training Characterization Using Five Measurement Methods. D.B. Perkins, J.W. Jawitz, N.W. Haws, P.S.C. Rao.

INTRODUCTION

Mechanized military training activities and land management practices vary in their relative potential to degrade or alter soil physical properties and landscape features [Trumbull et al., 1994; Garten et al., 2000; Prosser et al., 2000; Grantham et al., 2001; Dilustro et al., 2002; Dale et al., 2002; Fang et al., 2002; Kade and Warren, 2002; Fuchs et al., 2003; Halvorson et al., 2003; Quist et al., 2003; Wanner and Xylander; 2003]. Other military land management practices, such as periodic controlled burning of forest understory in forested military lands, has also been investigated to identify impacts to soil properties and the biotic community [Doerr et al., 1998; Prosser and Williams, 1998; Robichaud and Hungerford, 2000; and Huffman et al., 2001]. Even Infantry foot traffic has also been shown to affect soil surface properties [Whitecotton et al., 2000].

Tracked mechanized tank training at Fort Benning routinely occurs on ridgetops, creating a situation in which the impacts to soil quality are up gradient from all other catchment processes and features. Mechanized training strips all vegetation from the immediate training area, providing a continuous source of sediment from barren and exposed soils. The larger scope of this study is motivated by efforts to better identify impacts of mechanized training on soil properties, evaluation of hydrologic dynamics controlling stream hydrograph contribution from upland areas, and management/restoration implications for other existing and future training areas.

The research findings of other investigators [Garten et al., 2000; Grantham et al., 2001; Dale et al., 2002; Fuchs et al., 2003; and Quist et al., 2003], paired with observational evidence, suggest that there is a significant difference in soil physical properties between non-training sites and tracked training sites. Acting on this premise, we chose to characterize saturated hydraulic conductivity (K_{sat}) in typical non-training and tracked training sites within the Troup Soil Series. K_{sat} was chosen because it is not only a single easily measured soil hydraulic parameter, but it was hypothesized that it would be a sensitive enough to reveal differences between the non-training reference site and the heavily tracked training site.

It was also hypothesized that different K_{sat} measurement methods would have different levels of sensitivity to the differences between the two land uses. Here, the positive-pressure in-situ single-ring infiltration (N=73), flux infiltration (N=19), positive-pressure lab-based soil-core infiltration (N=31), a sand/silt/clay/bulk density Pedo-transfer function developed by Crosby et al. [1984] (N=21), and tension infiltration (N=12) methods were used to characterize differences in saturated hydraulic conductivity (K_{sat}) of the Troup Soil Series, which is the dominant upland soil series in the northern region of the base. We chose to characterize differences between heavily tracked mechanized training areas that are almost completely devoid of vegetation, being heavily tracked and a forested reference area where no mechanized training has occurred. Any differences between the mechanized training sites and reference site would be proportional to the degree to which military training activities might be represented as a single soil hydrologic parameter. We implemented the different measurement methods to explore the possibility that various techniques might affect the ability of a given measurement method to discern differences between the most severely impacted sites and the relatively pristine reference site. Particular attention was paid to a newly developed in-situ flux infiltration method that applies a variable flux boundary condition in an increasing step-wise manner. In addition, the flux infiltration method was validated using a select number of sample sites by inverse modeling paired with lab-measured soil-water retention curve parameters in Hydrus 2-D [Simunek et al, 1999].

Major differences in estimated K_{sat} means and ranges emerged between measurement methods. Here, we suggest reasons for the observed differences related to their ability to assess the degree of mechanized training disturbance.

MATERIALS AND METHODS

Site Description and Sampling

Fort Benning, located in SE Georgia, is a multi-faceted active military installation that is comprised of natural forest, military managed logging, a variety of sensitive wildlife, recreational hunting, as well as military facilities, housing, and diverse training areas. The natural resources of Fort Benning are regionally important because they represent a pocket of forests surrounded by agriculture and urbanization. Anthropogenic alterations of this forested system create an unfavorable shift in overall ecosystem balance and threaten its value as a regional natural resource as well as a national military training installation. The 73,533-ha military installation is located approximately 75 miles S/SW of Atlanta GA and has been in operation since 1918. The Fort Benning Military installation, home of the US Infantry, is located within the Upatoi Creek Watershed on the Coastal Plain Piedmont Fall Line in southwest Georgia. For a full site description see Dillustro et al. 2002. The topography of much of the base is small rolling hills with level ridgetops and gentle to moderately steep slopes. In the pristine (non-training) ridgetop areas, the surface soil horizons are commonly well-drained loamy-sands with a yellowish brown color (10YR 6/4, 10YR 5/6), which are typical of the Troup and Lakeland soil series that are prevalent at the base (USDA, 1997). Vegetation on the ridgetops and upper elevations within the catchments is primarily scrub oak and longleaf pine with smaller understory shrubs and grasses. The low-lying (riparian/near-stream) regions of the watersheds support mostly oak species with thick understory. Soils in these areas are a poorly drained, black to grayish-brown sandy loam (USDA, 1997). While the USDA soil survey classifies ridgetop soils at our study sites as Troup, remaining soils in mechanized training areas no longer resemble the pedogenesis described therein. Visual evidence of severe erosion is common in the mechanized training areas. Any organic material associated with the A horizon has eroded away and light yellowish brown (10Y hue) and even redder hues (10R or 2.5 YR) that may be more closely associated with the E and B horizons have resulted from continuous mechanized training. Also, sediment channels originate from the ridgetop training areas and eroded sediments are deposited at or near streams and swales.

We chose sampling locations in Troup soils within the Bonham-1 sub-watershed (91 ha) as our reference sites, as the sub-watershed has had no historic mechanized or tracked training (though periodic controlled burning of the understory vegetation occurs every three years). Sampling locations in the Bonham-1 sub-watershed were generally chosen using a systematic sampling pattern of sampling along transects running orthogonal to the stream, with sampling locations at every 10 m contour interval.

The Troup Soil Series representing mechanized training sites were located in the Bonham-2 sub-watershed, which is approximately 250 ha and is directly adjacent to Bonham-1. Sampling efforts in the Bonham-2 sub-watershed were concentrated on two intensively utilized mechanized training sites: Rowan Hill and Cannons. Sampling on Rowan Hill was conducted in transects radiating out from a central location on top of the hill, while random sampling was done at Cannons.

In-situ K_{sat} Estimation Methods

Single-Ring Infiltrometer

A calibrated 4-liter Mariotte-style carboy reservoir was used to maintain a constant pressure head in an 8-cm ID PVC ring that was driven into the soil to a depth of approximately 7-cm [Dane et al., 2002; Elrick and Reynolds, 1992]. Pressure heads of 7-9 cm were consistently maintained and flow volumes were monitored with time in 73 Troup Soils until a steady state flow rate was achieved. Typical measurement durations ranged from 15 to 45 minutes. The solution for 3-dimensional flow through a pressure infiltrometer, developed by Elrick and Reynolds [1992], was implemented. This solution utilizes a shape factor related to the bulbous shape and soil-water retention curve fitting parameters that describe the unsaturated fringe of the wetting front. The bulb-flow approach consistently reduced all measurement estimates by approximately one-third from a 1-D Darcy's Law approximation. This measurement technique was chosen because it is relatively simple to perform in the field, but it was observed that the surface soil structural integrity was disturbed to a certain degree as the PVC ring was driven into the soil with the mallet.

Flux Infiltrometer

The flux infiltrometer consists of: 1) a PVC pipe frame approximately 2 meters tall, 2) a plexi-glass shelf that slides up and down the PVC frame while holding a 4-liter Mariotte-style carboy bottle that acts as a reservoir, 3) a second plexi-glass static shelf near the base that serves to distribute the $^{1}/_{32}$-inch tygon capillary tubes evenly over the measurement area, and 4) a PVC manifold that distributes the water from the reservoir to the 19 capillary tubes that are inserted into holes in the plexi-glass base shelf at a hexagonal 1 $^{1}/_{8}$-inch spacing (Figure 1). Capillary tubes were used to create a lower flow rate range by increased flow resistance according to Poiseuille's Law of laminar flow through a uniform straight pipe. Water from the reservoir at the desired pressure head flowed through the tubes at a rate proportional to the pressure head difference and was calibrated in the lab. The Delta-T® TH_2O soil moisture meter was simultaneously used to monitor volumetric water content with time. Many investigators have developed and used analytical solutions for soil-water storage and soil hydraulic parameters (including K_{sat}) with a constant flux infiltration boundary condition at the lab and field scales [Clothier et al., 1981; Warrick and Hussen, 1993; Cheng Si et al., 2000; Inoue et al., 2000; and Si and Kachanoski, 2000]. Our intent with the flux infiltrometer measurement technique was not to implement an analytical solution for constant flux conditions, but instead to estimate the K_{sat} at a field sampling location by a step-wise increasing variable flux infiltration boundary condition.

The principle measurement protocol for estimating the K_{sat} was to maintain a flow rate less than K_{sat} until a steady state volumetric water content condition was achieved. After reaching steady state, the flow rate was slightly increased and volumetric water content was again monitored until the water content reached a new higher steady state value. Incremental increases in steady state flow conditions continued until surface ponding occurred, which was defined as water visibly "bulging" on the surface, and/or runs off. Once the ponding condition was met, water flux was completely stopped and a drainage period was monitored. K_{sat} was assumed to be halfway between the flow rate at which ponding occurs and the previous flow rate. For the bulk of the sampling sites, the possible error associated with this protocol was within +/- 3 cm/hr. A secondary measurement protocol was performed at flow rates greater than the saturated conductivity. Empirically derived formulas from Mein and Larson [1971] gave the basis for this approach, relating time to ponding, to K_{sat}, infiltration rate, and sorptivity.

This measurement method was selected because the soil was not disturbed and any soil structural features were preserved. 20 total K_{sat} measurements were made in Troup Soils. Since this apparatus was designed and constructed specifically for this field application, inverse modeling was done to optimize K_{sat} in the 2-D Richards' equation using the Hydrus 2-D software [Simunek et al., 1999].

Tension Infiltrometer

Two tension infiltrometers, designed and manufactured by Soil Measurement Systems® were used at each location to create a pair of measurements composed of two different tensions and two different flow rates. The appeal of this measurement method was that other investigators have found it to have been a formidable means of estimating soil hydraulic and solute transport properties within the framework of various analytical solutions [Lee et al., 1983; White et al., 1992; Clothier et al., 1992; Elrick and Reynolds, 1992; Logsdon and Jaynes, 1993; and Cook and Broeren, 1994;]. We implemented a K_{sat} estimate technique that was derived from Wooding's analysis of steady-state infiltration [Wooding 1968] and Gardner's expression of soil-water retention [Gardner 1958]. The tension infiltration method is not expressly designed to directly measure the K_{sat}, but a system of two equations modified by truncating the soil-water retention curve at the site-estimated air-entry pressure was assumed to be an appropriate technique for K_{sat} estimation. This measurement method as also selected because of its non-invasive nature, as the infiltrometer simply rests on the surface of the soil. Although only 12 total measurements were made in Troup Soils, it is an important contribution to the overall comparison of means and ranges of different measurement methods.

Lab-based or Indirect K_{sat} Estimation Methods

Soil-core Ponded Infiltration

Standard brass cores, taken with the core sampler (model 0200) from Soil Moisture Corp ©, were used to sample the top 0-3 cm depth of soil. Once the samples were back in the lab, the accepted measurement protocol suggested by Dane et al [2000] was followed. This measurement method was anticipated to produce results similar to those of the single ring infiltrometer. Soil disturbance during collection and transport of the sample was thought to be as severe or more severe than that by the single ring infiltrometer. The brass core contains soil that has been entirely altered during the collection procedure, whereas the single ring method may disturb the top 7 cm of soil but unaltered soil beneath is also incorporated into the measurement.

Rosetta Lite Version 1.0

Particle size distribution by dry sieve method was completed as well as bulk density for 25 Troup Soils. Rosetta Lite Version 1.0 software was implemented to estimate the K_{sat} from a neural network pedo-transfer function (PTF) [Crosby et al., 1984; Yates et al., 1992; Schaap and Leij, 1998; and Schaap et al., 1998]. Input parameters for this neural network analysis were sand, silt, and clay fractions and bulk density. Rosetta Lite Version 1.0 uses a calibration data set containing 2134 samples for soil-water retention curves and 1306 K_{sat} estimates derived from several different measurement methods [Schaap and Leij, 1998b]. Agricultural and non-agricultural soils in temperate climate zones of the northern hemisphere, mainly from the USA and Europe, constitute the sources of the calibration data set.

RESULTS

Method Variability

The mean and range of K_{sat}, measured at both mechanized training sites and the reference site, varied between methods. A noticeable difference in mean K_{sat} values was observed between the non-invasive measurement methods (flux and tension infiltration) and the invasive methods (single ring infiltration, soil-core infiltration, and PTF estimate) (Table 1). Even though it is difficult to conclude anything specific about the shape of the distribution of the flux or tension infiltration measurement method, the standard error in the mean of both methods suggest that we can glean enough information about the mean K_{sat} estimate to conclude that there is a marked difference from the other three methods.

A similarity was observed in mean K_{sat} estimates by the soil-core and PTF methods. This was expected, since the PTF method used the soil textural percentages and bulk density derived from the same soil core that the soil-core infiltration method used and also that the soil separate fractions consisted almost entirely of sand (>95% sand on average). The mean K_{sat} value for the soil-core and PTF methods was 28.5 cm/hr, which was significantly different from the single ring infiltration method mean estimate of 20.4 cm/hr (p=0.0018 from a one-tailed t-test at the 95% confidence interval). While the mean K_{sat} estimates derived from the single ring infiltration method and the soil-core and PTF methods were statistically different, they are significantly higher than the flux and tension infiltration method mean K_{sat} estimates.

Land-use Variability

To characterize the K_{sat} variability of the reference and mechanized training sites, samples for each measurement method were sub-divided into training and non-training groups (Figure 2). Although the soil-core and PTF methods had a statistically similar mean K_{sat} estimate associated with their entire populations, the PTF estimates from training and non-training sites are significantly different (p=0.016 from a one-tailed t-test at the 95% confidence interval) with the training samples having a higher mean. The mean K_{sat} for training and non-training sites from the soil-core method are not statistically different.

Grouped training and non-training samples for the single ring infiltration method had statistically different mean K_{sat} values, however the training group had a lower mean value relative to the non-training group. It is worth noting that the variance between training and non-training groups was also statistically different, as demonstrated by an F-test with a p value of 0.000559 at the 95% confidence interval.

The flux and tension infiltration mean K_{sat} values for training and non-training groups were not statistically different. As shown by the higher standard error in the mean for the tension infiltration method, more resolution could have been achieved had the sample count been higher. Nonetheless, the tension infiltration method serves to support the mean and range of measurements derived from the flux infiltration method.

DISCUSSION OF METHODS

When evaluating the results for all measurements within the training and reference sites by method, we suggest that there is an important distinction between the mean K_{sat} estimates derived from the invasive single ring, soil-core, and PTF methods and the non-invasive flux and tension methods within the Troup Soil Series. The more invasive methods yield higher K_{sat} estimates than the non-invasive method estimates. Also, the mean K_{sat} estimated from the invasive methods seem to slightly over predict what other investigators had found [USDA, 1997; and Dane et al., 1983]. Not only is there a difference between invasive and non-invasive mean K_{sat} estimates, but also that there is a gradient of K_{sat} means by measurement method. The soil-core and PTF methods yield the highest mean K_{sat} estimate, followed by the single ring method, and finally the flux and tension methods yield the lowest mean K_{sat} estimate (Figure 2). We hypothesize that the gradation of mean K_{sat} estimates stems from measurement/estimation artifacts associated with each measurement method and that different degrees of disturbance to the soil structure results in different sensitivities to soil surface features commonly associated with vegetative and biological processes.

The soil-core method was suspected to have the highest degree of disturbance and artifacts associated with the K_{sat} estimate. During collection of the soil-cores in the field, there was assumed to be various degrees of soil structural alteration. Along with physical alteration of the soil structure, it was also thought that any effects from surface features, such as organics, waxes, or other hydrophobic components found naturally in forest soils, were also attenuated and possibly permanently destroyed during the sampling protocol. Also, while extensive measures to preserve their original characteristics were taken, handling and transportation of the samples may have also contributed to a degree of physical alteration. The affect of these artifacts was conceived as preferential or sidewall-flow during K_{sat} measurement, yielding a higher mean.

Collection or preparation of physical samples used to characterize K_{sat} by the PTF was not entirely relevant, because the sample structure is obliterated to determine the soil separate fractions and bulk density. The suitability of the PTF itself was the principle issue related to its reliability. As previously stated, Different K_{sat} measurement methods made by different investigators to relate to the particle size distribution and bulk density made up the calibration data set of the PTF. Also, the organic matter content and land management practices, such as prescribed burning and tracked mechanized training, were not explicitly considered in the PTF. We considered the PTF and soil-core methods as complimentary K_{sat} estimates, since they are both derived from the same soil sample.

The single ring infiltration method protocol required that the PVC ring be driven into the soil at a specified depth and we suggest that, during this preparation step, the surface soil structure and possibly other physical properties were altered to various unknown degrees. This alteration may be more apparent in the non-training sites where organic matter, soil structure, and surface features would be more important. This level of disturbance is arguably less than that of the soil-core method, but there is still a higher level of disturbance than the flux and tension infiltration methods.

The flux infiltration method yielded a mean K_{sat} estimate significantly lower that the soil-core, PTF, and single ring infiltration methods. This method does not disturb, alter, or impact the soil structure during the measurement process and is considered to have the fewest measurement artifacts associated with it. Because of the non-invasive measurement protocol, we propose that this method is the most sensitive to surface features, organic matter influence, and soil structure.

Similar to the flux infiltration method, the non-invasive measurement protocol of the tension infiltration method eliminated any measurement artifacts associated with soil structural disturbance. It was anticipated that the tension infiltration method would yield a similar mean K_{sat} estimate to that of the flux infiltration method, which was confirmed by a one-tailed t-test that could not reject the hypothesis that the sample means were equal.

In addition to measurement artifacts related to the different K_{sat} estimation methods in this study, the support volume associated with each measurement method was evaluated to identify whether or not different scales of measurement were partially responsible for the different ranges of K_{sat} estimates. There was no evidence that the differences in support volumes contributed to the discrepancy in K_{sat} estimates between the non-invasive methods and the invasive methods. The range of K_{sat} estimates from the invasive methods were very similar, but the support volume associated with the single ring infiltration method is 6 to 30 times larger than that of the soil-core and PTF (Table 3). If the differences between the non-invasive methods and the invasive methods were attributed to the support volume, then we would expect two or more orders of magnitude difference in the support volume of the non-invasive methods, which was not observed. Although this is not definitive proof that the support volumes are not in some way relevant to the K_{sat} range of the measurement methods, for our purposes it demonstrates that the principal cause responsible for different K_{sat} estimates by method remains the degree of soil structural disturbance during measurement.

DISCUSSION OF VARIABILITY

Given that our study consisted of a relatively pristine non-training reference site and a heavily tracked mechanized training site, we gauged each measurement method's relative ability to detect differences between the two different land-uses. We propose that the sensitivity of the measurement method to surface features was the principal reason that differences between the training and non-training sites were or were not observed. Since the soil in the training areas is tracked, churned, homogenized, and devoid of organic matter, it was hypothesized that all of the measurement methods would yield similar K_{sat} estimates for these sites. This was true, with the exception of the soil-core and PTF, which are described as being high-disturbance methods, and by the same token less sensitive to soil structural effects. While there were no observed surface features in the training areas, as they are a product of vegetative and biological processes, the level of disturbance associated with the sample collection masked the lower hydraulic conductivity character. In the non-training areas, the soil-core estimated a mean K_{sat} comparable to the single ring infiltration and PTF methods. The degree of disturbance associated with the sampling protocol is suggested to be the primary reason that the K_{sat} range is higher than that of the flux and tension infiltration methods. Thus, the sensitivity of the soil-core method to characterize the hydraulic variability is virtually absent.

The mean K_{sat} in training areas for the PTF method was higher than the mean non-training K_{sat}. From the frequency distribution of K_{sat} estimates from the PTF method it was observed that approximately 50% of the samples fell into the 35 cm/hr bin, which shifted the mean. A certain level of soil homogeneity in the training areas with respect to the sand, silt, and clay content and bulk density at the 0-3 cm depth is suggested by the PTF frequency distribution, however the magnitude of the mean K_{sat} was observed to be exaggerated compared to the other methods. As previously mentioned, in non-training areas, the PTF estimated K_{sat} values similar to the single ring and soil-core methods. There is some sensitivity to detect soil hydraulic variability within these two land-uses, but due to suspect K_{sat} estimates in training areas, re-assessment of the appropriateness of this method is needed.

The lower mean K_{sat} in training areas derived from the single ring method contrasted the higher mean K_{sat} observed at the non-training sites. The lower mean K_{sat} value in the training areas coincides with the premise that the training and non-training sites are hydrologically different and more susceptible to runoff. The resulting mean estimate is similar in magnitude to that of the flux and tension methods. In non-training areas, the mean K_{sat} estimate was similar to the soil-core and PTF estimates. An explanation for this is that the degree of disturbance associated with the single ring infiltration method altered the soil surface structure so that the influence of surface features would be attenuated or void. It is therefore logical to suggest that the K_{sat} values derived from the single ring method are averaging the E and A soil horizons. It is important to emphasize that the support volume of the single ring method is still only secondarily responsible for the higher K_{sat} values in the non-training areas as the primary reason for the averaging of soil horizons is that the measurement protocol disturbed the soil structure and surface features.

The mean K_{sat} estimates from flux and tension infiltration methods were statistically similar at both training and non-training sites. Again, at training sites, the flux and tension methods estimated a mean K_{sat} similar to that of the single ring infiltration method. The training sites are devoid of vegetation, organic matter, and soil structure (homogenized), so the similarity was not surprising as the methods should all yield the same number given a reasonably homogeneous soil without surface features. When surface features are present, such as in non-training areas, the flux and tension mean K_{sat} values are significantly lower than the non-training estimates using any of the other methods. This is because of the sensitivity of the methods, as the soil structure and surface features were not altered in any way upon measurement and those factors produced a lower mean K_{sat}.

CONCLUSIONS

A gradient of disturbance and resulting mean Ksat estimates for the five measurement methods is observed. The more the measurement method protocol alters or disturbs the soil structure, the higher the resulting K_{sat} range will be and the less chance any surface features will be measured. We proposed that the soil-core and PTF methods represented the highest degree of soil disturbance. Furthermore, the single ring infiltration method was classified as an intermediate level of disturbance, while the flux and tension infiltration methods represented the least amount of soil disturbance. The highest mean K_{sat} values resulted from the soil-core and PTF methods, followed by the single ring method and the lowest mean was measured with the flux and tension methods.

The sensitivity of each measurement method to detect the hydraulic variability between the two extreme land-uses was dependent upon the degree and type of soil disturbance required by the measurement protocol. We suggest that the soil-core method did not distinguish a difference between the non-training and the training sites due to its high level of soil disturbance. The PTF was suspect in its relative magnitude of mean K_{sat} for the training and non-training sites, calling into question the appropriateness of its use in this application. Significant sensitivity to decipher the hydraulic variability of the two sites was observed with the single ring method. Due to the degree of disturbance during measurement, K_{sat} estimates in non-training areas did not compare well with the non-training K_{sat} estimates resulting from the flux and tension methods, which were significantly lower. Soil structure and surface features in non-training areas influenced the measured K_{sat} using the flux and tension infiltration methods.

REFERENCES

Dane, J.H., D.K. Cassel, J.M. Davidson, W.L. Pollans, and V.L. Quisenberry. 1983. Physical characterization of soils of the southern region – Troup and Lakeland Series. Alabama, Soil Coop. Series Bull. 262. Auburn University.

Dane, J.H. and G.C. Topp. 2002. Methods of soil analysis part 4: physical methods. Soil Science Society of America, Inc. Madison, WI, USA. 812-826.

Clothier, B.E., M.B. Kirkham and J.E. McLean. 1992. In situ measurements of the effective transport volume for solute moving through soil. Soil Science Society of America Journal. 56: 733-736.

Cook, F.J., and A. Broeren. 1994. Six methods for determining sorptivity and hydraulic conductivity with disc permeameters. Soil Science Society of America Journal. 157(1): 2-11.

Crosby, B.J., G.M. Hornberger, R.B. Clapp, and T.R. Ginn. 1984. A statistical exploration of the relationships of soil moisture characteristics to the physical properties of soil. Water Res. Res. 20(6): 682-690.

Dale, V.H., S.C. Beyeler, and B. Jackson. 2002. Understory vegetation indicators of anthropogenic disturbance in longleaf pine forests at Fort Benning, GA, USA. Ecological Indicators 1(3): 155-170.

Dilustro, J. J., B. S. Collins, L. K. Duncan, and R. R. Sharitz. 2002. Soil Texture, Land-use, and Vegetation of Fort Benning Upland Forest Sites. J. of the Torrey Botanical Soc. 129(4): 289-297.

Doerr, S.H., R.A. Shakesby, and R.P.D Walsh. 1998. Spatial variability of soil hydrophobicity in fire-prone eucalyptus and pine forests, Portugal. Soil Science 163(4): 313-324.

Elrick, D.E. and W.D. Reynolds. 1992. Infiltration of constant-head well permeameters and infiltrometers. In (G.C. Topp, W.D. Reynolds and R.E. Greed, eds.) Advances in Measurement of Soil Physical Properties: Bringing Theory in to Practice. Soil Science Society of America Special Publication Number 30. Madison: 1-24.

Fang, S., S. Wente, G. Z. Gertner, G. Wang, and A. Anderson. 2002. Uncertainty Analysis of Predicted Disturbance from Off-Road Vehicular Traffic in Complex Landscapes. Environ. Management 30(2): 199-208.

Fuchs, E.H., M.K. Wood, T.L. Jones, B. Racher. 2003. Impacts of tracked vehicles on sediment from a desert soil. Journal of Range Management 56(4): 342-352

Garten, C.T., Jr., T.L. Ashwood, and V.H. Dale. 2003. Effect of military training on indicators of soil quality at Fort Benning, Georgia. Ecological Indicators 3: 171-179.

Grantham, W. P., E. F. Redente, C. F. Bagley, and M. W. Paschke. 2001. Tracked vehicle impacts to vegetation structure and soil erodibility. J. Range Management 54(6): 711-716.

Gardner, W.R. 1958. Some steady-state solutions of the unsaturated moisture flow equation with application to evaporation from a water table. Soil Sci. 85:228-232.

Halvorson, J.J., L.W. Gatto, and D.K. McCool. 2003. Overwinter changes to near-surface bulk density, penetration resistance and infiltration rates in compacted soil. Journal of Terramechanics 40(1):24-30.

Huffman, E.L., L.H. MacDonald, and J.D. Stednick. 2001. Strength and persistence of fire-induced soil hydrophobicity under ponderosa and lodgepole pine, Colorado Front Range. Hydrologic Processes 15(15): 2877-2892.

Kade, A., and S.D. Warren. 2002. Soil and plant recovery after historic military disturbances in the Sonoran Desert, USA. Arid Land Resources Management 16(3): 231-243.

Lee, D.M., W.D. Reynolds, D.E. Elrick, and B.E. Clothier. 1985. A comparison of three field methods for measuring saturated hydraulic conductivity. Soil Science Society of America Journal. 65: 563-573.

Logsdon, S.D. and D.B. Jaynes. 1993. Methodology for determining hydraulic conductivity with tension infiltrometers. Soil Science Society of America Journal. 57: 1426-1431.

Perkins, D.B., N.W. Haws, B. Das, P.S.C. Rao, and J.W. Jawitz. Submitted 2003. Mechanized Military Training Impacts on Hydraulic Characteristics of Ridgetop Soils in a Forested Watershed. Journal of Env. Quality.

Perkins, D.B. 2003. Spatial and temporal characterization of soil-hydraulic properties and soil-water storage dynamics in forested watersheds. M.S. Thesis, Department of Agronomy, Purdue University, West Lafayette, IN, 47907

Prosser, I.P., L. Williams. 1998. The effect of wildfire on runoff and erosion in native Eucalyptus forest. Hydrologic Processes 12 (2): 251-265.

Prosser, C.W., K.K. Sedivec, and W.T. Barker. 2000. Tracked vehicle effects on vegetation and soil characteristics. J. Range Management 53(6): 666-670.

Quist, M.C., P.A. Fay, C.S. Guy, A.K. Knapp, and B.N. Rubenstein. 2003. Military training effects on terrestrial and aquatic communities on a grassland military installation. Ecological Applications 13(2): 432-442.

Robichaud, P.R., R.D. Hungerford. 2000. Water repellency by laboratory burning of four northern Rocky Mountain forest soils. Journal of Hydrology Sp. Iss. 231: 207-219.

Schaap, M.G., Leij, F.J. and M.Th. van Genuchten. 1998. Neural network analysis for hierarchical prediction of soil water retention and saturated hydraulic conductivity. Soil Sci. Soc. Am. J. 32: 847-855.

Schaap, M.G., and F.J. Leij. 1998. Using Neural Networks to predict soil water retention and soil hydraulic conductivity. Soil and Tillage Research 47:37-42.

Schaap, M.G., and F.J. Leij. 1998b. Database Related Accuracy and Uncertainty of Pedotransfer Functions. Soil Science 163:765-779.

Simunek, J., Sejna, M., and Th. Van Genuchten. 1999. Hydrus-2D /Meshgen-2D. Simulating water flow and solute transport in two-dimensional variably saturated media. IGWMC-TPS 53C version 2.0.

Trumbull, V.L., P.C. Dubois, R.J. Brozka, and R. Guyette. 1994. Military camping impacts on vegetation and soils of the Ozark Plateau. J. Environ. Management 49(4): 329-339.

United States Department of Agriculture (USDA). 1997. Soil Survey of Chattahoochee and Marion Counties, GA. Natural Resource Conservation Service.

Wanner, M. and W.E.R. Xylander. 2003. Transient fires useful for habitat-management do not affect soil microfauna (testate amoebae) – a study on an active military training area in eastern Germany. Ecological Engineering 20(2): 113-119.

White, I., M.J. Sully. 1992. Measurement of surface-soil hydraulic properties: Disk permeameters, tension infiltrometers, and other techniques. In (G.C. Topp, W.D. Reynolds, and R.E. Green, eds.) Advances in Measurement of Soil Physical Properties: Bringing Theory in to Practice. Soil Science Society of America Special Publication Number 30.

Whitecotton, R.C.A., M.B. David, R.G. Darmody, and D.L. Price. 2000. Impact of foot traffic from military training on soil and vegetation properties. J. Environ. Management 26(6): 697-706

Wooding, R.A. 1968. Steady infiltration from a shallow circular pond. Water Resour. Res. 4:1259-1273.

Yates, S.R., M.Th. van Genuchten, A.W. Warrick, and F.J. Leij. 1992. Analysis of measured, predicted, and estimated hydraulic conductivity using the RETC program. Soil Science Society of America Journal. 56:347-354.

Table 1. Descriptive statistics of K_{sat} estimates for the five measurement methods in the Troup Soil Series.

	Flux	Tension	Single ring	Soil-core	PTF
Mean cm/hr	**9.5**	**10.0**	**20.4**	**28.3**	**28.75**
Median cm/hr	9.7	7.3	19.1	25.25	27.3
Standard error	2.3	4.2	1.9	3.78	3.37
Standard deviation	6.8	10.1	11.7	14.89	10.76
CV (%)	72.4	NA	59.3	53.6	39.65
N	20	12	73	31	21

Table 2. Input parameters and optimized K_{sat} inversely modeled with Hydrus-2-D modeling of four flux infiltration measurement sites. Three sites were located in the reference area (ID numbers: 68, 69, and 71) and the other (ROW-2) was located in a mechanized training area. This table also includes the field-measured K_{sat} estimates from the flux infiltration method.

	Input Parameters						
	α	n	α_{hys}	θ_s	θ_r	Hydrus-modeled K_{sat}	Flux infiltration estimate of K_{sat}
69	0.059	2.06	0.107	0.273	0.042	9 cm/hr	11 cm/hr ± 2
71	0.042	2.29	0.099	0.396	0.100	9 cm/hr	7 cm/hr ± 1
68	0.040	2.35	N/A	0.356	0.070	16 cm/hr	22 cm/hr ± 4
ROW2	0.050	3.64	0.150	0.257	0.094	18 cm/hr	11 cm/hr ± 2

Table 3. Estimated or range of support volumes for Troup Soils in categories of invasive and non-invasive measurement methods, where the flux and tension infiltration measurement methods are categorized as the *non-invasive* methods and the single ring infiltration, ponded soil-core, and PTF methods are categorized as *invasive*.

Non-invasive Measurement Method	**Support Volume cm^3**
Flux infiltration	120
Tension infiltration	10-150
Invasive Measurement Method	**Support Volume cm^3**
Single ring infiltration	400-1800
Soil-core lab-based infiltration	60
Rosetta – PTF	60

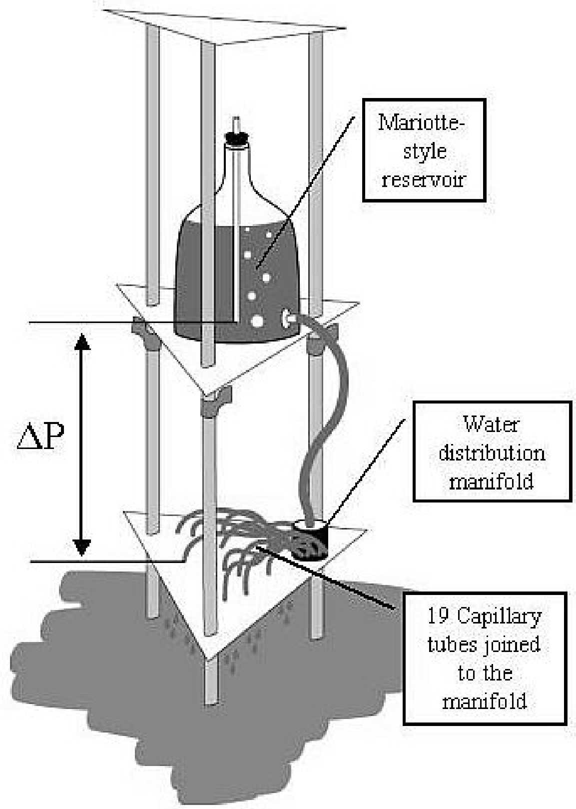


Figure 1. Schematic drawing of the flux infiltrometer used to measure K_{sat}. An adjustable-height Mariotte-style reservoir supplies water to 19 capillary tubes that deliver a specific flow rate of water to the soil.

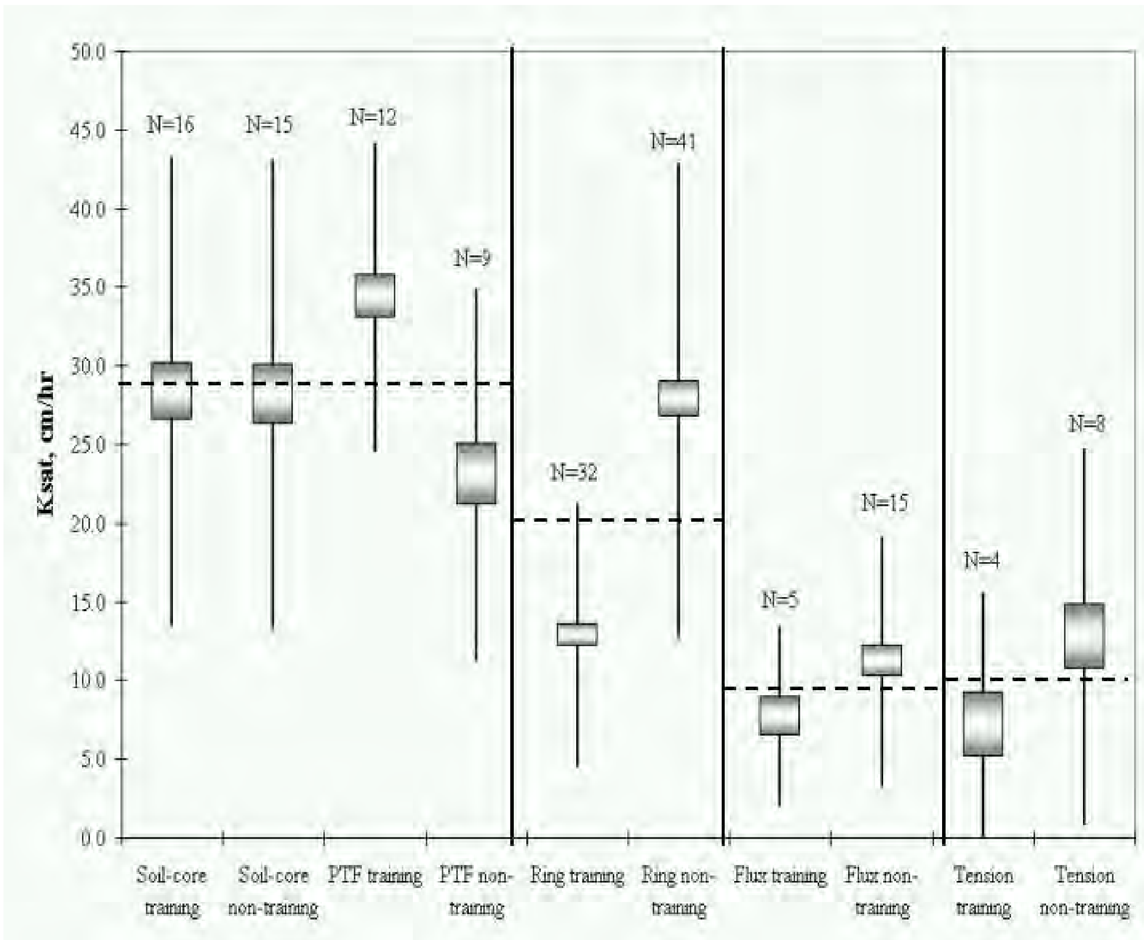


Figure 2. K_{sat} distribution statistics of land-use groups for each measurement method. The box represents the standard error in the mean, where the center of the box in the calculated mean K_{sat}. The whiskers represent the standard deviation from the mean. The solid vertical lines divide each measurement method (where the soil-core and PTF methods are considered as one population) while the dashed lines represent the combined or population mean K_{sat} for each method.

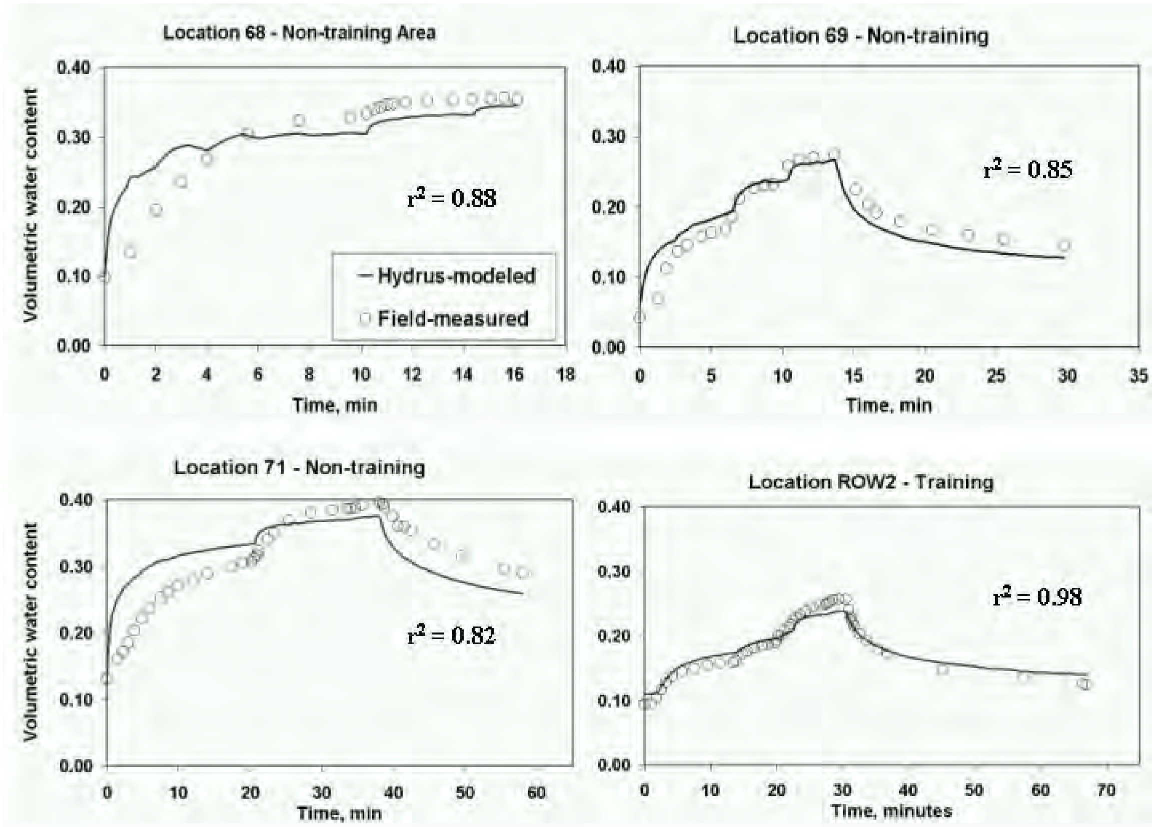


Figure X. Inverse modeling flux infiltration using Hydrus 2-D at select sample locations in Troup Soils. Three site ID's (68,69,and 71) are from the reference site and one site ID (ROW-2) is from a mechanized training site. K_{sat} was optimized using observed volumetric water content versus time and site-specific soil-water retention curve fitting parameters.

3.3.4
Mechanized Military Training Impacts on Hydraulic Characteristics of Ridgetop Soils in a Forested Watershed. D.B. Perkins, N. W. Haws, B. S. Das, P.S.C. Rao, and J.W. Jawitz

ABSTRACT

Soil hydraulic properties were characterized on a ridgetop training site and other sites not impacted by training at Fort Benning, a military installation in southwestern, GA to 1.) examine the influence of mechanized training on soil properties, and 2.) evaluate the utility of soil properties as disturbance-level indicators. The soil properties for the training site were then compared to those of the non-training sites on the base, and several agricultural sites located outside the base, yet with similar soil types. The assessment of mechanized military training impacts on to ridgetop soils shows a reduction in the mean pore size, as indicated by smaller scaling factors, an increase in the soil bulk density, and a decrease in the infiltration rates for training versus non-training locations. Measured soil properties in the non-training areas also show a greater variability than the data for the training sites. The more clustered distributions in the training sites are indicative of the repetitive training cycles that progressively mix and homogenize the soil. These differences in specific properties, however, do not explain the visual evidence of increased overland flow and significant soil erosion in the training areas. The observed increase in runoff and erosion is most likely a result of the combined effects of loss of all ground cover, intensified raindrop impact, surface sealing, and loss of spatial variability. More work is needed to establish training cycle threshold levels for the measured soil properties and ecosystem integrity. It is suggested that ridgetop impact be quickly assessed using visual or remote sensing techniques and that greater research emphasis be given to characterizing the indirect, yet more widespread, down-slope impacts of mechanized training activities localized on the ridgetops.

Introduction

Military training activities, such as the building of roads for movement of heavy military vehicles, construction of specialized military facilities, and the removal of vegetative cover, often lead to severe land degradation causing increased sediment loading (Fang et al., 2002) to streams and unfavorable shifts in the ecological balance. Some of the most severe impacts are caused from training with mechanized, tracked vehicles. This training is often located on ridgetop areas of watersheds that are more easily accessed and offer a more strategic vantage point than regions lower in a catchment.

Application of appropriate environmental stewardship and mitigation principles to management of military-impacted watersheds requires the knowledge of 1) how different military activities modify existing hydrologic environments, and 2) how the existing ecosystem responds to the impacts of military activities. Several studies have shown that soil physical properties are influenced by the type of land use and degree of disturbance (Trumbull et al., 1994; Garten et al., 2000; Prosser et al., 2000; Whitecotton et al., 2000; Grantham et al., 2001; Dilustro et al., 2002). In a study of the impact of foot traffic from military training on soil and vegetation properties at the United States Air Force Academy, Whitecotton et al. (2000) reported mean soil bulk density values increased from 1.04 g/cm^3 to 1.30 g/cm^3 to 1.37 g/cm^3 for low-, moderate-, and high-use sites, respectively. Other studies report similar increases in bulk density (Prosser et al., 2000; Grantham et al. 2001) and erosion potential (Fang et al., 2002) for military sites with mechanized (tracked vehicle) training operations.

The purpose of this study was to characterize the effects of mechanized training on the soil physical and hydraulic properties on a ridgetop training site at Fort Benning, a military installation in southwestern, GA. Soil-water retention curves (summarized by the similar-media scaling technique), soil bulk density, and steady-state infiltration rate are compared for a number of training and non-training sites on the base and off-base locations. These comparisons were then used to assess impact characteristics and evaluate the utility of the observed differences in soil properties as indicator of the degree of environmental disturbance.

Site Description

The Fort Benning Military installation, home of the US Infantry, is located within the Upatoi Creek Watershed on the Coastal Plain Piedmont Fall Line in south-west Georgia (Figure 1). Since operations began in 1918, the installation has grown to 73,533 ha (Dilustro et al., 2002). The installation topography is comprised of level ridgetops and gentle to moderately steep slopes. The low-lying (riparian) regions of the watershed have a canopy of pine and oak species with thick understory. Soils in these areas are a poorly drained, black to grayish-brown sandy loam (USDA, 1997). In the pristine (unimpacted) ridgetop areas, the surface soil horizons are commonly loamy-sands with a yellowish brown color (10YR 6/4, 10YR 5/6), which are typical of the Troup and Lakeland soil series that are prevalent at the base (USDA, 1997). Vegetation on the ridgetops is primarily scrub oak and longleaf pine (Dilustro et al., 2002) with smaller understory shrubs and grasses.

Currently, military impacts on the base range from foot traffic to periodic controlled burns to heavy, tracked vehicles as well as an extensive network of unpaved roads. Years of localized mechanized vehicle (tank) training activities at the site have left several ridgetops barren and susceptible to further degradation. Direct impacts include complete loss of vegetation and a mixing of the surface and subsurface soil horizons, manifest by a reddish hue (Figures 2a, 2b) that is characteristic of Troup subsoils (USDA, 1997). Indirect impacts are increased runoff, soil erosion, and the formation of large erosion channels (Figures 2c, 2d). Downslope from the training areas, large sediment deposition areas have formed and represent at least one of the secondary, more dispersed impacts of the localized training areas (Figures 3a - 3c).

Soil Core Sampling and Laboratory Protocols

Undisturbed soil cores were collected in several sub-watersheds at Fort Benning. Thirty-three cores were collected from non-training sites and 15 cores were collected from training areas. Thirty-three samples were collected from non-training sites and 15 samples were collected from training areas. Sampling of non-training sites (i.e. sites with little or no mechanized training) was conducted in the Bonham-1 (B-1), Bonham-2 (B-2) and several Sally-Branch (SB-1 to SB-4) subwatersheds (see Figure 3). Bonham-1, the most extensively sampled area, encompasses 91 ha and is relatively pristine, having little or no heavy equipment training (though periodic controlled burning of the understory vegetation occurs). Bonham-2 is approximately 250 ha and lays directly adjacent to Bonham-1. Bonham-2 also consists of relatively undisturbed areas with the notable exception of Rowan Hill, where intensive tank training occurs. Sampling locations in on B-1 and B-2 were generally chosen using a systematic sampling pattern of sampling along transects running orthogonal to the stream, with sampling sites located at about every 10 m contour interval. This was done to represent hydrologic, slope, and other gradients in the watershed. The B-1 watershed included several parallel transects, whereas B-2 only included a single transect that bisected the watershed.

Finally, soil cores representing non-training areas were collected at a limited number of locations in two subwatersheds (approximately 80 ha each) within the Sally Branch watershed, located adjacent to the Bonham watershed. Sally Branch is a moderately disturbed site, with several areas of recent controlled burns, but little mechanized training activities; otherwise this watershed and shares similar characteristics as Bonham.

Soil cores for training areas were mainly collected on radial transects originating near the center of Rowan Hill, a ridgetop located on the north-eastern boundary of Bonham-2. Frequent, episodic tank training on the Rowan Hill ridgetop has eroded a significant portion of surface soil, exposing the Bt2 horizon (sandy clay loam) in some places. Natural vegetation is completely absent from the central ridgetop where the soil has been recurrently churned and compacted. A few cores from the training areas were also collected in the southwest section of Bonham-2 at sites with similar impacts, yet smaller in spatial extent.

At each sampling location, two undisturbed soil cores (5.7 cm diameter, 60 cm^3 volume) were collected from the top 6 cm depth (one from 0-3 cm and one from 3-6 cm) using Core Sampler (Model 0200, Soil Moisture Corp., Goleta, CA). Most of the soils collected during sampling were identified as Troup series (loamy, siliceous, thermic Grossarenic Kanduidults) in the soil survey of Chattahoochee, GA (USDA, 1997), while others from Lakeland series (thermic, coated Typic Quartzipsamments), with the remainder belonging to the Ailey (loamy, siliceous, thermic Arenic Kanhapludults), Fuquay (loamy, siliceous, thermic Arenic Pliinthic Kanduidults), and Cowarts (fine-loamy, siliceous, thermic Typic Kanhapludults) series. All of these soil series have a similar loamy sand texture in the surface horizon and are primarily distinguished only in the depth to the diagnostic subsurface horizon (USDA, 1997).

Soil water retention characteristics (SWRCs) of the soil cores were measured using the TEMPE cell method (Flint and Flint, 2002). After soil-water retention curves were measured from the soil cores, the soil was extracted and dried at 105 ºC for a minimum of 24 hours. The bulk density of each sample was calculated using the dry weight and known volume of soil contained in the brass soil core.

The resulting volumetric soil-water contents ($\theta(h_i)$) with their associated pressures were then fitted with the water retention function developed by Kosugi, 1994:

$$\theta(h_i) = \theta_r + 0.5\left(\theta_s - \theta_r\right) erfc\left(\frac{\ln(h_i / h_m}{\sigma\sqrt{2}}\right) \tag{1}$$

where θ_s = volumetric water content at saturation; θ_r = volumetric water content at residual saturation; σ is the standard deviation of the soil pore size distribution function; and h_m is the pressure head corresponding to median pore size (r_m) for a given soil sample. The Kosugi water retention model was adopted to facilitate the use of a physically based scaling (Kosugi and Hopmans, 1998) approach for summarizing measued SWRCs. In this scaling approach, the median pore size is assumed as the characteristic length scale and is used to directly estimate the scaling factor (α_i) for water retention curves (Kosugi and Hopmans, 1998):

$$\alpha_i = r_{m,i} / r_{m,*} \quad = h_{m,*} / h_{m,i} \quad \Rightarrow \qquad \ln r_{m,i} = \ln \alpha_i + \ln r_{m,*} \tag{2}$$

where i and * in the subscript corresponds to the ith sample and the reference soil, respectively. For each sample there exists a complete water retention curve. Kosugi and Hopmans (1998) showed that the characteristic length scales for the reference soil may be estimated from the relationships:

$$h_{m,*} = \exp\left[\frac{1}{n}\sum_{i=1}^{n}\ln(h_{m,i})\right] \tag{3}$$

$$\left(\sigma_*\right)^2 = \frac{1}{n}\sum_{i=1}^{n}\sigma_i^2 \tag{4}$$

An advantage of this scaling approach (herein after referred to as Kosugi scaling) over other conventional approaches is that the scaling factors are directly estimated from the measured retention parameters using Eq. (3) and (4). Moreover, Eq. (2) suggests that the scaling factor is directly proportional to the median pore size of a soil because the median pore size for a reference soil water retention curve is estimated to be a single value (Eq. 3). Thus, smaller the scaling factor, smaller is the median pore size.

Kosugi scaling factors were computed for three data combinations. First, scaling factors were computed for using the data for only the 0-3 cm (top) cores. Then, scaling factors were computed using only the data for the 3-6cm (bottom) cores. Finally, scaling factors were derived from the pooled data for the top and bottom cores. No significant differences were found between the scaling combinations, and for consistency, only the scaling results of the bottom cores are shown here.

After SWRCs were measured for the soil cores, the soil was extruded and oven-dried at 105 ºC for a minimum of 24 hours. The soil bulk density of each core was calculated using the dry weight and known volume of soil contained in the brass ring.

In-*situ*, steady-state infiltration rate measurements (I_s) were also measured at most of the soil-core sampling locations using a single-ring, constant-head infiltrometer. The constant pressure-head tests were performed using a 10-cm diameter (9.3-cm I.D.) by 15-cm long PVC ring inserted 7.5 cm into the soil surface. The constant water head was supplied to the ring via a 4-liter Mariotte-style supply reservoir. Flow rates (Q) were measured by monitoring the water levels in the Mariotte reservoir, and the I_s values were estimated using the method described by Elrick and Reynolds (1992), which considers corrections for a 3-dimensional (or "bulb") wetting front underneath the single-ring infiltrometer.

Soil cores from Troup and Lakeland soils collected outside of Fort Benning were also included in the analyses. The off-site cores were collected and analyzed by other researchers in an earlier regional study (Dane et al., 1983), and were gathered from a total of 26 field plots in Alabama (2 plots, peach orchard with grass cover), Florida (9 plots, corn and soybean), North Carolina (12 plots, unknown treatment), and South Carolina (3 plots, unknown treatment). As such, the off-site samples provided basis for additional comparison of land management affects on soil properties. Most of the off-site samples were collected in 7.6 cm high by 7.6 diameter cores. Though these cores were slightly larger than the cores used in the study here, they were assumed to be similar enough so that issues of sample support size did not need to be considered for the water retention and bulk density measurements. However, the infiltration measurements for the off-site data (generally measured from the cores under unit gradient) were not included in the comparative analysis here because of experimental artifacts arising from different measurement methods and measurement support areas (Haws et al., in review).

Results

The water retention characteristics for the off-site and Fort Benning soil core coalesce well to a single reference water retention curve (Figure 4), with an average goodness of fit, R^2, of 0.99 for the scaled retention curve and no less than 0.91 for each individual water retention curve. The scaling results lend credibility to the similar media approach and justify the use of scaling factors as a comparative measure of differences in soil hydraulic properties for the soils examined here. The cumulative distributions (Figure 5a) show the scaling factors of the training cores to be clustered between 0.70 and 0.90, while the scaling factors of the non-training cores are more uniformly distributed from between 0.50 and 2.0. The mean of the non-training cores (1.11) is larger than the training cores (0.83) and the variance of the non-training cores (0.15) is three times as great as the variance of the non-training cores (0.05) (Table 1). The two distributions, the training and non-training scaling factors show significant differences with a confidence level of greater than 99 percent. Regression of scaling factors with slope and elevation (data not shown) revealed no correlation ($R^2 < 0.06$), and a subset of scaling factors for the non-training cores located on the ridgetops appeared to follow a similar distribution as the full set of non-training cores. Therefore, the differences between the training and non-training cores are attributed to the training impacts and not hillslope position.

The off-site data show a bi-modal distribution of scaling factors with the scaling factors computed for the Alabama and Florida sites tightly grouped around 0.72 and the scaling factors for the North Carolina and South Carolina sites ranging from 1.38 to 2.10. The cumulative frequency distributions (Figure 5b) for the soil bulk density values also show distinct differences between the training and non-training cores, while the training and off-site data are more similar. The mean bulk density value for the training cores (1.54 g/cm^3) is greater than the mean of the non-training cores (1.38 g/cm^3). As with the distribution of scaling factors, the soil bulk density values for the training area appear much more clustered than the values for the non-training cites, and the P-value of the one-tailed t-test (0.0008) is significant to the 99 percent confidence level. The bulk density values for the off-site data resemble the trend of the values of the training cores. The mean of the off-site values (1.58 g/ml^3) is only slightly greater than the mean of the training values, and the distribution of off-site values is even narrower.

The narrower distribution and higher soil bulk density values for the training areas may be explained by the fact that the training area sample locations were confined to a single ridgetop, while the non-training cores were collected from various watershed locations; a limited number of bulk density measurements precluded a rigorous evaluation of this possibility. However, the similarities between soil bulk density values for the training cores and off-site field plot samples (which represent managed landscapes) suggest that the distributions of soil bulk density values reflect disturbance and not simply the small spatial extent of the sampling site. In addition, the few training-area cores not taken on Rowan Hill fit well within the range of the Rowan Hill cores.

Discussion and Conclusions

<u>Training versus non-training sites</u>

Smaller scaling factors imply smaller mean pore sizes (Eq. 2) of the training soils compared to the non-training soils. This is likely a result of organic matter losses from removal of much of the surface horizon due to erosion and mixing of the surface soil with the more clayey subsoil during tank training activities. Exposed tree roots and the more reddish hue of the soils on the impacted ridgetops compared to the bleached sands in the unimpacted areas provide visual evidence of these specific impacts.

The higher soil bulk density values and lower infiltration rates of the training versus non-training areas are further indications of the loss of organic matter combined with compaction from repeated tank track. Similar differences in the soil bulk density values between training and non-training sites were also observed in another study at Fort Benning.. Garten et al. (2003) reported comparable mean soil bulk density values of 1.38 g/cm^3 in areas of light military use (infantry training only) and 1.53 $g/\ cm^3$ for areas of tracked vehicle training with no overstory vegetation and approximately 95 percent bare ground. They also found that the mechanized training sites had significantly less particulate organic matter than the lighter use and reference sites, which concurs with the visual observations of loss of surface soils and mixing of the surface and subsurface horizons. The comparable soil bulk density values of the training areas and the off-site field plot cores (which would similarly be subject to periodic tractor compaction) also provide supporting evidence that increased values of the soil bulk density are general indicators of landscape disturbance.

The lower variance and more clustered distributions of the scaling factors, bulk density values, and infiltration rates for the training sites versus non-training sites point to an important effect of a reoccurring landscape disturbance, such as the periodic training cycles that routinely homogenize the ridgetop soils. This impact of training is also conveyed in the similar distributions of soil bulk density values between training and off-site data. Mechanized training would tend to fragment spatial correlation and increasingly homogenize the soil properties. The loss of natural vegetation (trees, understory shrubs, litter cover) in the training areas and planting of row crops in the off-site field plots would accentuate the landscape homogenization of soil properties.

Significance of Differences in Measured Properties

While differences in the soil properties for the training and non-training sites are statistically significant, by themselves, they offer no compelling explanation of the visual signs of large increases in runoff and erosion. In particular, the mean steady-state infiltration rate of the training sites (12.0 cm/hr) is less than half that of the non-training sites (26.8 cm/hr), but it is still greater than the maximum 100-yr, 24-hr rainfall intensity of 10 cm/yr (Georgia Soil and Water Conservation Commission, 2000). However, a simple comparison of a single rainfall intensity with the mean infiltration rate does reveal how combinations of the rainfall and infiltration distributions might result in spatially/temporally variable runoff. Also, the effects of the combined land and soil disturbance features may synergistically increase the runoff and erosion potential. The loss of vegetation and the absence of a litter layer intensify the effect of raindrop impact, leading to increased soil detachment. Bare soil surface also promotes surface sealing, which, in turn results in reduced infiltration rates and a more rapid runoff response (Assouline and Mualem, 2002; Wetzel, 2003). The effects of a surface seal would not be manifest in the infiltration measurements conducted in this study since the insertion of the infiltrometer disturbs any crust formation on the soil surface. Consequently, the actual infiltration rates at the training sites may be much less than the measured values.

Homogenization of the soil properties at the training site also increases the runoff potential. Fiedler et al. (2002) found consistently greater (> 35%) runoff from plots with bare soil than on similar plots that also had vegetation (prairie short-grass). They observed that ponding did occur in the bare parts of plots with patchy vegetation, but that ponded water was infiltrated as it flowed onto the vegetated area, resulting in little effective runoff. Through additional numerical simulations, they suggested that because of this "interactive-infiltration" process, the effective large-scale infiltration rate decreases as spatial variability increases. Other studies have likewise shown the importance of runoff-runon and spatial variability in the effective infiltration of a hillslope or landscape (Merz and Plate, 1997; Nachabe et al., 1997; Corradini et al., 1998).

Soil Properties as a Disturbance Indicator

Although the measured mechanized training impacts on the soil physical properties are enlightening, it must be conceded that these results do little more than revealing obvious effects of the episodic loss of vegetation, soil compaction and mixing, and the progressive erosion of the surface soil and loss of organic matter. No complex indicator is needed to evaluate if a mechanized training area is significantly disturbed as the visual evidence is most compelling. In fact, because vegetation loss and soil homogenization potentially play the principal roles in increased runoff and soil erosion, simple observations of bare ground and mixed soil horizons may be the key indicators of the environmental integrity of mechanized training sites. Furthermore, because the training area sampled in this study was already highly disturbed, the soil physical properties alone do little to elucidate the more fundamental questions of how the disturbance history proceeded, when the ecosystem reached a threshold disturbance level, and how the training impacts might be reversed. Additional work is needed to establish these mid-term indicators of disturbance levels, identify ecosystem thresholds, and relate the degree of the environmental disturbance to changes in the values of the soil properties. Because mechanized training occurs in short, intense intervals, an ecological indicator based on soil properties that is assessed according to the number of training intervals, may be more relevant to establishing indicator level than monitoring the real-time evolution of the properties. In addition, better correlation analysis needs to be conducted to determine the direct versus indirect impact that mechanized military training has on the soil physical properties. For example, does the tracked vehicle disturbance directly lead to the higher bulk density values, loss of organic matter, and large erosion potential, or are these more the secondary effects of the loss of vegetation?

Another limitation of this and most other studies of military impacts is that they concentrate solely on characterizing the localized area of ridgetop disturbance. However, downslope areas of the watershed can be highly vulnerable to indirect impacts induced in the ridgetop training sites. Increased runoff and sediment deposition in riparian areas are seldom considered, yet potentially impact larger spatial areas and threaten the natural habitats of a more diverse range of species than the isolated hilltops. Future research of disturbance-level indicators of training impacts need to better relate the local upslope disturbance and mitigation measures to watershed-wide consequences.

REFERENCES

Assouline, S. and Y. Mualem. 2002. Infiltration During Surface Sealing: The Effect of Areal Heterogeneity of Soil Hydraulic Properties. Water Resour. Res. 38(12): Art. No. 1286.

Corradini, R., R. Morbidelli, F. Melone. 1998. On the Interaction of Infiltration and Hortinian Runoff. J. Hydrol. 204: 52-67.

Dane, J. H., D. K. Cassel, J .M. Davidson, W. L. Pollans, and V. L. Quisenberry. 1983. Physical Characteristics of Soils of the Southern Region - Troup and Lakeland Series. Alabama, Soil Coop. Series Bull. 262, Auburn University.

Dilustro, J. J., B. S. Collins, L. K. Duncan, and R. R. Sharitz. 2002. Soil Texture, Land-use, and Vegetation of Fort Benning Upland Forest Sites. J. of the Torrey Botanical Soc. 129(4): 289-297.

Elrick, D. E. and W. D. Reynolds. 1992. Infiltration from Constant-Head Well Permeameters and Infiltrometers. Advances in Measurement of Soil Physical Properties: Bringing Theory into Practice. Soil Sci. Soc. of Am. Special Pub. No. 30.

Fang, S., S. Wente, G. Z. Gertner, G. Wang, and A. Anderson. 2002. Uncertainty Analysis of Predicted Disturbance from Off-Road Vehicular Traffic in Complex Landscapes. Environ. Management 30(2): 199-208.

Fiedler, F. R., G. W. Frazier, J. A. Ramirez, and L. R. Ahuja. 2002. Hydrologic Response of Grasslands: Effects of Grazing, Interactive Infiltration, and Scale. J. Hydrol. Eng. 7(4): 293-301.

Flint, L. E. and A. L. Flint. 2002. Porosimetry. *In* Dane, J. H. and G. C. Topp (Eds.), Methods of Soil Analysis, Part 4, Physical Methods, Soil Sci. Soc. Am., Inc., 241-253.

Garten, C.T., Jr., T.L. Ashwood, and V.H. Dale. 2003. Effect of military training on indicators of soil quality at Fort Benning, Georgia. Ecological Indicators 3: 171-179.

Georgia Soil and Water Conservation Commission. 2000. Manual for Erosion and Sediment Control in Georgia. Fifth edition. 4310 Lexington Road, Athens, GA 30603

Grantham, W. P., E. F. Redente, C. F. Bagley, and M. W. Paschke. 2001. Tracked Vehicle Impacts to Vegetation Structure and Soil Erodibility. J. Range Management 54(6): 711-716.

Haws, N. W., B. Liu, E. J. Kladivko, P. S. C. Rao, C. W. Boast, and D. P. Franzmeier. (in review). Spatial Variability and Measurement Scale for Infiltration Rate on an Agricultural Landscape. Submitted to Soil Sci. Soc. Am. J.

Kosugi, K. 1994. Three parameter lognormal distribution model for soil-water retention. Water Resour. Res. 32:2697-2703.

Kosugi, K. and J. W. Hopmans 1998. Scaling Water Retention Curves for Soil with Lognormal Pore-Size Distribution. Soil Sci. Soc. Am. J. 62: 1496-1505.

Merz, B. and E. J. Plate. 1997. An Analysis of the Effects of Spatial Variability of Soil and Soil Moisture on Runoff. Water Resour. Res. 33(12): 2909-2922.

Mualem. Y. 1976. A new model for predicting the hydraulic conductivity of unsaturated porous media. Water Resour. Res. 12:513-522.

Nachabe, M. H., T. H. Illandasekare, H. J. Morel-Seytoux, L.R. Ahuja, and H. Ruan. 1997. Infiltration Over Heterogeneous Watershed: Influence of Rain Excess. J. Hydrol. Eng. 2(3): 140-143.

Prosser, C. W., K. K. Sedivec, and W. T. Barker. 2000. Tracked Vehicle Effects on Vegetation and Soil Chaaracteristics. J. Range Management 53(6): 666-670.

Rogers, J.S. and A. Klute. 1971. The hydraulic conductivity-water content relationship during nonsteady flow through a sand column. Soil Sci. Soc. Am. Proc. 35:695-700.

Stakman, W.P., G.A. Valk., and G.G. van der Harst. 1969. Determination of soil moisture retention curves, I. 3rd revised ed. Institute for Land and Water Management Research, Nageningen, the Netherlands.

Su, C. and R.H. Brooks. 1980. Water retention measurements for soils. J. Irrig. Drain. Div., Proc. Am. Soc Eng. 106:105-112.

Trumbull, V. L., P. C. Dubois, R .J. Brozka, and R. Guyette. 1994. Military Camping Impacts on Vegetation and Soils of the Ozark Plateau. J. Environ. Management 40(4): 329-339.

United States Department of Agriculture (USDA). 1997. Soil Survey of Chattahoochee and Marion Counties, Georgia. Natural Resource Conservation Service.

Wetzel, K. F. 2003. Runoff Production Processes in Small Alpine Catchments Within the Unconsolidated Pleistocene Sediments of the Lainbach Area (Upper Bavaria). Hydrological Processes 17(12): 2463-2483.

Whitecotton, R. C. A., M. B. David, R. G. Darmody, and D. L. Price. 2000. Impact of Foot Traffic from Military Training on Soil and Vegetation Properties. Environ. Management 26(6): 697-706.

Table 1: Summary statistics for scaling factors, bulk density, and infiltration rate for training (Tr), nontraining (NT), and off-site (OS) cores.

Statistic	**Scaling Factor**			**†Bulk Density**			**†Infiltration Rate**	
	Tr	**NT**	**OS**	**Tr**	**NT**	**OS**	**Tr**	**NT**
Minimum	0.50	0.42	0.63	1.40	1.07	1.48	1.1	12
Maximum	1.44	1.95	2.10	1.75	1.60	1.65	35.8	53.0
Mean	0.83	1.11	1.17	1.54	1.38	1.58	12.0	26.8
Variance	0.05	0.15	0.191	0.01	0.02	0.003	86.5	154
Skew	1.15	0.38	0.33	0.39	-0.66	-0.69	1.22	0.97
P-value	0.0013			0.0008			0.0001	

†All units in g, cm, hr.

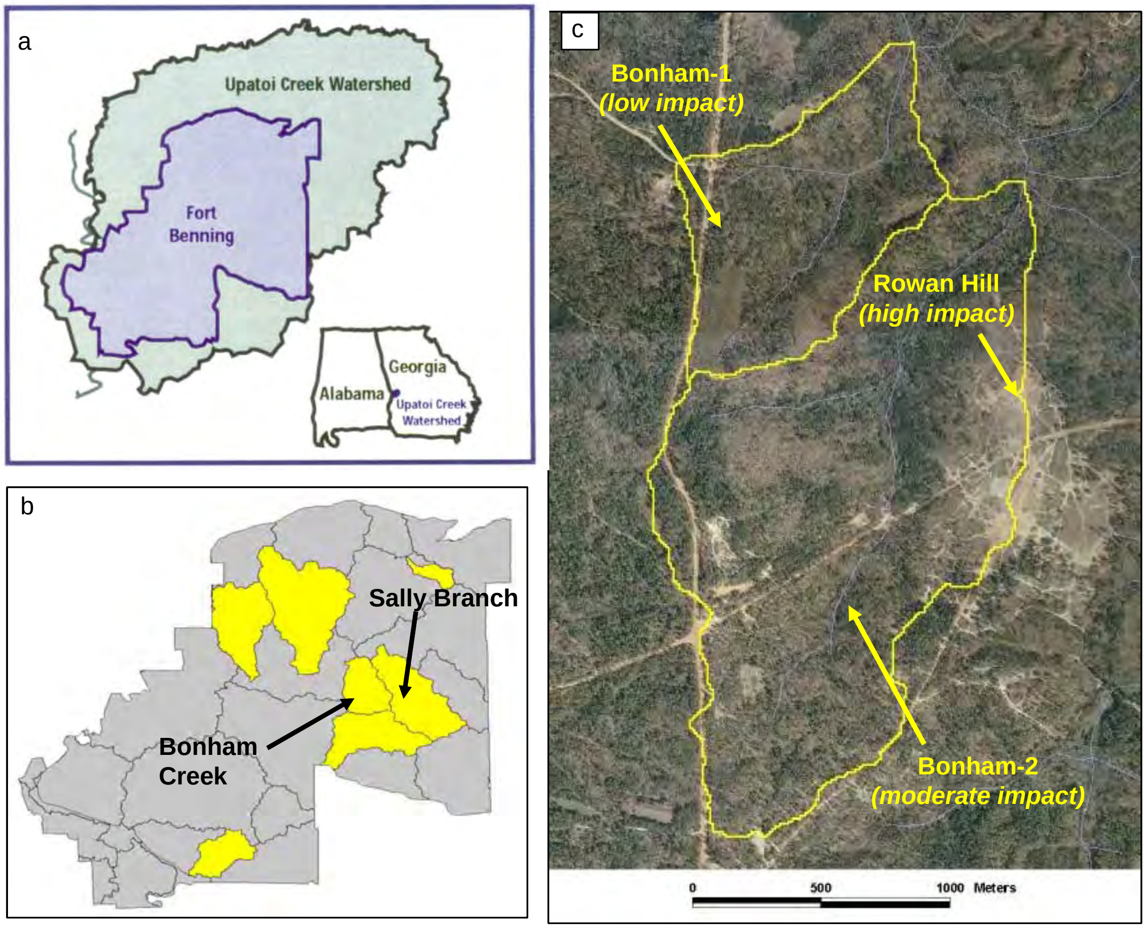


Figure 1
Location maps for (a) Fort Benning, (b) Bonham Creek and Sally Branch watersheds, and (c) Bonham-1 and Bonham-2 (including the Rowan Hill training site) subwatersheds. The highlighted areas in (b) show other watersheds sampled as part of a larger analysis of which this study was a part.

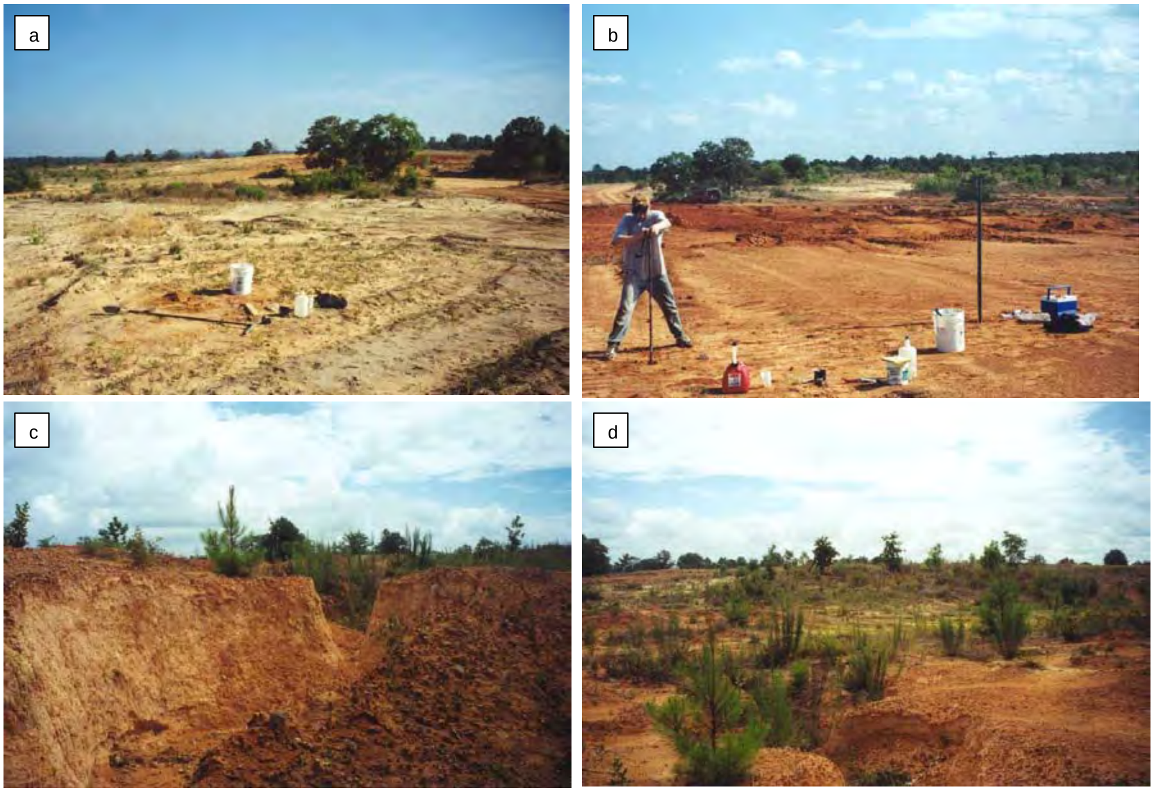


Figure 2
Impacted ridgetop at Rowan Hill: (a) a less impacted site outside of the main training area, (b) a central training site completely devoid of vegetation and with mixing of surface and subsurface horizons (indicated by reddish hue of soil), (c) and (d) large runoff and erosion channels leading downslope from Rowan Hill.

Figure 3
Downslope impacts from ridgetop training: (a) and (b) sediment deposits from the Cannons training site, and (c) sediment deposits downslope from Rowan Hill.

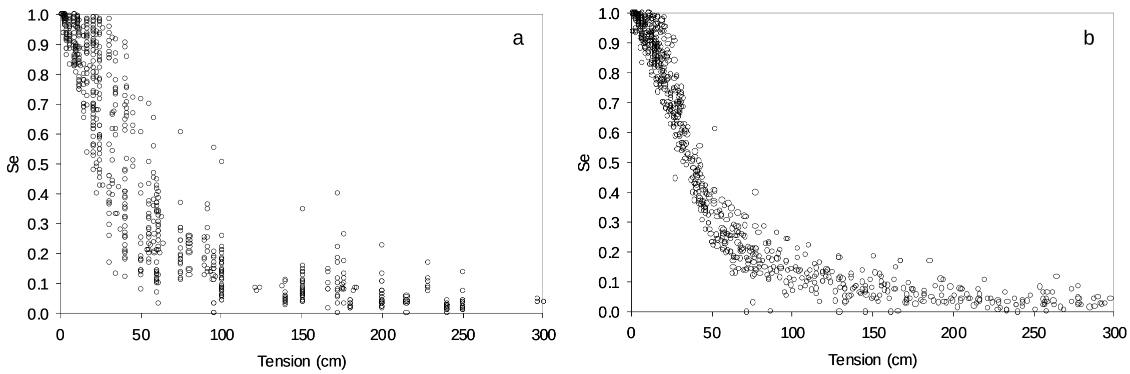


Figure 4
Unscaled (a) and scaled (b) data for the soil water retention curves measured from the Fort Benning and off-site soil cores.

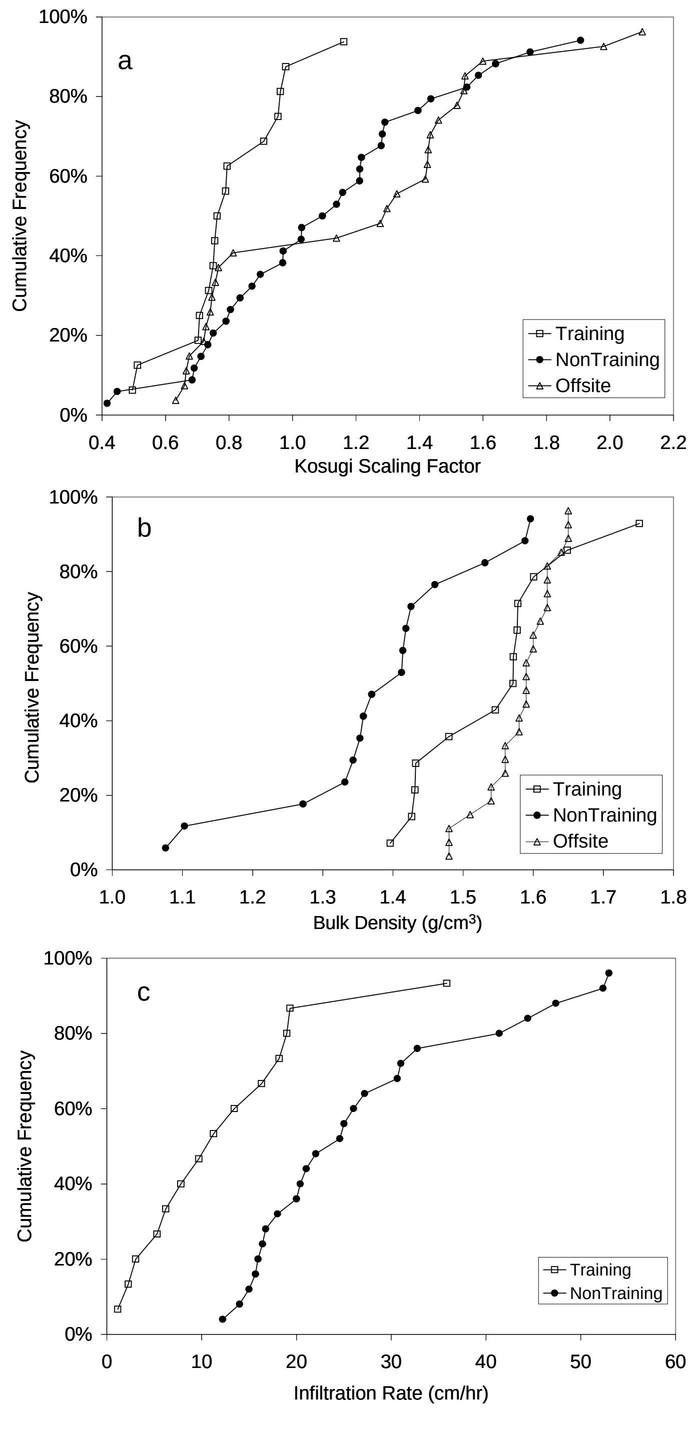


Figure 5
Cumulative frequency distributions of scaling factors (a), bulk densities (b), and infiltration rates

Stream Water quality

- **3.4.1 Stream TOC and TKN concentration decreased with increasing soil and vegetation disturbance (proportion of bare ground) in the watershed, reflecting depletion of soil organic matter and detritus in uplands and reduced leaching in soils due to short-circuited flow paths (gulleys) from uplands to streams.**

- **3.4.2 Enzyme activities relative to patterns of biogeochemistry and soil water content in riparian wetlands varied with distance from stream edge and help explain temporal patterns of groundwater Total Kjeldahl Nitrogen (TKN) related to leaf fall and canopy loss in riparian forests.**

3.4.1

Ecological Indicators in Forested Watersheds: Relationships between Watershed Characteristics and Stream Water Quality in Fort Benning, GA. Bhat, S., J.M. Jacobs, K. Hatfield, and J. Prenger.

ABSTRACT

Watershed hydrology depends on many factors including land use, climate, and soil conditions. However, the relative impacts of different types of land use on the surface water are yet to be ascertained and quantified. This research examines effects of land use on the surface water quality at different watershed scales. 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. Stream pH, temperature, and conductivity were positively correlated with total length of all roads within the watershed. Stream TKN was correlated negatively with disturbance index. TP was negatively correlated with road density whereas positively correlated with number of road crossing streams. Chloride showed a positive correlation with the total length of all roads within the watershed. TOC was negatively correlated with military land and a disturbance index based on percentage bare land on slopes greater than 3%. This study identified strong relationships among selected watershed physical characteristics that are more susceptible to human induced disturbances and water quality parameters. Regression results suggested that 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.

KEY WORDS: Fort Benning, ecological indicators, watershed characteristics, land use, streams, water quality, military training

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 (NRC) of the United States recently conducted a critical evaluation of indicators used to monitor ecological changes from either natural or anthropogenic causes. According to this report (NRC, 2000), during recent decades efforts have been increasing to develop reliable and comprehensive environmental indicators because of growing environmental concerns. 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. NRC (2000) 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 predict stream N and P concentrations.

Within a military installation context, land managers are challenged to use land for military training purposes in a manner which 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 range lands, 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, Georgia military installation. We hypothesize 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 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 ^{o} and 15^{o}C. An average daily maximum and minimum temperature of 15.5^{o} and –1^{o}C are reported in the coldest months, January and February . Annual precipitation averages 105 cm 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, including harvesting and thinning continues, and the loblolly and longleaf pine forests are subjected to regular low-level fires for management purposes (Dale et al., 2002).

METHODS OF STUDY

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 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.

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. 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. 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. TM imagery and digital elevation maps were obtained from Strategic Environmental Research and Development Plan (SERDP)'s Ecosystem Management Project (SEMP) database (http://sempdata.wes.army.mil). 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 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 Ft. Benning installation.

Some watershed characteristics are strongly correlated at significance level of 0.05 or lower (Table 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, military areas with normalized difference vegetative index (NDVI) and stream density, and DIN, and road density with number of roads crossing streams.

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). 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). 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 4, 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. TKN, TP, and Cl 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.

Effects of disturbance categories

Table 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 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 4 shows that each water quality parameter had a significant relationship with one or more watershed physical characteristics (Table 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. pH, temperature, 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. DIN was negatively correlated with TKN and TOC.

Graphical relationships provide insight into nonlinear relationships that exist between indicators and response variables. Figures 3-6 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, 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 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. TP 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 crossing streams, whereas temperature was captured by % area loamy 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). In Figures 3-6, we plot 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 6). The plots in Figure 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 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 3); number of roads crossing streams with pH and TP (Figure 5); and disturbance index with TKN (Figure 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 roadways and percent bare land are better indicators 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 percentage of non-forest area at the watershed scale to be a good predictor of stream nitrogen concentrations.

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 loam 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.

CONCLUSIONS

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. Our 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. TKN and TOC variations were well captured by DIN and extent of military land, respectively. TSS variability, on the other hand, as may be expected 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. We suggest 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.

REFERENCES

American Public Health Association, 1992. Standard Methods for the Examination of Water and Wastewater. American Public Health Association.

Basnyat, P., Teeter, L., Flynn, K., Lockaby, B., 1999. Relationships between landscape characteristics and nonpoint source pollution inputs to coastal estuaries. Environ. Manage. 23, 539-549.

Bolstad, P.V., Swank, W.T., 1997. Cumulative impacts of land use on water quality in a southern Appalachian watershed. J. Am. Water Resour. Assoc. 33, 519-533.

Close, M.E., Davies-Colley, R.J., 1990. Baseflow chemistry in New Zealand rivers 2. Influence of environmental factors. New Zealand J. Mar. Fres. Res. 24, 343-356.

Dale, V.H., Beyeler, S.C. 2001. Challenges in the development and use of ecological indicators. Ecol. Indicators. 1, 3-10.

Dale, V.H., Beyeler, S.C., Jackson, B., 2002. Understory vegetation indicators of anthropogenic disturbance in longleaf pine forests at Fort Benning, Georgia, USA. Ecol. Indicators, 1, 155-170.

Fisher, D.S., Steiner, J.L., Endale, D.M., Stuedemann, J.A., Schomberg, H.H., Franzluebbers, A.J., Wilkinson, S.R., 2000. The relationship of land use practices to surface water quality in the Upper Oconee Watershed of Georgia. For. Ecol. Manage. 128, 39-48.

Frick, E.A., Hippe, D.J., Buell, G.R., Couch, C.A., Hopkins, E.H., Wangsness, D.J., Garrett, J.W., 1998. Water Quality in the Apalachicola-Chattahoochee-Flint River Basin, Georgia, Alabama, and Florida, 1992–95. U.S. Geological Survey Circular-1164.

Garten, Jr., C.T., Ashwood, T.L., Dale, V.H., 2003. Effect of military training on indicators of soil quality at Fort Benning, Georgia. Ecol. Indicators. 3, 171-179.

Harrel, R.C., Dorris, T.C., 1968. Stream order, morphometry, physio-chemical conditions, and community structure of benthic microinvertibrates in an interminnent stream system. Ameri. Natur. 80, 221-251.

Hill, A.R., 1981. Streams phosphorus exports from catchments with contrasting land uses in southern Ohio. Water Resour. Bul. 17, 627-634.

Hunsaker, C.T., Levine, D.A., 1995. Hierarchical approaches to the study of water quality in rivers. BioScience. 45, 193-203.

Hynes, H.B.N., 1960. The Biology of Polluted Waters, Liverpool University Press, Liverpool.

Iverson, R.M., Hinckley, B.S., Webb, R.M., Hallet, B., 1981. Physical effects of vehicular disturbance on arid landscapes. Science. 212, 915-917.

Johnes, P., Moss, B., Phillips, G., 1996. The determination of total nitrogen and total phosphorous concentrations in freshwaters from land use, stock headage and population data: testing a model for use in conservation and water quality management. Fresh. Biol. 36, 451-473.

Johnson, L. B., Richards, C., Host, G.E., Arthur, J.W., 1997. Landscape influences on water chemistry in midwestern stream ecosystems. Fresh. Biol. 37, 193-208.

Karr, J.R., Chu, E.W., 1999. Restoring Life in Running Waters: Better Biological Monitoring. Island Press, Washington, D.C.

Lowrance, R., 1992. Groundwater nitrate and denitrification in a coastal plain riparian forest. J. Env. Qual. 21, 401-405.
Lowrance, R., Todd, R.L., Asmussen, L.E., 1984. Nutrient cycling in an agricultural watershed: II. Streamflow and artificial drainage. J. Environ. Qual. 13, 27-32.
Meador, M.R., Goldstein, R.M., 2003. Assessing Water Quality at Large Geographic Scales: Relations Among Land Use, Water Physicochemistry, Riparian Condition, and Fish Community Structure. Environ. Manage. 31, 504-517.
National Research Council, 2000. Ecological Indicators for the Nation. National Academy Press, Washington, D.C.
O'Neill, R.V., Hunsaker, C.T., Jones, K.B., Riitters, K.H., Wickham, J.D., Schwartz, P.M., Goodman, I.A., Jackson, B.L., Baillargeon, W.S., 1997. Monitoring environmental quality at the landscape scale. BioScience. 47, 513-519.
Osborne, L., Wiley, M., 1988. Empirical relationships between land use/cover patterns and stream water quality in an agricultural watershed. J. Environ. Manage.. 26, 9-27.
Perry, C.D., Vellidis, G., Lowrance, R., Thomas, D.L., 1999. Watershed-scale water quality impacts of riparian forest management. J. Water Resour. Plan. Manage. 125, 117-125.
Richards, C., Haro, R.J., Johnston, L.B., Host, G.E., 1997. Watershed and reach-scale properties as indicators of macroinvertebrate species traits. Fresh. Biol. 37, 219-230.
Richards C., Host G. E., 1994. Examining land use influences on stream habitats and macroinvertebrates: a GIS approach. Water Resour. Bul. 30, 729-738.
Richards, C.,Johnston, L.B., Host, G.E., 1996. Assessing the influence of landscape-scale watershed features on physical habitat and stream biota. Can. J. Fisher. Aquat. Scien., 53, 295-311.
Semkin, R.G., Jeffries, D.S., Clair, T.A., 1994. Hydrochemical methods and relationships for study of stream output from small catchments. In: Molden, B., Cerny, J. (eds.), Biogeochemistry of Small Catchments: A Tool for Environmental Research. John Wiley & Sons, Chichester, pp. 163-188.
Sliva, L., Williams, D.D., 2001. Buffer zones versus whole catchment approaches to studying land use impact on river water quality. Water Resear. 35, 3462-3472.
Sponseller, R.A., Benfield, E.F., Valett, H.M., 2001. Relationships between land use, spatial scale and stream microinvertibrate communities. Fresh. Biol. 46, 1409-1424.
Thurow, T.L., Warren, S.D., Carlson, D.H., 1993. Tracked vehicle traffic effects on the hydrologic characteristics of central Texas rangeland. Trans. Ameri. Soc. Agri. Engr. 36, 1645-1650.
Wang, L., Lyons, J., Kanehl, P., Bannerman, R., 2001. Impacts of urbanization on stream habitat and fish across multiple spatial scales. Environ. Manage. 28, 255-266.

Table 1. Physical characteristics of study watersheds in Fort Benning, Georgia.

Physical Characteristics	Watersheds						
	Bonham-1	Bonham-2	Bonham	Little Pine Knot	Oswichee	Randall	Sally
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[1]							
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[2]							
Sand, %	78	69	69	72	24	68	49
Loam, %	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

Table 1. Physical characteristics of study watersheds (Cont.)

Physical Characteristics	Watersheds						
	Bonham-1	Bonham-2	Bonham	Little Pine Knot	Oswichee	Randall	Sally
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. 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***

Table 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, ^oC	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	*

Table 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

	PIN	DCD	MXD	WET	MIL	NDVI	PBL	DIN	SND	LOM	RDL	RDN	NRC	STD
pH	0.36	-0.47	-0.79*	0.24	0.09	0.32	-0.45	-0.65	-0.50	0.57	0.94***	-0.75*	0.93***	-0.14
Temperature	0.78	-0.37	-0.55	0.62	0.00	0.11	-0.25	-0.36	-0.02	0.16	0.95***	-0.42	0.59	-0.11
Conductivity	0.54	-0.40	-0.38	0.44	-0.10	0.10	-0.51	-0.60	0.14	0.03	0.82*	-0.54	0.51	-0.01
TKN	0.00	-0.12	-0.39	0.11	-0.50	0.60	-0.72	-0.84*	-0.19	0.39	0.19	-0.72	0.36	0.39
TP	-0.09	-0.33	-0.83*	-0.11	0.10	0.46	-0.42	-0.63	-0.78*	0.81*	0.57	-0.78*	0.90**	-0.04
TOC	0.15	0.39	-0.24	0.18	-0.87**	0.80*	-0.81*	-0.76*	0.15	0.07	0.09	-0.35	0.06	0.82*
Cl	0.56	0.13	-0.49	0.37	-0.47	0.55	-0.68	-0.65	0.06	0.07	0.79*	-0.32	0.47	0.44
TSS	-0.14	0.42	-0.11	-0.29	-0.53	0.54	-0.87**	-0.72	0.15	-0.14	0.30	-0.15	0.19	0.65

Table 5. Stepwise multiple regression models for water quality parameters. pH is unitless, temperature is measured in degrees centigrade, conductivity is measured in μS/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
pH	Pine, Road Length 5.50 – 0.0382 PIN + 0.00924 RDL	0.98***
Cl	Military, Road Length 2.19 + 0.145 MIL + 0.00344 RDL	0.90**
TP	No. of Roads Crossing Streams 0.00451 + 0.000236 NRS	0.81**
Conductivity	Sandy Soil, No. of Roads Crossing Streams - 24.2 + 0.552 SND + 0.666 NRS	0.80*
Temperature	Loamy Soil, Road Density 26.1 – 0.051 LOM – 1.49 RDN	0.77*
TOC	Military Land 3.45 – 0.274 MIL	0.76**
TSS	Percent of Bare Land 11.2 – 0.648 PBL	0.76**
TKN	Disturbance Index 0.445 - 0.0109 DIN	0.70*

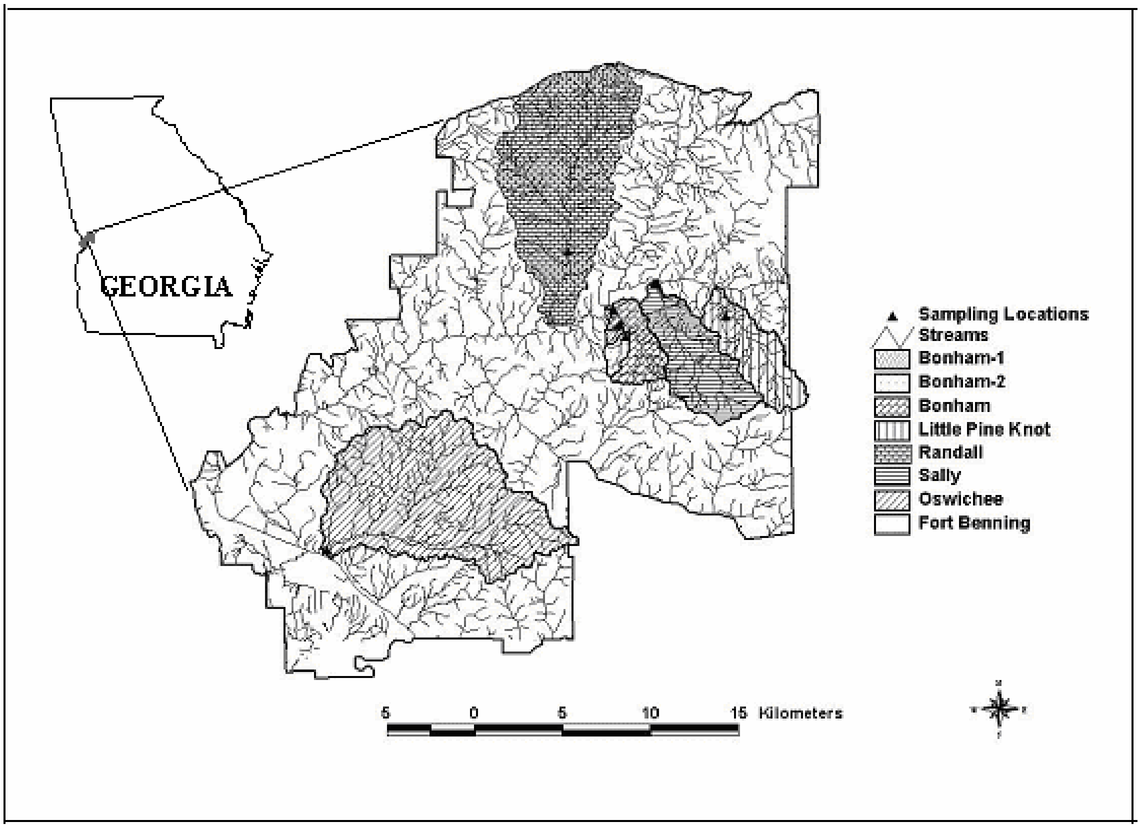


Figure 1. Study watersheds, Bonham-1, Bonham-2, Bonham, Little Pine Knot, Randall, Sally, and Oswichee, in the Fort Benning military installation. Also shown are stream network and sampling locations.

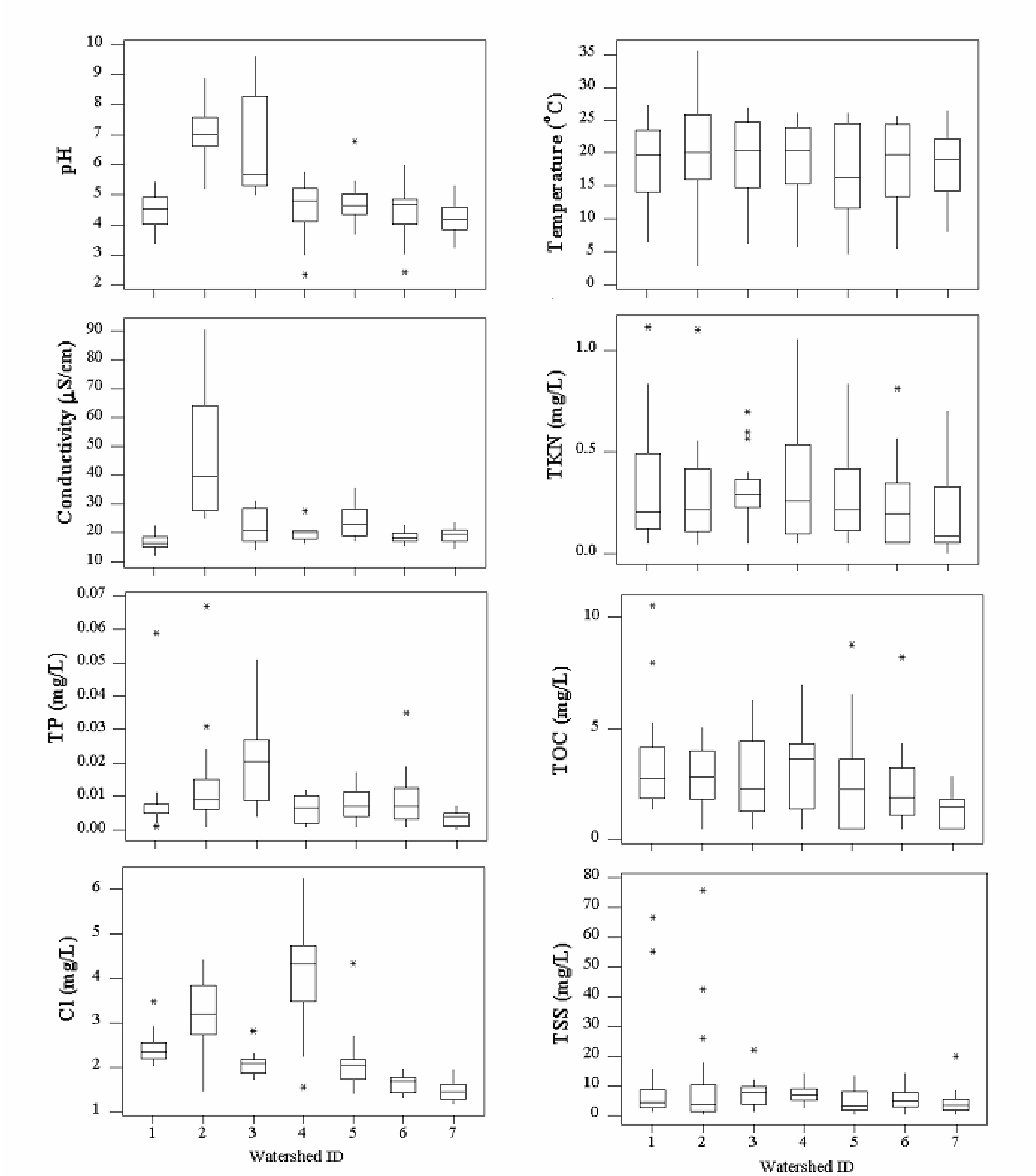


Figure 2. Box plots of water quality parameters. Each plot consists of outliers, most extreme data, 75th, 50th, and 25th percentile values. Watershed IDs represent- 1: Bonham-1, 2: Randall, 3: Oswichee, 4: Little Pine Knot, 5: Sally, 6: Bonham, and 7: Bonham-2.

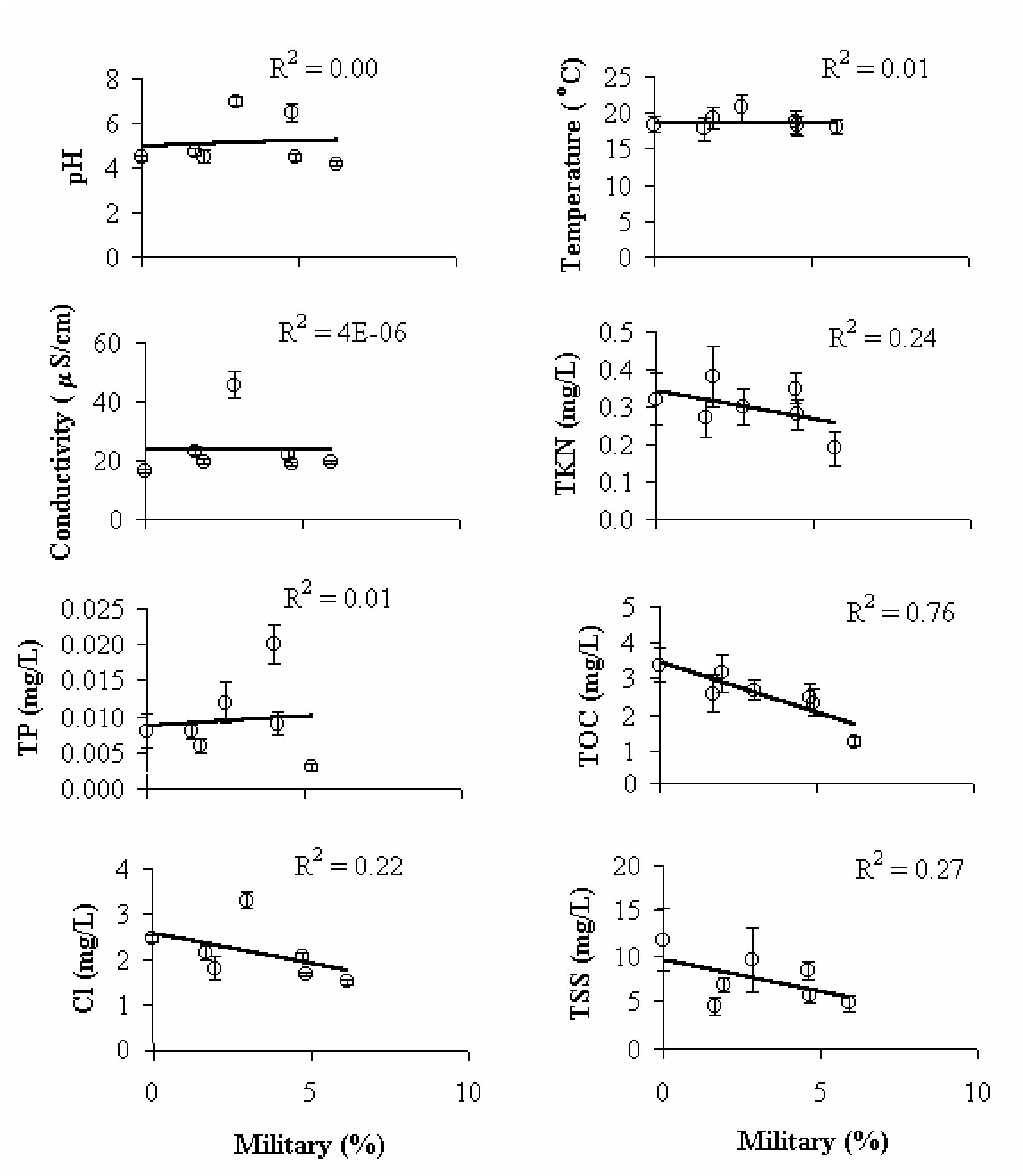


Figure 3. Relationships between military land and water quality parameter. Vertical bars represent standard errors.

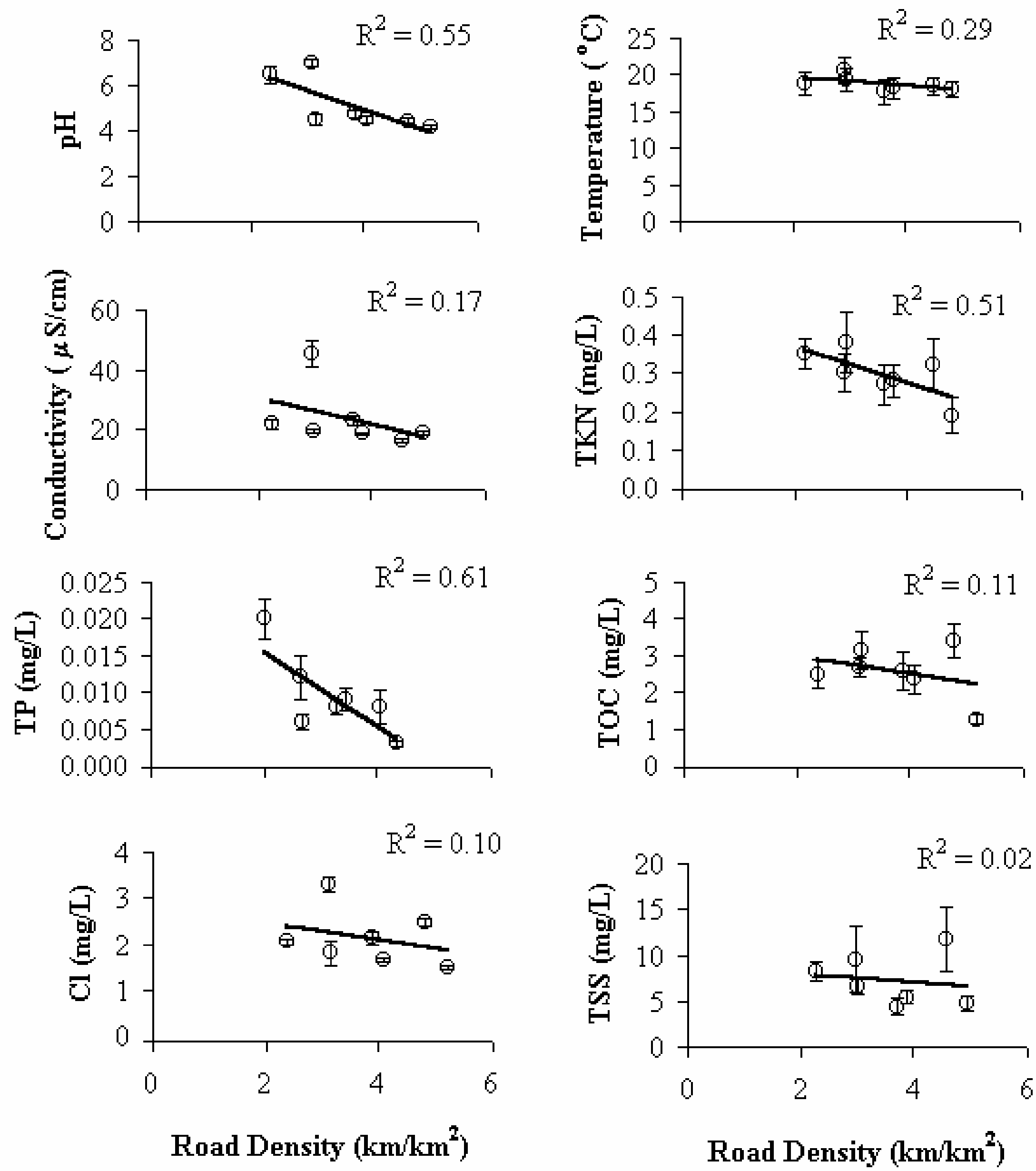


Figure 4. Relationships between road density and water quality parameter. Vertical bars represent standard errors.

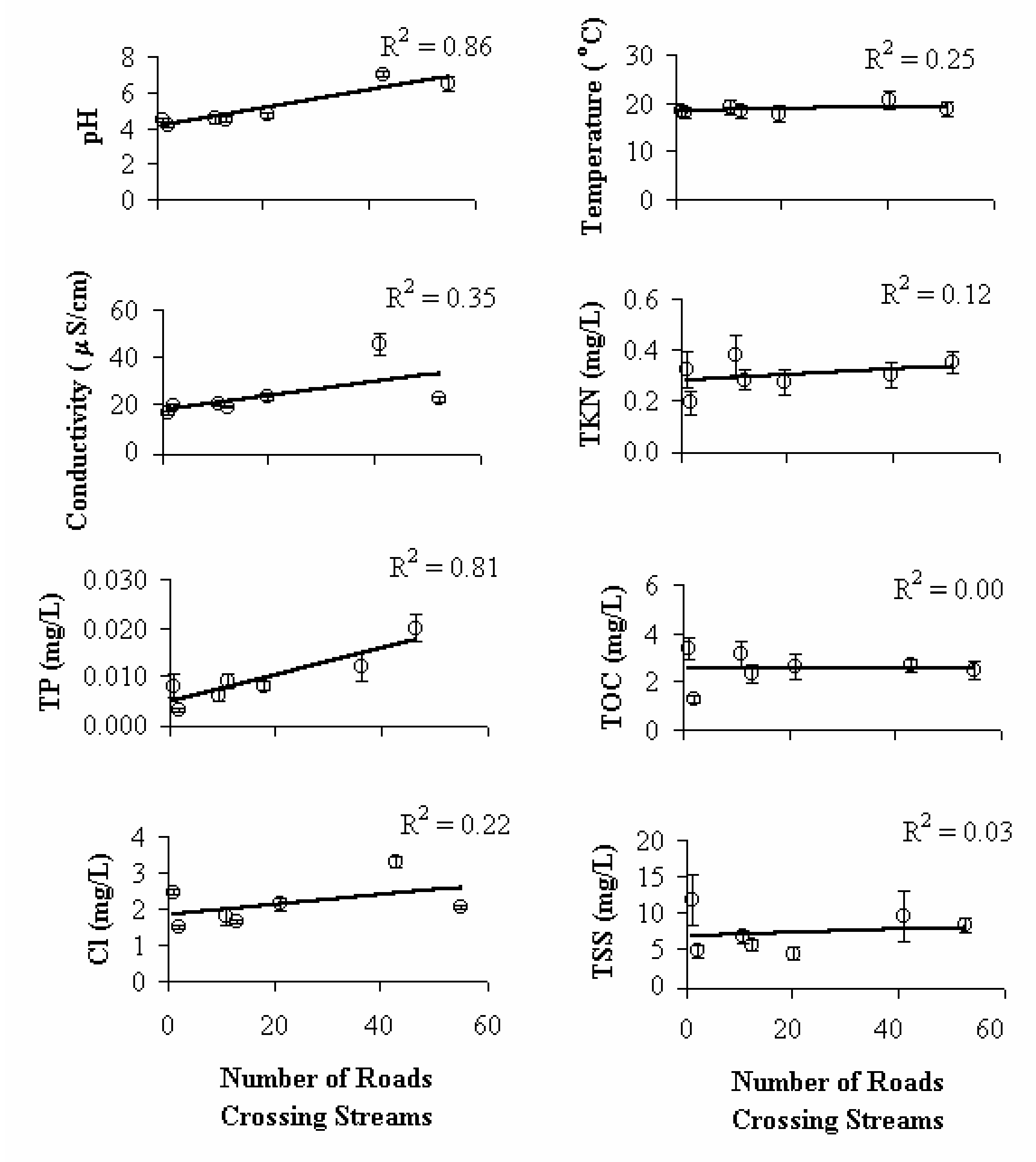


Figure 5. Relationships between number of roads crossing streams and water quality parameter. Vertical bars represent standard errors.

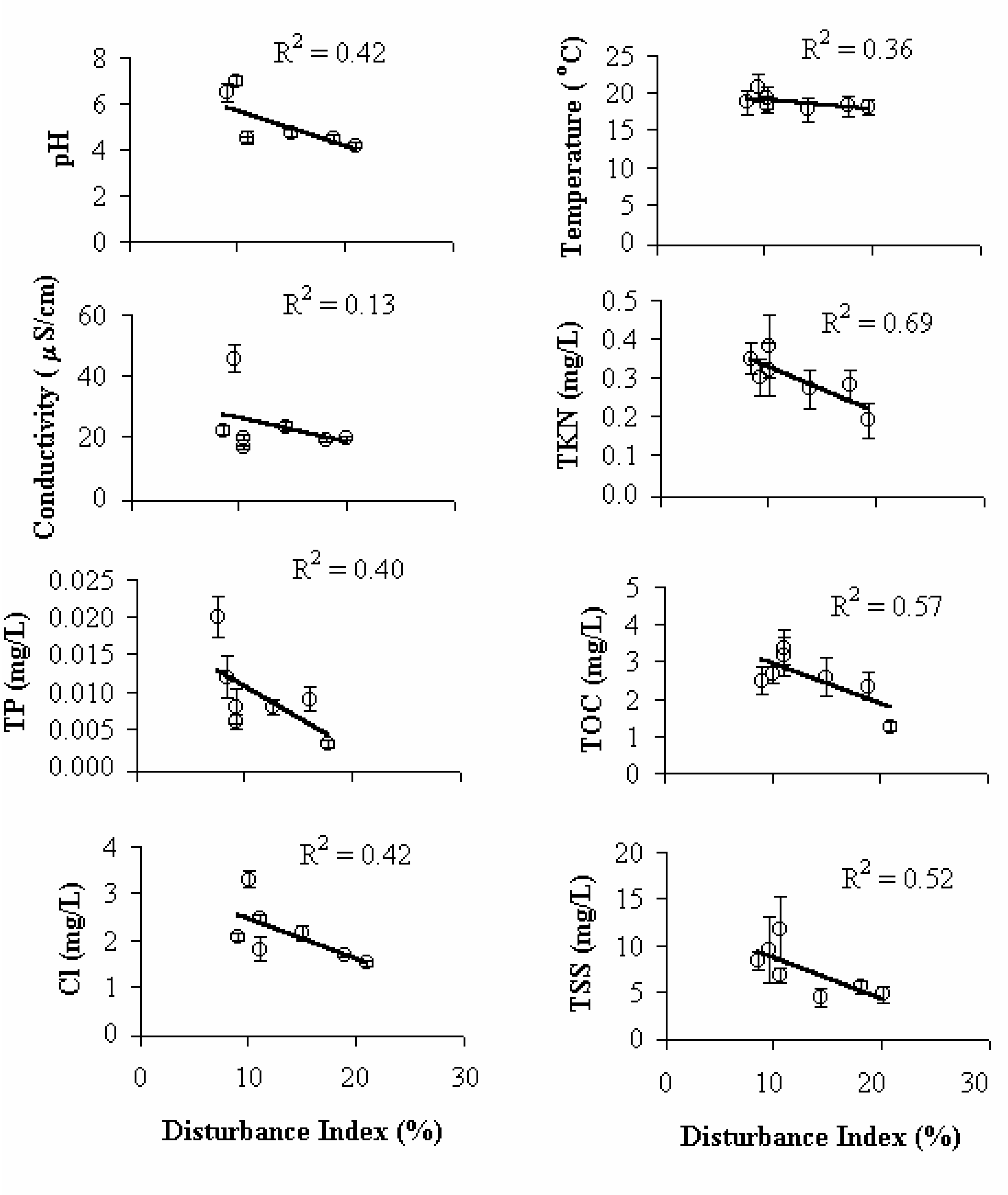


Figure 6. Relationships between disturbance index and water quality parameter. Vertical bars represent standard errors.

3.4.2
Microbial Nutrient Cycling in the Riparian Zone and its Influence on Stream Chemistry. Prenger, J.P., Bhat, S., J.M. Jacobs, and K. R. Reddy.

ABSTRACT

Riparian wetlands are diverse ecosystems that contribute organic matter and nutrients to streams either directly or through lateral transport of dissolved organic matter (DOM) from the forest floor. Plant and microbial communities are influenced by soil water content, sediment grain size, and micro-topography, and these in turn influence the amount and quality of organic inputs to streams. Microbial processes, including enzymatic hydrolysis of high molecular weight compounds to make them available to microbial or plant populations, regulate storage and transformation of soil nutrients and their availability to transport through groundwater. The current study describes patterns of biogeochemistry and select enzyme activities relative to position and soil water content in riparian wetlands of two second order streams, one impacted by erosion due to mechanized military training and forestry management practices. Soil cores were obtained in December 2002 on either side of the streams at approximately 80 meter intervals, as well as in three transects normal to stream flow. Stream and groundwater water chemistry were monitored monthly in transects normal to stream flow in one second order watershed. Variability in microbial enzyme activities and soil total nitrogen (TN) were most closely associated to soil water content, while groundwater Total Kjeldahl Nitrogen (TKN) showed temporal patterns related to leaf fall and canopy loss in riparian forests and varied with distance from stream edge. Patterns of peptidase activity were complex, with minima observed at approximately 30% soil moisture content.

Keywords: Disturbance; land use; Total Kjeldahl Nitrogen; soil water content; soil enzymes; peptidase.

INTRODUCTION

Bottomland riparian areas are diverse ecosystems in which a variety of natural disturbances create temporal and spatial mosaics (Naiman and Décamps, 1997). This zone includes the area between low and high water marks, as well as the band of vegetation influenced by the water holding capacity of soils, elevated water tables or flooding (Naiman and Décamps, 1997). Plant distribution is influenced by soil saturation, texture, and microtopography (Hupp, 2000). Vegetation modifies hydraulic conditions, affecting sediment trapping and helping to determine soil structure (Naiman and Décamps, 1997; Hupp, 2000). Riparian areas contribute litter and organic matter to streams either directly or through lateral transport of dissolved organic matter (DOM) from the forest floor (Fisher and Likens, 1973; Bishop et al., 1994; Naiman and Décamps, 1997). DOM contributed by fringing wetlands is dominated by fresh and labile organic material (Naiman and Décamps, 1997).

Hydrology strongly influences movement of organic matter through riparian areas (Huggenberger et al., 1998). Throughfall contributes low-molecular weight organic matter leached from canopy and understory vegetation in both upland and riparian zones (Pusch et al., 1998). Water moving through soil accumulates soluble organic compounds from decomposing plant material as well as from root exudates, and becomes enriched in the more refractory forms of DOM (Pusch et al., 1998). The combination of soil saturation and high organic content produces anaerobic conditions that may have profound effects on redox gradients (Pusch et al., 1998), nutrient availability and export (Mulholland, 1992).

The dynamic nature of microbial biomass makes it sensitive to changes in nutrient and organic matter loading rates and other perturbation. Microbial processes regulate storage and transformation of soil nutrients, and flow of nutrients through the soil microbial fraction can be substantial (Martens, 1995). Organic material in wetlands is largely composed of relatively high molecular weight compounds which require hydrolysis by extracellular enzymes before becoming available to microbial or plant populations, or to transport through groundwater (Burns, 1982; Chrost, 1991; Sinsabaugh et al., 1991). Exogenous enzymes released by microbial cells are integral to degradation of organic matter and plant detritus and thus affect soil and water quality (Halemejko and Chrost, 1984; Sinsabaugh et al, 1991). Activity of microbiota associated with organic carbon deposits can reduce oxygen concentrations in groundwater, thereby affecting redox conditions, nutrient adsorption, and denitrification processes (Mulholland, 1992).

This study is part of an effort to develop indicators of watershed integrity and soil degradation on military lands as part of the Strategic Environmental Research and Development Program's (SERDP) Ecosystem Management Project (SEMP) conducted at Ft. Benning, GA, USA. The goal of this research was to develop sensitive and selective indicators of ecological change as resource management tools in support of an environmentally sustainable military mission. The specific objective for the current study was to determine nutrient storages in the riparian areas of streams and biogeochemical cycles that contribute to stream water quality.

MATERIALS AND METHODS

Site Description

The study area is within the Ft. Benning military reservation in west-central Georgia, in the Carolina and Georgia sand hills major land resource area (USDA, 1997). Upland soils in the area are primarily well to excessively drained Ultisols and Entisols, supporting forests of slash (*Pinus elliotii*), longleaf (*P. palustris*), and loblolly (*P. taeda*) pines. Excessively-drained Lakeland soils (Entisol) of sandhill communities are associated with longleaf pine, turkey oaks (*Quercus laevis*), blackjack oaks (*Q. marilandica*), and post oaks (*Q. stellata*) near ridgetops in the central and northern portion of the reservation. Loamy soils of relatively high clay content occur in upland areas in a band across the southern portion of the installation. Wetlands and hydric soils are generally restricted to bottomlands along streams and creeks. Military related impacts result from the direct removal of or damage to vegetation, digging activities, and ground disturbance from vehicles. The mechanized forces in particular use tracked and wheeled vehicles that cause soil disturbance and movement that may result in soil erosion and stream sedimentation. This study focused on two 2nd order watersheds: Bonham-1 is approximately 0.76 km^2, while Bonham-2 is approximately 2.21 km^2.

Water and Soil Sampling

Soil cores were obtained from transects along two 2nd order streams, Bonham-1 and -2, in December 2002; a subset of Bonham-2 sites was re-sampled in June 2003 (Fig. 1). Soil samples were taken at about 5 meters from stream bank on either side at approximately 80 meter intervals. Three transects perpendicular to stream flow were sampled in December 2002 within the riparian zone at approximately 5, 10 and 15 m from streambank and 50-150 m upstream of groundwater wells. Each soil sample consisted of a composite of two 10-cm deep sub-samples taken using a 6.5-cm diameter polycarbonate corer within a 1 m^2 quadrat. Canopy cover and throughfall were monitored at 9 stations in different forested and herbaceous communities in the Bonham-1 and 2 watersheds, whereas precipitation was monitored in a clearing on the boundary between the Bonham-1 and Bonham-2 watersheds (Bryant et al., 2004). Groundwater and soil water chemistry were monitored from October 2001 to September 2003; biweekly from October 2001 to November 2002 and monthly thereafter in two transects perpendicular to a second order stream in Bonham-2 (Bhat et al., 2004). Groundwater wells were located at approximately 3-5 meters from stream edge and at 5-10 m intervals upslope, depending on the width of riparian area.

Chemical Analyses

Well mixed soil samples were analyzed for select biogeochemical parameters, as described below. Total C (TC) and N (TN) content were determined on dried, ground soil samples by dry combustion (Nelson and Sommers, 1996) using a Carlo-Erba NA-1500 CNS Analyzer (Haak-Buchler Instruments, Saddlebrook, NJ). Total P (TP) and ash content were determined by combusting approximately 0.2-0.5 g oven-dried, finely ground soil at 550^o C for 4 hrs, digesting the ash with 6M HCl and continuous heating on a hot plate, and filtering through No. 41 Whatman filter (Anderson, 1976), followed by analysis of P by automated ascorbic acid method (Method 365.4, USEPA, 1983). Water extractable carbon (WEC) was determined by extraction of the wet soil equivalent of 2.5 g soil dry weight in 25 mL of distilled deionized water with shaking for 1 hr, followed by filtration through 0.45 µm membrane filter (Kuo, 1996) and analysis on a Dohrmann DC-190 TOC Analyzer. Extractable ammonium (NH_4-N) and organic carbon (TOC) were

determined using the method of Mulvaney (1996) by extraction of the wet soil equivalent of 0.5 g (dry weight) soil with 25 mL 0.5M K_2SO_4 followed by filtration through Whatman No. 41 filter paper. The NH_4-N in extracts was determined using a Technicon™ Autoanalyzer (EPA 350.1); TOC in extracts was determined using a Dohrmann DC-190 TOC Analyzer. Total nitrogen in water samples was determined by Kjeldahl digestion (TKN) and Technicon™ Autoanalyzer (EPA 351.2).

Enzyme Analyses

Enzyme activities were assayed using the fluorescent model substrate 4-methylumbelliferone (MUF) (Chrost and Krambeck, 1986; Hoppe, 1993; Sinsabaugh et al., 1997) at the approximate ambient pH of 6.0. All soil enzyme analyses were performed on well mixed fresh material from which all visible roots and living plant material was removed. Enzyme analyses were completed within 2 weeks of sampling. Soil samples (~ 1 g) were placed in approximately 9 mL distilled water, and clumps were broken up by brief agitation with a Tissue Tearor Model 398 (Biospec Products, Bartlesville, OK). Immediately prior to enzyme assays, a 1/100 or 1/200 dilution of soil or detritus was prepared in water by serial dilution. Two hundred µL of well-suspended soil slurry was transferred by pipette into 8 wells of a 96 well microtiter plate, and 50 µL of substrate solution added to 4 wells (with 4 blanks). Samples were incubated (2 hr for phosphatase, 24 hr for all others) in the dark at room temperature except for dehydrogenase which was incubated at 30^{o}C. Phosphatase and β-glucosidase assays were stopped by addition of 10 µL 0.1 N NaOH. Substrate was added to blanks and immediately read on a Bio-Tek Model FL600 fluorometric plate reader (Bio-Tek Instruments, Inc., Winooski, VT). Dehydrogenase assays were stopped with 50 µL acetone, incubated an additional 2 hr and read. Substrate solutions were as follows: for acid phosphatase, 500 µM methyl-umbelliferyl (MUF)-phosphate in 5 mM MES pH 6.0; for β-glucosidase, 500 µM MUF-glucoside in 5mM MES pH 6.0; for peptidase, 500 µM MUF-guanidinobenzoate in in 5mM MES pH 6.0; for dehydrogenase, 500 µM 5-cyano-2,3-ditolyl tetrazolium chloride (CTC) in 100mM Tris pH 7.8. Concentrations were calculated from a standard curve of MUF or CTC-Formazan. Excitation (Ext.) and emission (Em.) spectra for the two fluoro-chromes were: Ext. 360$\pm$40, Em. 460$\pm$40 (MUF-P, MUF-G); and Ext. 530$\pm$25, Em. 645$\pm$40 (CTC). Enzyme activities were calculated as µM product g $^{-1}$ dry soil hour $^{-1}$. Normalized enzyme activities were divided by their respective maximum values and reported on a scale of 0-1.

Statistical Analysis

Pairwise comparison of means with Tukey-Kramer HSD and multivariate correlation matrix were performed using JMP version 4.0.5 (SAS Institute, Cary NC). All variables were examined for normality of variance and log transformed where necessary. Analysis of variance of univariate data included multiple comparison of means by Tuckey - Kramer and in all cases was at experiment-wise α=0.05.

RESULTS

Normalized enzyme activities generally showed a trend that paralleled soil moisture (Fig. 2). Although differences were not significant along transects perpendicular to streamflow, acid phosphatase (APA), β-glucosidase (BGA), and dehydrogenase (Dehyd) activities were all higher at the upland edge and trended downward closer to the stream edge. In contrast, peptidase activity (Pep) was lowest at the upland edge and higher at stream edge. Total organic carbon (TOC), water extractable carbon (WEC), and total carbon (TC) also paralleled soil moisture in these transects (Fig. 3), but because of high variance, these differences were not significant. Differences in total nitrogen (TN) were also not significant (Fig. 4); however, NH_4-N was significantly higher at the upland edge than in the middle or stream edge samples (Fig. 4).

Soil moisture was strongly correlated with extractable organic carbon and nitrogen, and with soil enzyme activities (Table 1). As expected, ash content had a negative correlation with these parameters and WEC as well. The microbial enzyme activities were most closely related to soil moisture, TC, WEC, extractable TOC, and TP (Table 1). Total N and extractable NH_4-N were not strongly correlated to any of the other parameters, nor were soil peptidase levels.

Enzyme activities were log-normally distributed; when the log transformed activities were plotted against soil water content (Fig. 5), strong linear relationships were observed. The exception was peptidase, the activity of which was better described by a quadratic relationship. The pattern in peptidase activity showed an inflection from decreasing to increasing enzyme levels that begins at about 20% soil moisture, which approximately coincides with a change in slope of Log TN levels relative to soil moisture (Fig. 6). The log of peptidase activities in soils with less than 30% soil moisture did not differ significantly between December (n=77) and June (n=18) samples; however in soils with greater than 30% soil moisture, log peptidase activities were significantly higher ($p<0.05$) in June (n=34) than December (n=70).

Groundwater monitored in transects perpendicular to the Bonham-2 stream showed a peak in TKN levels beginning in October that was most prominent in the near stream monitoring wells (3-5 m from stream bank). Figure 7 shows groundwater mean TKN values from approximately ten days before and after the December soil samples and six days before June samples were obtained. Similarly, Figure 8 shows mean TKN values of soil water taken at 20 cm depth from the soil surface for the same dates. Again because of high variance, no significant differences were observed. Peak TKN levels in Bonham-2 stream water were closely associated with time of leaf fall, with a secondary peak occurring in March and April (Fig. 9). A corresponding peak in ground water TKN was not observed in March/April. The response of stream TKN to stormwater flows was rapid (Fig. 10); streamflow peaked within 1-2 hours after peak precipitation during a February storm, and the increase in nitrogen concentration coincided closely with that of flow.

DISCUSSION

Microbial activity and decomposition of soil organic matter is affected by available carbon and phosphorus, carbon quality, soil moisture (Table 1; Figs. 2 and 5) and redox conditions (Mulholland, 1992; Smith et al., 2003). The lack of correlation between nitrogen levels and microbial activities in the current study may be explained to some extent by losses due to nitrification / denitrification (Mulholland, 1992; Godde and Conrad, 1998; Smith et al., 2003) under the alternating flooded and dry conditions common in riparian soils. The significantly higher NH_4-N in soils from the upland edge of the riparian zone (Fig. 4) may indicate increased inputs from upland areas, or lower nitrification rates, possibly due to higher soil moisture (Fig. 2) and transient anoxia (Mulholland, 1992). The more porous material typical of alluvial sediments encourages steep redox gradients and longer residence times (Pusch et al., 1998), contributing to nitrification / denitrification losses. Alluvial sediments often deepen near streams (Huggenberger et al., 1998), and reduction of nitrification due to soil drying in near stream sediments may increase relative discharge of NH_4^+ (Ohte et al., 1997). Lower NH_4-N in soils nearer the stream in December may indicate higher nitrification rates, but more likely is due to flushing by fall precipitation into groundwater and ultimately the stream (Figs. 7-10).

The hyperbolic curve of log peptidase activities relative to soil moisture (Fig. 5) indicates mineralization of organic nitrogen is affected in a complex way by soil water content. The slope of log TN versus soil moisture changes at about the same point as log peptidase activities (approximately 20-30% water content; Fig. 6), which may indicate a change in N mineralization rates at this level of soil saturation. The activity of nitrifiying bacteria is negatively affected by both dehydration and excess moisture (due to substrate limitation effects) brought about by changes in soil water potential (Stark and Firestone, 1995). Godde and Conrad (1998) found that soil water contents in the range of 30 to 60% favored de-nitrification due to creation of anaerobic zones in soil micropores, but that increases from 60-80% water content produced no further increase in activity. Since the nitrification / denitrification cycle requires alternating aerobic and anaerobic conditions, the patterns of TN and peptidase activity relative to water content might be explained by increasing nitrogen availability due to increased organic matter, microbial activity, and mobility of NH_4-N up to about 20% soil moisture, followed by increased losses due to nitrification / de-nitrification above 30% soil moisture.

In western Georgia, leaf senescence and dormancy results in reduced plant uptake of nutrients and deposition of organic material to the soil surface in late October and November. Decomposition and decreased uptake in the riparian area provides a pulse of N to ground and stream water, but has less effect on soil pore water (Figs. 7 and 9 vs. Fig. 8). The rate of decomposition is expected to slow down during the colder and dryer winter months and increase in spring as temperatures begin to rise, which may be indicated by the smaller nutrient pulse in late winter and early spring. Groundwater in the riparian area nearest the stream shows a transient increase in nitrogen soon after leaf fall begins (Figure 7), and precipitation flushes these nutrients into the stream (Figs. 9 and 10), thereby affecting in-stream WQ.

Although soil pore water concentrations show some seasonal variation, levels do not seem to respond directly to leaf fall and senescence (Fig. 8) and are instead higher on average during the June sampling period. One explanation for this may be the higher overall microbial activities observed in June, in particular on the upland edge, resulting in greater release of organic N. Nitrification and de-nitrification losses in soil pore spaces (Godde and Conrad, 1998) would reduce the contribution of soil pore water N to the groundwater N pool, most prominently in the upland fringe which has higher water content (Fig.2) and organic carbon (Fig. 3).

Microbial enzyme activities are closely associated with availability of labile organic material. High molecular weight compounds in soil organic matter require hydrolysis by extracellular enzymes before utilization by microbial or plant populations (Burns, 1982; Chrost, 1991; Sinsabaugh et al., 1991). Relative activities of these enzymes can therefore give information regarding nutrient availability within microbial communities associated with different disturbance, vegetation, or soil conditions (Sinsabaugh and Moorhead, 1994). Nutrient acquisition enzymes may provide more stable signals of nutrient availability and cycling than soil chemistry, to some extent integrating conditions over time. In addition, microbial activity responds to other soil conditions, which in turn may be determined by physical disturbance within the watershed. Peptidase activity shows a complex pattern that makes simple correlations weak (Table 1), but may be controlled by nitrogen availability (see Figs. 2 and 4) and soil moisture (Fig. 5). Nitrogen mineralization is influenced by soil saturation conditions and therefore may respond to soil water storage; research in this watershed by Perkins et al. (2004) has determined that soil water retention patterns are altered due to physical disturbance regimes. Land use patterns which influence soil saturation during seasonal fluctuations in litter processing may therefore influence soil nitrogen levels and thereby stream chemistry.

CONCLUSIONS

Most normalized enzyme activities and soil chemical parameters (TOC, WEC, TC, TN, and NH_4-N) paralleled soil moisture, with higher levels on the upland edge of the riparian area. In contrast, peptidase activity and groundwater TKN in fall had the opposite pattern with higher activities and concentration in near stream soils. Because of high variance, only NH_4-N differences were significant. Peak TKN levels in water were closely associated with time of leaf fall, with a secondary peak occurring in March and April. The microbial parameters were most closely related to soil moisture, TC, WEC, TOC, and TP. Total N and extractable NH_4-N were not strongly correlated to any of the other parameters, nor were soil peptidase levels. Log transformed enzyme activities showed strong linear relationships with soil water content, with the exception of peptidase, the activity of which was better described by a quadratic relationship. Peptidase levels in soils of greater than 30% soil moisture were significantly different between December and June soil samples.

REFERENCES

Anderson, J. M., 1976. An ignition method for determination of total phosphorus in lake sediments. Water Research 10, 329-331.

Bhat, S., J. Jacobs, K. Hatfield, and J. Prenger. 2004. Ecological indicators in forested watersheds in Fort Benning, GA: relationship between land use and stream water quality. Ecol. Indicat. In Review.

Bishop, K., C. Pettersson, B. Allard, and Y.-H. Lee. 1994. Identification of the riparian sources of aquatic dissolved organic carbon. Environ. Int. 20:11-19.

Bryant, M. L., S. Bhat, and J. M. Jacobs. 2004. Throughfall characterization and modeling for heterogenous forest communities in the Southeastern U.S. J. Hydro. In Press.

Burns, R.G. 1982. Enzyme activity in soil: location and a possible role in microbial ecology. Soil Biol. Biochem. 14:423-427.

Chrost, R.J. 1991. Environmental control of the synthesis and activity of aquatic microbial ectoenzymes. p. 25-59. In R.J. Chrost (ed.). Microbial enzymes in aquatic environments. Springer-Verlag, New York.

Chrost, R.J. and H.J. Krambeck. 1986. Fluorescence correction for measurements of enzyme activity in natural waters using methylumbelliferyl-substrates. Arch. Hydrobiol. 106:79-90.

Fisher, S.G., and G.E. Likens. 1973. Energy flow in Bear Brook, New Hampshire: an integrative approach to stream ecosystem metabolism. Ecological Monographs 43:421-439.

Godde, M., and R. Conrad. 1998. Simultaneous measurement of nitric oxide production and consumption in soil using a simple static incubation system, and the effect of soil water content on the contribution of nitrification. Soil Biol. Biochem. 30:433-442.

Hoppe, H.G. 1993. Use of fluorogenic model substrates for extracellular enzyme activity (EEA) measurement of bacteria. p. 423-431. *In* P.F. Kemp et al. (eds.) Handbook of methods in aquatic microbial ecology. Lewis Publishers, Boca Raton, FL.

Huggenberger, P., E. Hoehn, R. Beschta, and W. Woessner. 1998. Groundwater control on riparian/fluvial systems. Freshwater Biol. 40:407-425.

Hupp, C.R. 2000. Hydrology, geomorphology, and vegetation of coastal plain rivers in the south-eastern USA. Hydrol. Process. 14:2991-3010.

Kuo, S., 1996. Phosphorus. In: Bigham, J.M. (Ed.), Methods of Soil Analysis. Part 3, Chemical Methods. Soil Science Society of America, ASA, Madison, Wisconsin. pp. 891.

Martens, R. 1995. Current methods for measuring microbial biomass C in soil: Potentials and limitations. Biol. Fertil. Soils 19:87-99.

Mulholland, P.J. 1992. Regulation of nutrient concentrations in a temperate forest stream: Roles of upland, riparian, and instream processes. Limnol. Oceanogr. 37:1512-1526.

Mulvaney, R.L. 1996. Nitrogen-Inorganic Forms. p. 1123. In Methods of Soil Analysis, Part 3. Chemical Methods,. J. M. Bigham, (ed.) SSSA-ASA. Madison, Wisconsin, USA.

Naiman, R.J., and H. Décamps. 1997. The Ecology of Interfaces: Riparian Zones. Annu. Rev. Ecol. Syst. 28:621-658.

Nelson, D.W. and L.E. Sommers. 1996. Total Carbon and Total Nitrogen. p. 973. In Methods of Soil Analysis Part 3 Chemical Methods. J.M. Bigham (ed) SSSA ASA. Madison, Wisconsin.

Ohte, N., N. Tokuchi, and M. Suzuki. 1997. An in situ lysimeter experiment on soil moisture influence on inorganic nitrogen discharge from forest soil. J. Hydrol. 195:78-98.

Perkins, D.B., N. W. Haws, B. S. Das, P.S.C. Rao, and J.W. Jawitz. 2004. Mechanized Military Training Impacts on Hydraulic Characteristics of Ridgetop Soils in a Forested Watershed. J. Environ. Quality. In review

Pusch, M., D. Fiebig, I. Brettar, H. Eisenmann, B.K. Ellis, L.A. Kaplan, M.A. Lock, M.W. Naegeli, and W. Traunspurger. 1998. The role of micro-organisms in the ecological connectivity of running waters. Freshwater Biol. 40:453-495.

Sinsabaugh, R.L., and D.L. Moorhead. 1994. Resource allocation to extracellular enzyme production: a model for nitrogen and phosphorus control of litter decomposition. Soil Biol. Biochem. 26:1305-1311.

Sinsabaugh, R.L., R.K. Antibus, and A.E. Linkins. 1991. An enzymatic approach to the analysis of microbial activity during plant litter decomposition. Agric. Ecosyst. Environ. 34:43-54.

Sinsabaugh, R.L., S. Findlay, P. Franchini, and D. Fischer. 1997. Enzymatic analysis of riverine bacterioplankton production. Limn. Oceanogr. 29-38.

Smith, K.A., T. Ball, F. Conen, K.E. Dobbie, J. Massheder, and A. Rey. 2003. Exchange of greenhouse gases between soil and atmosphere: interactions of soil physical factors and biological processes. European J. Soil Sci. 54:779-791.

Stark, J.M., and M.K. Firestone. 1995. Mechanisms for soil moisture effects on activity of nitrifying bacteria. Appl. Environ. Microbiol. 61:218-221.

USDA, 1997. Natural Resources Conservation Service Major Land Resource Area Boundaries. http://www.nrcs.usda.gov/technical/land/mlra/

Table 1. Correlation matrix for soil physical / chemical parameters and enzyme activities for bottomland soils. Soil moisture (% water), ash content, total P, C, N, WEC, extractable NH_4-N, TOC, and soil enzymes were measured on all bottomland samples from December 2002 and June 2003

	% water	Ash	TP	TN	TC	WEC	K_2SO_4 ext. NH_4-N	K_2SO_4 ext. TOC	APA	BG	Pep	Dehyd
pH	**-0.5384**	0.4107	-0.3648	-0.2175	-0.3802	-0.3698	-0.1449	-0.5323	-0.4776	-0.3539	0.3121	-0.4674
% water		**-0.8621**	**0.7833**	0.4049	**0.762**	0.542	0.4623	**0.7385**	**0.8768**	**0.7967**	0.0812	**0.8487**
Ash			**-0.8274**	-0.3997	**-0.9151**	**-0.6787**	-0.4726	**-0.7866**	**-0.8496**	**-0.859**	-0.1612	**-0.9289**
TP				0.2957	**0.6529**	0.3797	0.5007	**0.7721**	**0.6313**	**0.6371**	0.0523	**0.6979**
TN					0.4776	0.3276	0.33	0.3113	0.4155	0.3872	0.0572	0.38
TC						**0.7171**	0.4625	**0.7032**	**0.7942**	**0.7953**	0.1553	**0.884**
WEC							0.3513	**0.6651**	**0.677**	**0.626**	0.0901	**0.703**
K_2SO_4 NH_4-N								0.4376	0.4058	0.3945	0.1166	0.4884
K_2SO_4 TOC									**0.7263**	**0.6261**	-0.0158	**0.7**
APA										**0.8688**	0.1998	**0.8382**
BG											0.2152	**0.8247**
Pep												0.1529

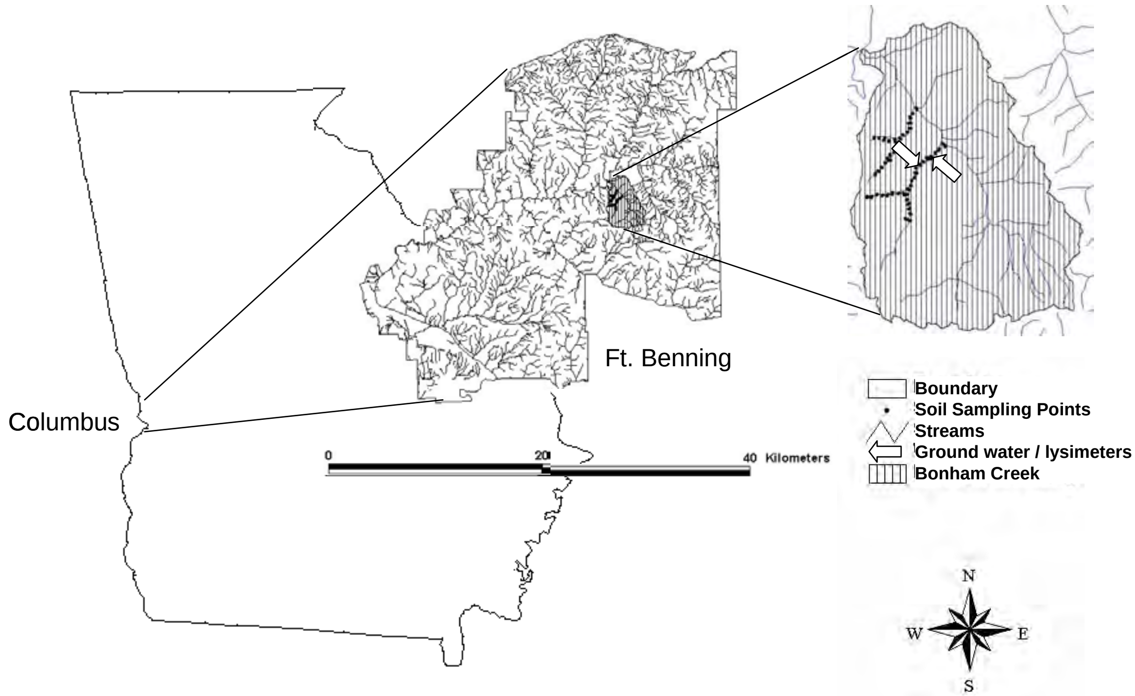


Figure 1.
Two 2nd order stream watersheds within the Ft. Benning Military Reservation (Bonham-1 upper left, Bonham -2 lower right) with soil (triangles) and groundwater (arrows) sampling sites indicated. Inset shows location of Bonham watershed within Ft. Benning in western Georgia.

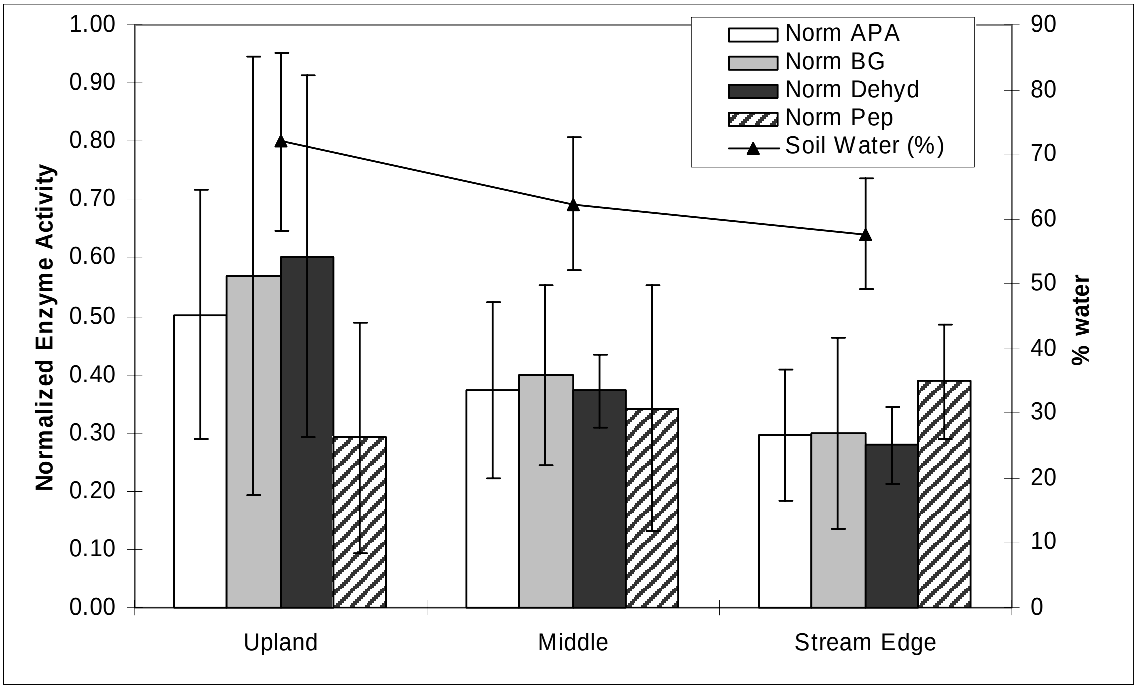


Figure 2.
Mean normalized soil enzyme activities and soil moisture levels from three transects perpendicular to Bonham-2 stream flow. Error bars indicate one standard deviation. Enzyme activities were normalized to the highest value for each enzyme from all sites. Two soil samples (0-10 cm depth) were composited from within 1 m^2 at approximately 5, 10 and 15 m from streambank. APA, acid phosphatase activity; BGA, β-glucosidase; Pep, peptidase; Dehyd, dehydrogenase.

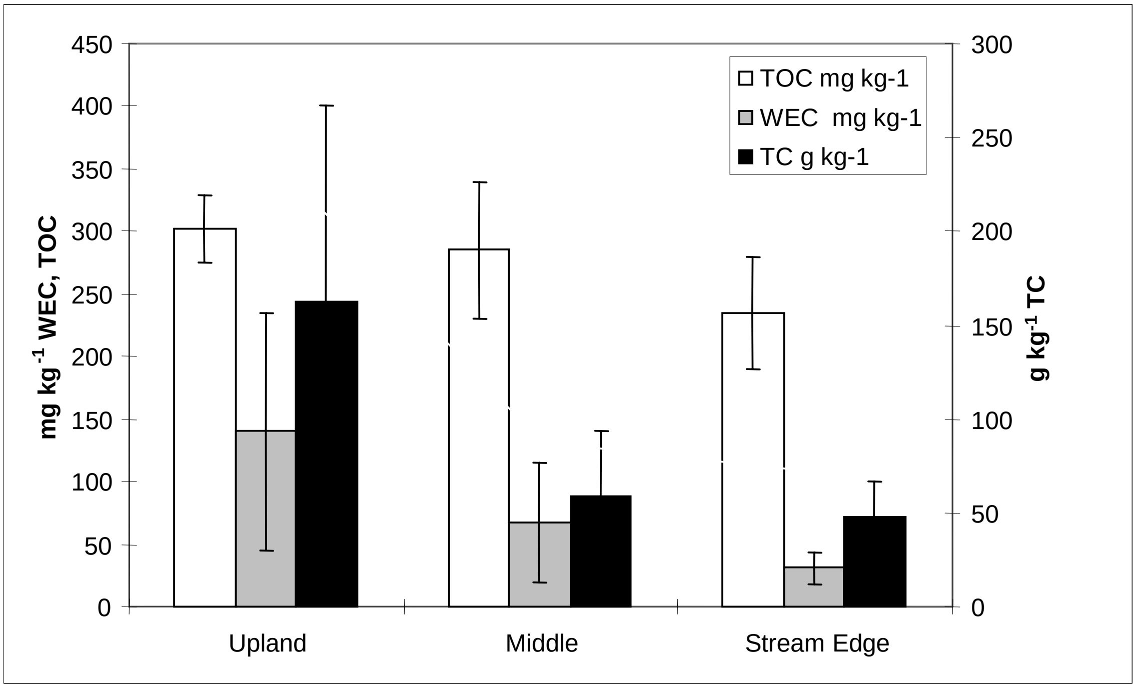


Figure 3.
Total organic carbon (TOC), water extractable carbon (WEC), and total carbon (TC) from transects perpendicular to Bonham-2 stream flow. Values are means of three transects; error bars indicate one standard deviation. Two soil samples (0-10 cm depth) were composited from within 1 m^2 at approximately 5, 10 and 15 m from streambank. Units are mg Kg^{-1} (TOC and WEC) and g Kg^{-1} (TC).

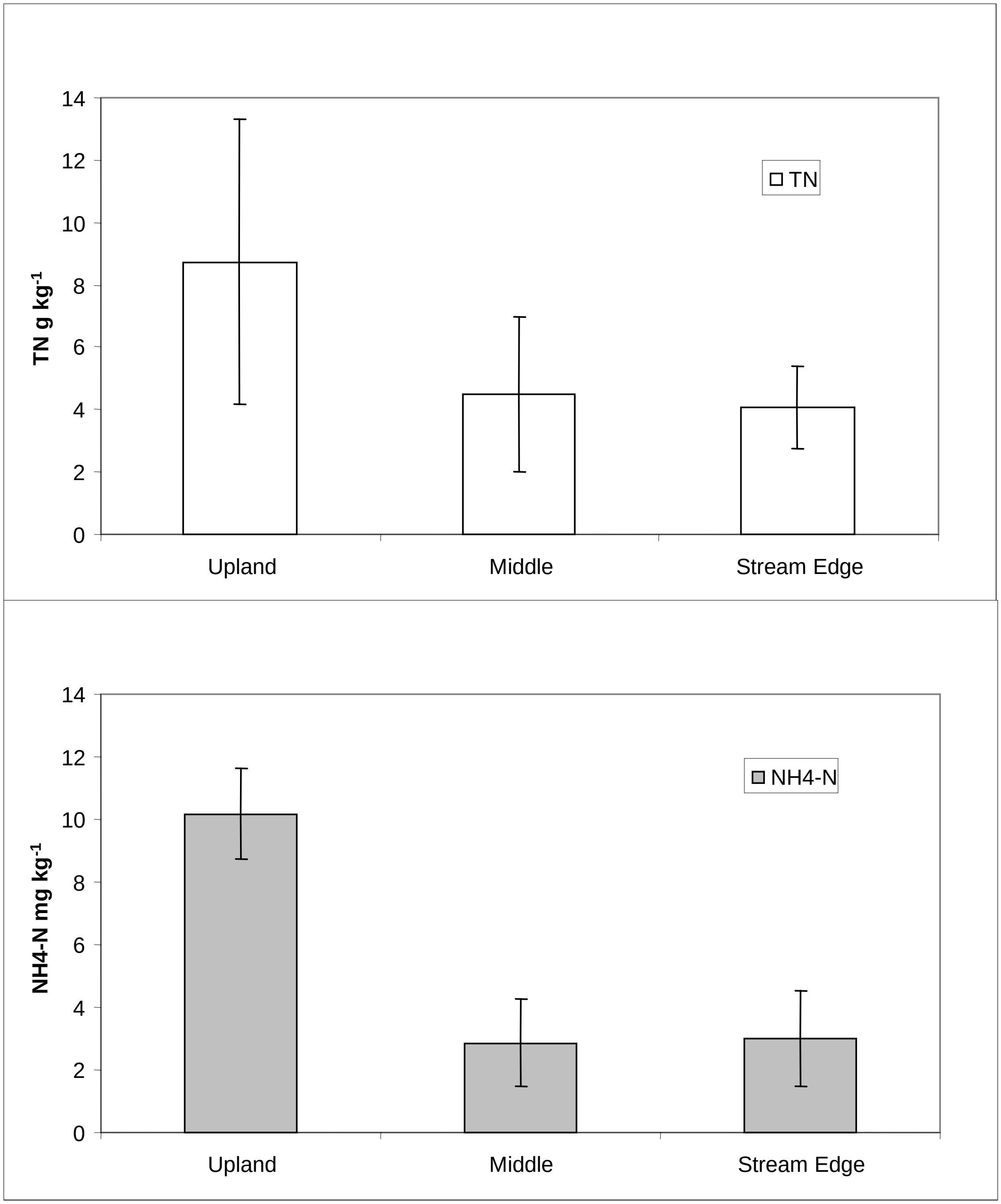


Figure 4.
Total nitrogen (TN) and NH_4-N from transects perpendicular to Bonham-2 stream flow. Values are means of three transects; error bars indicate one standard deviation. Two soil samples (0-10 cm depth) were composited from within 1 m^2 at approximately 5, 10 and 15 m from streambank. Units are mg Kg^{-1} (NH_4-N) and g Kg^{-1} (TN).

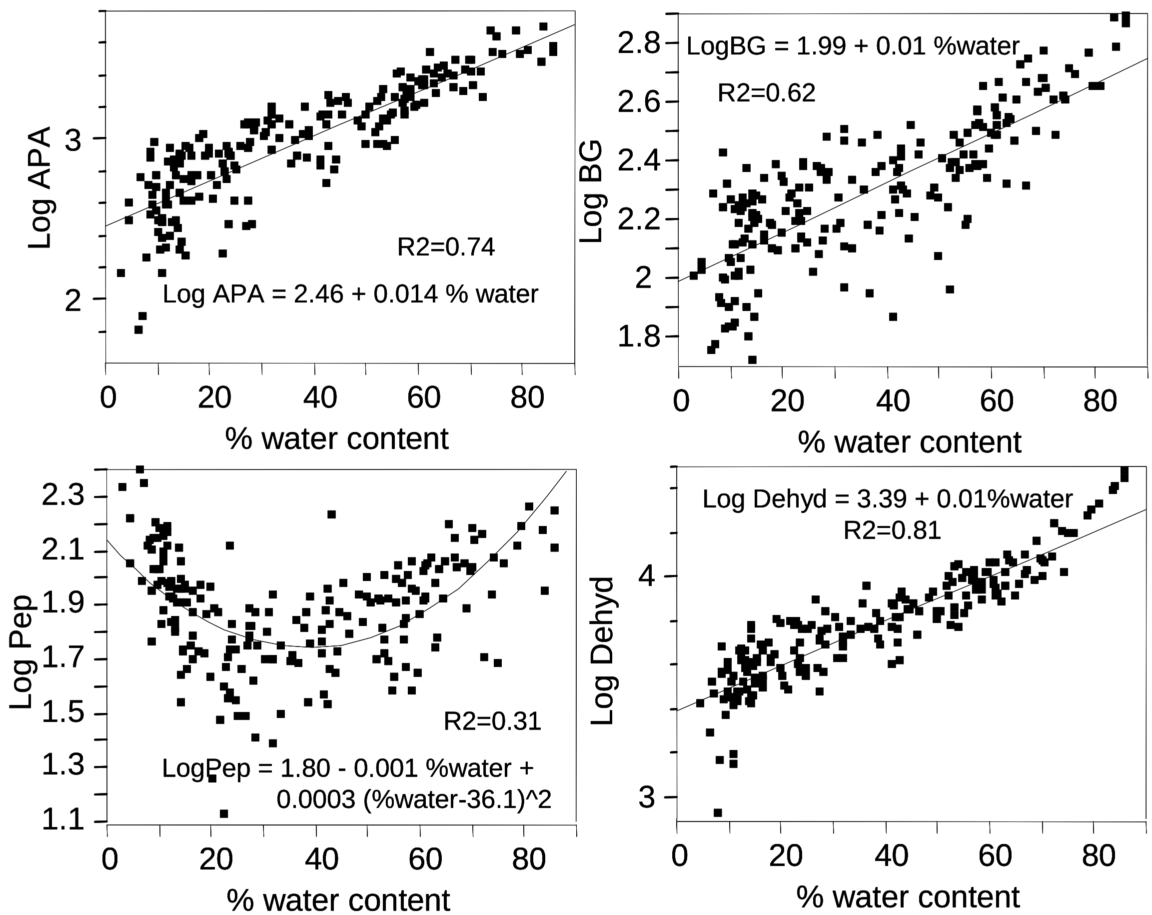


Figure 5.
Microbial enzyme levels relative to % soil moisture. Two soil samples (0-10 cm depth) were composited from within 1 m^2 at approximately 80 m intervals along transects 5 meters from stream bank on either side of Bonham-1 and -2. Bonham-1 and -2 were sampled in December 2002 (n=147), and a subset of Bonham -2 (n=52) sites were resampled in June 2003.

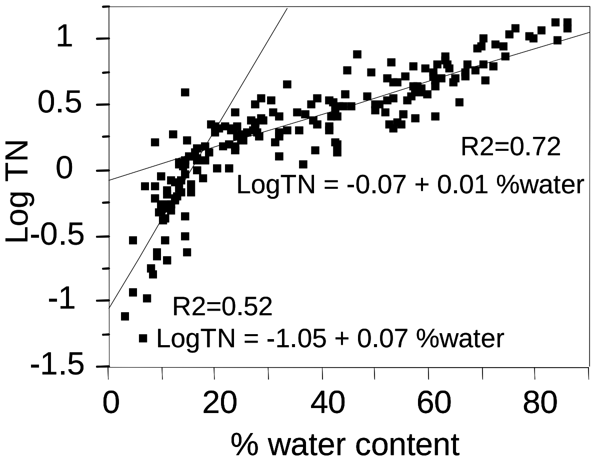


Figure 6.
Log total nitrogen versus % soil moisture. Two soil samples (0-10 cm depth) were composited from within 1 m^2 at approximately 80 m intervals along transects 5 meters from stream bank on either side of Bonham-1 and -2. Bonham-1 and -2 were sampled in December 2002 (n=147), and a subset of Bonham -2 (n=52) sites were resampled in June 2003.

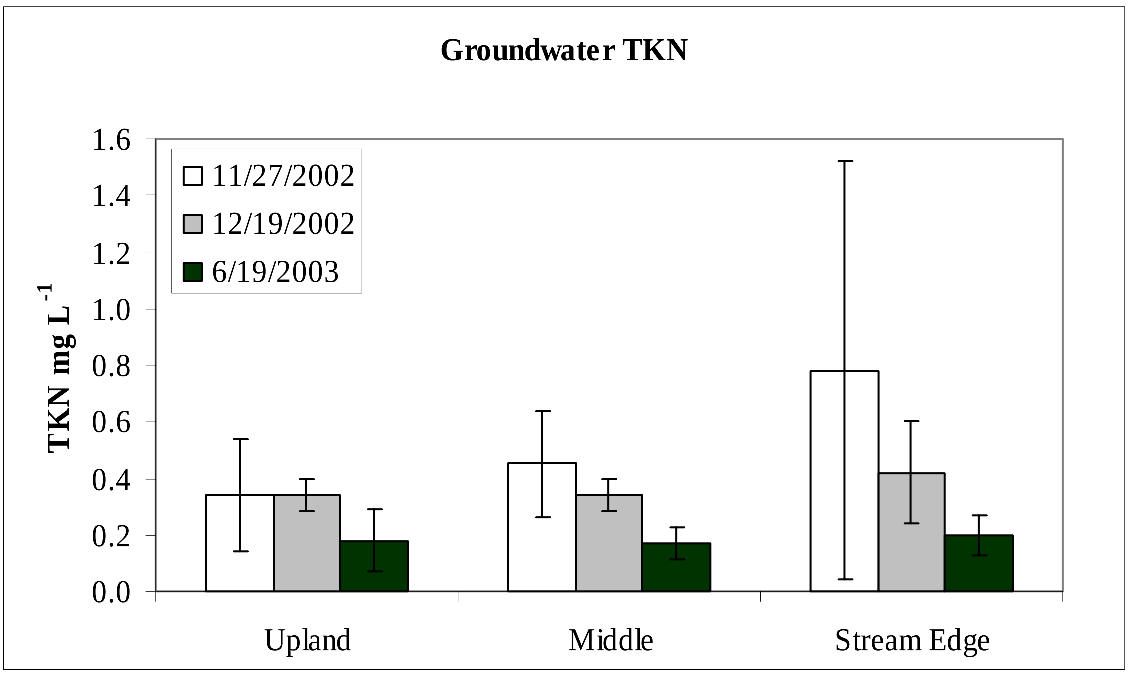


Figure 7.
Groundwater nitrogen levels in Bonham 2 watershed. Ground water from transects perpendicular to Bonham 2 (see Figure 1) were sampled monthly and analyzed for TKN (total Kjeldahl nitrogen). Data are means of values for groundwater wells beginning at about 5 m from stream bank and spaced approximately 10 m apart for upland edge, middle transect, and near stream.

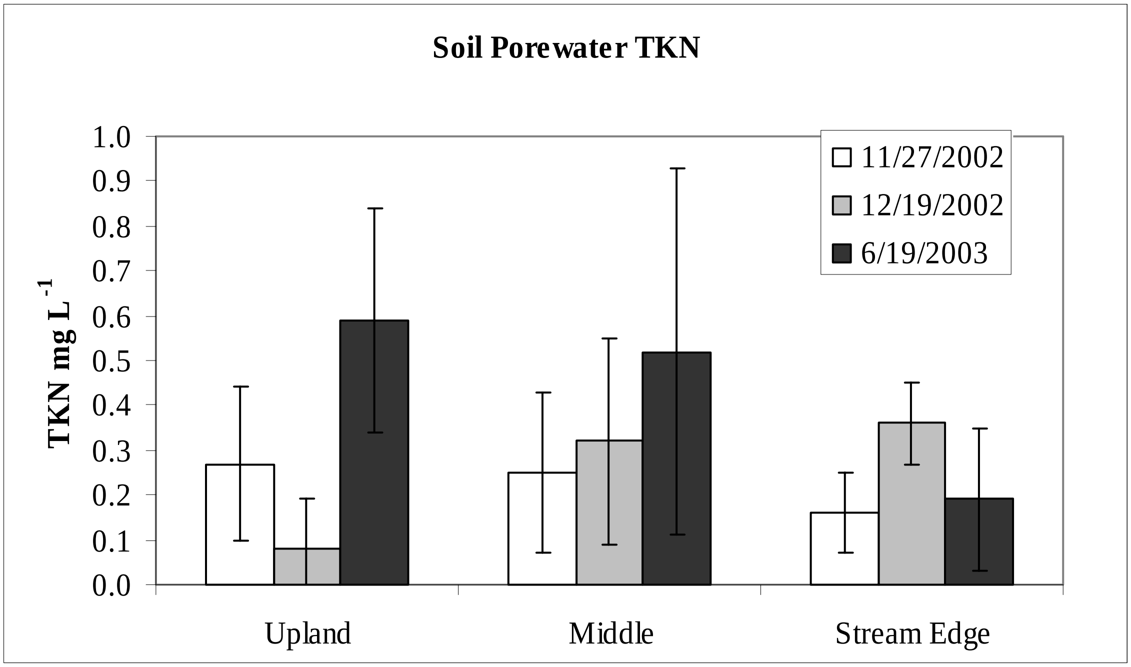


Figure 8.
Soil water nitrogen levels at 20 cm depth in Bonham 2 watershed. Soil water from transects perpendicular to Bonham 2 (see Figure 1) were sampled monthly and analyzed for TKN. Data are means of values for soil water at depth 20 cm from the surface beginning at about 5 m from stream bank and spaced approximately 10 m apart for upland edge, middle transect, and near stream.

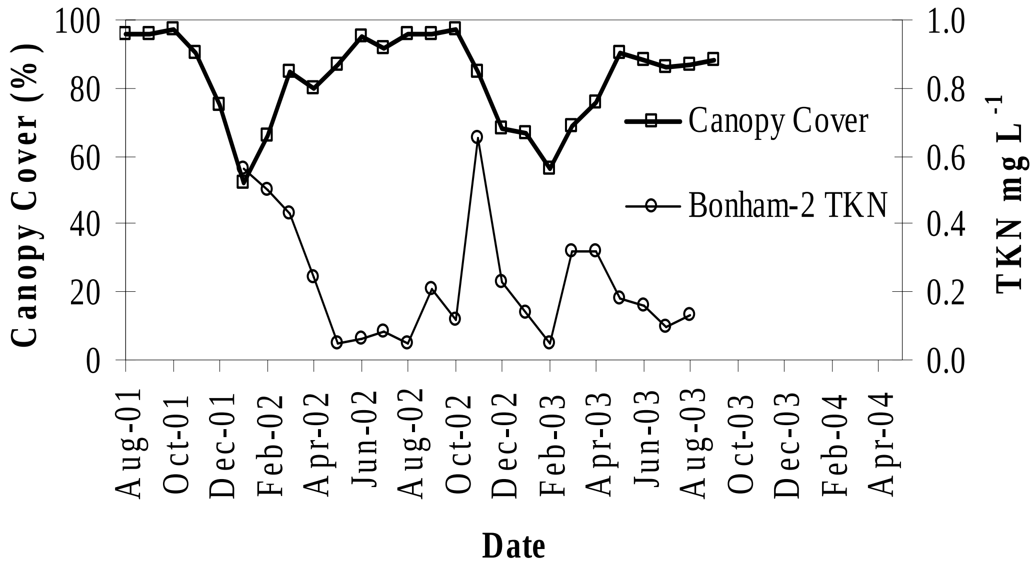


Figure 9.
Canopy cover and stream TKN values for Bonham 2 for the period January 2001 to September 2003.

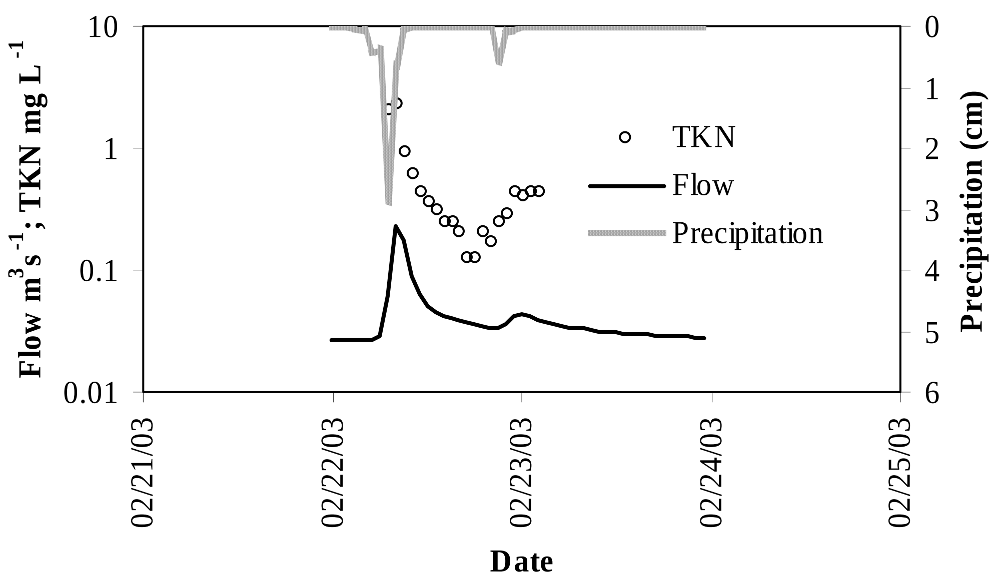


Figure 10.
Bonham-2 stream flow ($m^3\ sec^{-1}$), precipitation (cm), and stream TKN levels ($mg\ L^{-1}$) for a storm event, February 2003.

Modeling and Synthesis

- **3.5.1 Multivariate statistical analyses were applied to 20 biogeochemical parameters in order to discriminate samples based on landscape position, vegetation type, watershed of origin and disturbance class.**

- **3.5.2 Near Infrared Reflectance Spectroscopy (NIRS) for soil analysis is rapid, low-cost technique for determination of several individual soil biogeochemical properties and direct evaluation of derived soil quality metrics or indices.**

3.5.1

Multivariate analysis of soil biogeochemical parameters for assessment of ecological condition. Dabral, S., W. D. Graham, J.P. Prenger, and W. F. DeBusk.

ABSTRACT

Multivariate statistical analyses were applied to 20 biogeochemical parameters measured on over 550 soil samples taken from a range of landscape, vegetation, watershed, and disturbance regimes at the Ft. Benning military installation near Columbus, Georgia. Principal components analysis identified that the total organic matter present in the soil samples (measured as total carbon, total nitrogen, and total phosphorous) was the dominant contributor of variability between the soil samples. Canonical Discriminant Analysis showed that canonical variables, derived from a training set of 217 pre-classified samples could be successfully used to discriminate samples based on landscape position, vegetation type, watershed of origin and disturbance class. The canonical variables derived for landscape disturbance should provide early indicators of changes in soil quality which may signal impending ecological change. Logistic regression was used to predict the probability of a specific site being disturbed or non-disturbed based on the observed categorical variables and measured biogeochemical variables that were found to effect disturbance. Landscape position and watershed along with total phosphorous, oxalate extractable phosphorous, mehlich extractable phosphorous, total carbon, total nitrogen, KCl extractable ammonium, mehlich extractable calcium and oxalate extractable aluminum were found to be important predictors of land disturbance. The logistic regression model correctly classified 79% of the sites as undisturbed and 83.4% of the sites as disturbed based on the data used to develop the model. Total phosphorous, oxalate phosphorous and total carbon showed high odds ratio indicating that a high concentrations of these variables is a good indicator of lack of disturbance in all watershed and landscape position classes. Total nitrogen, KCl extractable ammonium, and oxalate aluminum showed low odds ratios indicating that a high concentrations of these variables is a good indicator of disturbance.

INTRODUCTION

The development of biological and biogeochemical indicators of ecological condition has been a topic of considerable interest in recent years. While direct measure of chemical and physical properties can provide evidence of environmental perturbation, an increasingly wide array of biological parameters (Dale *et al*. 2002; Hargreaves *et al*. 2003) and processes (Bunn *et al.* 1999; Knoepp *et al*. 2000; Peng *et al*. 2002) have been proposed and studied as diagnostic tools for determining the status and trends of ecosystems, watersheds and other management units. Biogeochemical indicators of environmental change can be more sensitive than chemistry alone; however, temporal or spatial variation may mask differences due to ecological condition (Hargreaves *et al*. 2003). Although the use of indicators for ecological diagnostics is generally limited to qualitative or semi-quantitative characterization, this approach provides a tool for both researchers and resource managers that is simpler than, and potentially as effective as, many of the comprehensive ecological models currently available. Furthermore, indicators potentially may be developed for and applied to any geospatial scale, from microhabitats to regional landscapes, and may consist of a single parameter or multiple parameters, such as the Trophic State Index (Carlson, 1977).

The study reported here was part of the Ecosystem Management Project (SEMP), a component of the Strategic Environmental Research and Development Program (SERDP). SERDP is a collaboration among the Departments of Defense and Energy and the Environmental Protection Agency, which addresses natural resource management at U.S. military installations. Our broad objective was to develop indicators of ecosystem condition and impending ecological change, in support of military land management and sustainability of the military mission. Anthropogenic impacts of interest were primarily related to mechanized military training, but also include other land uses and management practices such as timber production/harvest and controlled burning.

Ecological indicators in terrestrial systems often focus on primary productivity (Peng *et al*. 2002), vegetation communities (Dale *et al*. 2002) or soil quality (Knoepp *et al*. 2000). Indicators of soil quality frequently relate to cycling of soil organic matter (Gregorich *et al*. 1997), an important determinant of nutrient availability and soil structure. The specific purpose of this study was to evaluate a group of soil biogeochemical parameters as indicators of ecological condition of lands utilized for military training and related activities at Fort Benning, Georgia. These potential indicators were primarily associated with nutrient storage and bioavailability in upland and wetland soils, as affected by varying degrees of physical disturbance to both soils and vegetation. The biogeochemical variables selected were chosen as because of their importance as components of the nutrient cycle and therefore their central role in supporting vegetation communities. The losses of soil structure and plant communities are among the initial impacts observed for the types of disturbance in this study.

The current study includes twenty soil-biogeochemical variables measured at 217 locations throughout 5 watersheds in Phase 1 and eighteen variables measured at 152 locations in Phase 2, making this dataset ideal for applying and testing multivariate techniques.

SITE DESCRIPTION

The study area is the Ft. Benning military reservation, located near Columbus in west-central Georgia (see Figure 1)**.** This area lies immediately to the south of the fall line, within the fall line hills district of the coastal plain physiographic province and the Carolina and Georgia sand hills major land resource area. The topography of this area is nearly level to gently sloping ridgetops, moderately steep and steep hillsides, and nearly level valleys along stream channels and other tributaries. Upland soils in the area are primarily well- to excessively-drained Ultisols and Entisols, supporting forests of slash, longleaf and loblolly pines. Sandhill communities, associated with excessively-drained Lakeland soils (Entisol) and featuring longleaf pine, turkey oaks, blackjack oaks and post oaks, are commonly associated with ridgetops in the central and northern portion of Ft. Benning. “Loam hills”, featuring soils of relatively high clay content, occur in a band across the southern portion of the installation. Wetlands and hydric soils are generally restricted to bottomlands along streams and creeks.

Fort Benning Military Installation provides facilities including ranges and maneuver training areas to train soldiers in the science of combined infantry principles, weapon systems, and military tactics. Some 60% of the total land area at Fort Benning is designated as maneuver areas. Military training and forest management at Ft. Benning result in direct removal of or damage to vegetation and disruption of soil structure from mechanized artillery, tracked vehicles, earth moving activities, and tree harvest. The mechanized forces in particular use vehicles that cause soil disturbance and movement that may result in erosion and stream sedimentation. Areas of mechanized training are concentrated in two 3rd or 4th order watersheds, while forestry activities occur throughout most of the study area.

METHODS

Data Collection

Soil samples representing a wide range of military and non-military land uses and anthropogenic disturbance regimes were collected from the Fort Benning study site. Phase I sampling was conducted from January to August, 2000 within 6 watersheds of order 3 or 4, associated with Sally Branch, Bonham, Halloca, Randall, Wolf and Shell Creeks (see Figure 1). Phase 1 soil sampling sites were located along transects which were transverse to the orientation of the mainstream channel, providing approximately uniform coverage of the watershed. Each sample point consisted of a 1 m^2 square plot, within which 5 individual samples were taken in a cross pattern. The individual samples were then composited for analysis as a single sample. Soil was sampled to a depth of 20 cm, using a soil push probe with an inside diameter of 1 inch. The sampling scheme was designed to capture the full range of spatial variability of soil properties within each monitoring unit. Sites along each transect were stratified according to landscape position, such that approximately one third of the sites were located in bottomlands (wetlands), and two-thirds were located in uplands (sideslopes, hilltops and ridges). At each site the dominant vegetation type was noted, and the site was classified according to estimated overall level of disturbance based on visual observation of soil and vegetation disturbance and documented land use. General criteria used to classify site disturbance is summarized in Table 1.

Phase 2 sampling, conducted in December 2000 and June 2001, was designed to characterize well-defined ecological and anthropogenic impact gradients on a smaller scale, and at a greater spatial resolution. In Phase 2, upland transects were sampled in areas of high military disturbance, low disturbance, and planted pines (2 stands ca. 5 years and 12 years). Wetland transects were sampled in watersheds with low and moderate military impact, and a watershed dominated by managed timberland. Soil samples were taken at 20 m intervals from 6 transects in upland areas and 6 in bottomland (wetland) areas. In uplands, each soil sample consisted of a composite of five 20-cm deep subsamples taken by 1-inch diameter soil probe within a 1 m^2 quadrat. Wetland soils were sampled to a depth of 10 cm (December) or 5 cm (June), using a 6.5-cm diameter polycarbonate corer. Each sample represented a composite of three subsamples taken within a 1 m^2 quadrat.

Phase 1 and Phase 2 soil samples were analyzed for pH, ash, total carbon, phosphorus and nitrogen (TC,TP, TN), water-extractable carbon and phosphorus (WEC, WEP), Oxalate extractable phosphorus, iron, and aluminum (Oxal P, Oxal Fe, Oxal Al), Microbial biomass Carbon (MBC), KCl extractable ammonium (KCl NH_4), and Mehlich extractable phosphorus, iron, aluminum, calcium, magnesium, and potassium (Meh P, Meh Fe, Meh Al, Meh Ca, Meh Mg, Meh K). Phase 1 soil samples were also analyzed for Microbial biomass phosphorus and nitrogen (MBP, MBN).

Soil Chemical Analyses

Chemical analyses on Phase 1 and Phase 2 soil samples were performed by the University of Florida Wetlands Biogeochemistry Laboratory. Soil pH was measured in 1:1 soil: water slurry in the lab. Ash content was determined from residue after ashing at 550 ^oC (Anderson, 1976). Total phosphorus (TP) analysis was done by dry ashing followed by dissolution in 6M HCl (Anderson, 1976). Total carbon (TC) and total nitrogen (TN) determination was by dry combustion (Nelson and Sommers, 1996). Water-extractable phosphorus (WEP) and water extractable carbon (WEC) were determined by the method of Kuo (1996). Oxalate extractable phosphorus, iron, and aluminum (Oxal P, Oxal Fe, Oxal Al) analyses were performed by the method of Bertsch and Bloom (1996). Determination of microbial biomass carbon (MBC) was by the fumigation-extraction procedure (Horwath and Paul, 1994) as modified by DeBusk and Reddy (1998). KCl extractable ammonium (KCl NH_4) was determined by the method of Mulvaney (1996). Mehlich extractable phosphorus, iron, aluminum, calcium, magnesium, and potassium (Meh P, Meh Fe, Meh Al, Meh Ca, Meh Mg, Meh K) extract were done by the method of Amacher (1996). Phase 1 soil samples were also analyzed for microbial biomass phosphorus (MBP) by the method of Brookes *et al.* (1982) and microbial biomass nitrogen (MBN) by the method of Sparling *et al.* (1990). Nitrogen, phosphorus and carbon levels were in extractions and were determined by EPA methods 351.2, 365.1 and 415.1 (1993). Mehlich extractable metals were determined by ICP- EPA method 200.7.

Statistical Analyses

The multivariate techniques used in this study were principal component analysis, canonical discriminant analysis and logistic regression. The statistical software package, SAS (Release 8.01) was used to analyze the data. Log-transformed values of the biogeochemical measurements were used in the analyses to re-scale all variables so that they contained similar ranges of variability, to reduce the magnitude of variation of the variables, and to improve normality of the distributions. Each observation was classified according to landscape position (upland and bottomland), disturbance type (low, moderate and severe), watershed of origin (Bonham, Randall, Halloca, Sally Branch, Shell Creek, Wolf) and vegetation type (hardwood/deciduous, pine/evergreen, mixed, herbaceous/shrubs). Two hundred and seventeen observations were used for all statistical analyses after deleting the records with missing data.

For logistic regression twenty biogeochemical variables: pH, ash, total carbon, phosphorus and nitrogen (TC, TP, TN), Water-extractable carbon and phosphorus (WEC, WEP), Oxalate extractable phosphorus, iron, and aluminum (Oxal P, Oxal Fe, Oxal Al), Microbial biomass carbon, phosphorus and nitrogen (MBC, MBP, MBN), KCl extractable ammonium (KCl NH_4), and Mehlich extractable phosphorus, iron, aluminum, calcium, magnesium, and potassium (Meh P, Meh Fe, Meh Al, Meh Ca, Meh Mg, Meh K) were used as continuous independent variables and the vegetation type, landscape position and watershed were used as categorical independent variables to predict the land disturbance condition (dependent variable). To avoid the problem of zero cell frequency, medium and severely disturbed categories were collapsed into one category for the logistic regression model. The dichotomous response was modeled as undisturbed versus disturbed. It was assumed that the binary response of land disturbance (occurrence of disturbance vs. non-disturbance) is correlated with the levels of categorical variables.

The four-dimensional distribution table (table 2A-D) shows the distribution of the data based on land disturbance, landscape position, watershed and vegetation type. Out of the 217 sampling sites, 109 were visually classified as disturbed and 108 undisturbed. Out of 109 sites that were disturbed 99 were in uplands (90%) and 10 in bottomlands (10%). Out of 108 sites that were undisturbed 44 were in uplands (40%) and 64 in bottomlands (60%).

For logistic regression the backward elimination method was used which starts with the full model containing all variables and eliminates any variable not found to be important in predicting disturbance. Statistical significance was judged based on a significance (α) level of 0.20. The effects of fixed variables was seen in terms of population averaged odds ratio, which are calculated by taking the exponent of the estimated parameter values of the fixed part of the model (Hosmer and Lemeshow, 1989). Odds ratio provide a convenient measure for assessing the relative change in the odds of a correct classification given a one-unit change in the explanatory variable. An odds ratio of 1 indicates that no change in the odds of a correct classification is associated with a one-unit change in the explanatory variable. An odds ratio greater than 1 indicates that the odds of a correct classification increases as the variable increases by one unit. Conversely, an odds ratio less than 1 indicates a decrease in the odds of a correct classification when the variable value increases. The deviation of the estimated odds ratio from 1 may be interpretated as representing the "sensitivity" of that land disturbance classs to changes in the biogeochemical and spatial stratification variable. For categorical predictors the odds ratio respresent the comparative values between the corresponding level and the reference level.

To test the prediction power of the fitted logistic regression model, the statistical measures that were used are: sensitivity, specificity and prediction accuracy. Sensitivity is a ratio of the number of correctly classified events (undisturbed) and the the total number of events. Specificity is a ratio of the number of correctly classified non-events (disturbed) and the total number of non-events. The response is predicted to be an event if the estimated probability value based on the fitted logistic regression model is greater than the cutoff value. Both sensitivity and specificity are very sensitive to the cut off value and one cannot improve both by selecting the cut off value. Prediction accuracy was evaluated using: $p \times$ sensitivity + $(1-p) \times$ specificity, where p is the probability of the event (undisturbed) (SAS, 1999).

RESULTS AND DISCUSSION

Principal Component Analysis

Principal component analysis (PCA) was performed on the correlation matrix of the Phase 1 soil-biogeochemical variables to reduce the data set to a few meaningful, composite variables that represent the variability of the 20-biogeochemical parameters measured at the Fort Benning military base. Table 3 summarizes the correlation matrix used to develop the principal components, and variables with correlations greater than 0.7 are highlighted in bold in the table. The strongest correlations were observed between TC and TN (0.93), Meh Ca & Meh Mg (0.91), Meh Mg & Meh K (0.89), TN and TP (0.78), Meh Ca & Meh K (0.78), and Oxal Fe and TP (0.71). The correlations between TC-TN and TN-TP indicate that most of the N (as expected) and much of the P in these samples are associated with organic matter. On the other hand, the relationship of Oxalate Fe and TP demonstrates a strong interaction of Fe with P such that Fe may exert significant control over the availability of P for microbial or plant growth.

Table 4 summarizes the mean and standard deviation of the log variables and the PCA results including the loadings (coefficients multiplying each original variable to form the PC) and the eigenvalue of each PC. The amount of variability explained by each PC depends on the relative value of its eigenvalue with respect to the total sum of eigenvalues (Jackson, 1991). The first five principal components were able to explain 74.20% of the total variation of the original data set. These five principal components are uncorrelated to each other and therefore do not have any overlap of information. PC 1 explained 35.87% of the total variance. Variables that scored high in the first principal component were: TC, TN, and TP. This indicates that the total amount of organic matter present in the soils accounts for a large portion of the variability between the observations. PC 2 explained 15.9% of the total variance and showed high loadings of WEP, WEC, Oxal P, and Meh Ca, Mg, and K. This group of variables represents the variability in bioavailable nutrients and metals in the soil samples. The third, fourth and fifth principal component explained 10.9%, 6.5% and 5.1% of the total variance and generally showed high loadings of pH, WEP, WEC, Meh Al and P, Oxal Al, and MBP and C. This likely indicates that the general association of higher microbial biomass with the more highly organic and acidic bottomland soils, and the Al and P in clay mineral accumulations in the bottomlands, account for a small but significant portion of the variability among samples.

Canonical Discriminant Analysis

Canonical discriminant analysis was used to derive a small number of new variables that maximize the difference between various pre-defined sub-populations. Like PCs, these new variables, termed canonical variables, are linear combinations of original measurements. Canonical discriminant analysis was applied to determine whether soil biogeochemistry varied significantly due to disturbance type (low, moderate and severe), landscape position (bottomland and upland), watershed (Bonham, Randall, Haloka, Wolf, Sally Branch and Shell Creek) and vegetation type (Hardwood/Deciduous, Mixed, Pine/Evergreen and Herbaceous/Shrubs).

The canonical discriminant analysis for disturbance level showed that the first 2 canonical variables were statistically significantly different than zero according to the Wilks' Lambda test. Figure 2 shows the observations plotted against the first 2 canonical variables. This figure shows there is a good separation between the observations coming from the low, moderate and severely disturbed sites and therefore it is possible to discriminate the level of disturbance using the 20 soil biogeochemical variables. Note that Figure 2 indicates that there is a continuous gradation and some overlap from low to medium to severe disturbance sites. Therefore classification into 3 discrete levels may be an oversimplification, but relative position along the canonical variable axes should nevertheless be a good indicator of disturbance level.

Table 5 shows the canonical scores for the first 2 canonical variables and indicates that canonical variable 1 is dominated by TC, TP, Oxal Al, Meh Ca and pH. Low disturbance sites exhibit high concentrations of TC and TP, indicating higher accumulations of organic matter. Severely disturbed sites show low concentrations of nutrients and high Oxal Al and pH, an indication of higher levels of inorganic soils or minerals. Canonical variable 2, which further separates the severely disturbed sites from the low to moderately disturbed sites is dominated by TC and TN content, with severely disturbed sites having low TC and high TN (note that the average value of log TN is negative).

The canonical discriminant analysis for landscape position showed that the first canonical variable was significantly different than zero, and able to discriminate between bottomlands and uplands. Figure 3 shows the plot of the observations against the first canonical variable for landscape position. This plot shows that the first canonical variable effectively separates the uplands from the bottomlands. Table 6 shows that the variables that dominate the first canonical variable based on the landscape position were: TC and TN with bottomlands showing higher TC and TN.

Canonical discrimination based on watershed of origin showed that the first four canonical variables were significantly different than zero. Figure 4 shows the data plotted against the first and the second canonical variable based on the watershed. This figure shows clear separation between data from Bonham, Haloka and Shell Creek, but extensive overlap between data from Randall and Wolf. Figure 5 shows that the third canonical variable is able to discriminate the Wolf data from that of Randall. The scores on the first four canonical variables based on watershed are shown in Table 7. High positive canonical coefficients for WEP and Oxal Fe, and high negative canonical coefficients for MBC, Meh Fe and Ca dominate canonical variable 1, which separates the observations from the Bonham watershed from the remaining watersheds. Canonical variable 2, which separates observations from the Haloka watershed from those from the Shell watershed, is dominated by high positive coefficients for Oxal P, MBC, Meh Mg and WEC, and high negative coefficients for TN, Meh K and Ca. Canonical variable 3, which provides some separation for the Wolf and Randall watersheds, is dominated by large negative canonical coefficients for TC and MBC.

Canonical discrimination for vegetation type showed that the first 2 canonical variables were significantly different than zero. The scores on the first two canonical variables are shown in Table 8, and Figure 6 shows the plot of the observations against the first and the second canonical variables for vegetation type. Canonical variable 1 separates the deciduous vegetation (H/D) from the other types, and is dominated by high positive canonical coefficients for TN and high negative canonical coefficients for TC. Canonical variable 2 provided good separation between mixed (M) and herbaceous/shrubs (H/S), and is dominated by high positive canonical coefficients for TC and high negative canonical coefficients for TN. Neither canonical variable was able to separate pine/evergreen (P/E) from the mixed vegetation (M).

Logistic Regression Analysis

Since canonical discrimination showed that biogeochemistry differed among watersheds and landscape position as well as disturbance, a logistic regression model was developed which allowed for the influence of these three categorical variables in addition to the 20 continuous biogeochemical concentrations. As discussed previously, for the logistic regression analyses disturbance was collapsed into 2 categories to avoid having classifications with no observations in them.

Table 9 shows the final logistic regression model with regression coefficients and odds ratio for each variable found to be statistically significant at the 0.2 level. The biogeochemical variables that were found to be important are: TC, TP, TN, Oxal P & Al, Meh P & Ca, KCl NH_4 with an interaction term between TC and TN. The important categorical predictor variables were found to be watershed and landscape position. Vegetation type was not found to be an important predictor of disturbance.

The odds of finding an undisturbed site in the uplands was approximately 1/3 of that for bottomlands provided all the other variables were the same (watershed type and 8 biogeochemical variables). Since the military related activities are the primary cause of disturbance, and these activities are typically confined to the uplands at Fort Benning Military Reservation, the chances of having an undisturbed site in uplands are smaller than for bottomlands. The odds of having an undisturbed site in the watershed Wolf was found to be approximately 2 times that of Shell Creek (reference watershed), while the odds of finding an undisturbed site in Bonham, Halloca and Randall were substantially lower than for Shell Creek.

For the continuous variables, the probability of a site being undisturbed increased with an increase in TC by approximately 787 times with the level of landscape position, watershed type and 8 soil biogeochemical variables held constant. The positive coefficient associated with Oxal P and TP indicated that the probability of a site being disturbed increased with an increase in Oxal P content by approximately 447 times and

with an increase in TP by approximately 66 times respectively. Meh P, TN, KCl NH_4 and Oxal Al showed the smallest odds ratio indicating that the higher concentrations of these variables is a good indicator of disturbance.
The final model shown in table 9 was used to predict the probability of a specific site falling into each of the two land disturbance classes (disturbed and undisturbed) based on the conditional information from the important categorical and biogeochemical variables. Figure 7 shows the probability of each of the sampling sites being an undisturbed site. There is a good sepration between disturbed and undisturbed sites at 0.5 cutoff level. Table 10 shows sensitivity, specificity and classification accuracy between the disturbance classes at a probability cutoff of 0.1, 0.5 and 0.9. The best classification between the disturbance classes was achieved at 0.5. The model correctly classified 79% of the sites as undisturbed and 83.4% of the sites as disturbed. The distribution of each correctly classified disturbed and undisturbed site within each landscape position and watershed type is shown in table 11 A-E. It can be seen from table 11 A-E that for all of the watersheds most of the disturbed uplands and undisturbed bottomlands have been correctly classified. However, the undisturbed uplands and disturbed bottomlands are most commonly misclassified.

Phase 2 observations were used to independently test the logistic regression model. Figure 8 shows the probability of a site being undisturbed at the 38 phase 2 sampling sites in Bonham watershed. Out of 32 sites that were disturbed 28 were classified correct. However, the 6 undisturbed sites that were incorrectly classified as being disturbed came from upland areas. As mentioned earlier, for phase 1 sampling 90% of the disturbed sites were in uplands and only 10% in the bottomlands making the correct classification of undisturbed uplands difficult. Since the data used to develop the model had a relatively low frequency of undisturbed uplands the model had difficulty predicting this outcome.

SUMMARY AND CONCLUSIONS

In this study Canonical discriminant analyses showed that the full biogeochemical signature of soil samples reveals significant differences that can be attributed to landscape position, vegetation type, watershed of origin and anthropogenic disturbance level at Ft. Benning. In particular, the ability of canonical discrimination function to discriminate site disturbance at low end of the scale (e.g. between low and moderate impact) is valuable, since this is more difficult by visual field methods. It is beneficial to use canonical variable plots to observe where the unclassified sample falls within the continuum of low-medium-high disturbance samples.

Logistic regression was used to predict the probability of land disturbance based on landscape position and watershed type along with TP, Oxal P, Meh P, TC, TN, KCl NH_4, Meh Ca and Oxal Al. Total phosphorous, oxalate phosphorous and total carbon showed high odds ratio indicating that a high concentrations of these variables is a good indicator of lack of disturbance in all watershed and landscape position classes. Total nitrogen, KCl extractable ammonium, and oxalate aluminum showed low odds ratios indicating that a high concentrations of these variables is a good indicator of disturbance. Logistic regression model correctly classified 79% of the phase 1 sites as undisturbed and 83.4% of the phase 1 sites as disturbed if a 0.5 probability cutoff value was used.

The fact that the soil biogeochemical signatures were useful for discriminating disturbance level suggests that monitoring programs should include soil biogeochemistry as an ecological indicator. Major ecological impacts at the Ft. Benning military reserve are ultimately observable through erosion and loss of vegetation. However routine measurement of soil nutrient content provides a means of assessing soil disturbance before major structural changes in soil strata or vegetation communities take place, and therefore should be a useful indicator of impending change.

REFERENCES

Amacher, M.C., 1996. Double Acid (Mehlich I) Method. p. 757. *In Methods of Soil Analysis Part 3 Chemical Methods.* J.M. Bigham (ed.), SSSA-ASA. Madison, Wisconsin.

Anderson, J. M., 1976. An ignition method for determination of total phosphorus in lake sediments. *Water Research* 10:329-331.

Bertsch, P.M. and P.R. Bloom, 1996 Ammonium Oxalate Extractable Iron and Aluminum. p. 530. *In Methods of Soil Analysis Part 3 Chemical Methods*. J.M. Bigham (ed), SSSA-ASA. Madison, Wisconsin.

Brookes, P.C., D.S. Powlson, and D.S. Jenkinson. 1982. Measurement of microbial biomass phosphorus in soil. *Soil Bio. Biochem.* 14:319-329.

Bunn, S.E., P.M. Davies, and T.D. Mosisch. 1999. Ecosystem measures of river health and their response to riparian and catchment degradation. *Freshwater Biol.* 41, 333-345.

Carlson, R.E. 1977. A trophic state index for lakes. *Limnology and Oceanography*. 22, 361-369.

Dale, V.H., S.C. Beyeler, and B. Jackson. 2002. Understory vegetation indicators of anthropogenic disturbance in longleaf pine forests at Fort Benning, Georgia, USA. *Ecological Indicators* 1, 155-170.

DeBusk, W.F., and K.R. Reddy. 1998. Turnover of detrital organic carbon in nutrient-impacted Everglades marsh. *Soil Sci. Soc. Am. J.* 62:1460-1468.

Gregorich, E.G., M.R. Carter, J.W. Doran, C.E. Pankhurst, and L.M. Dwyer. 1997. *Biological attributes of soil quality*. In: Gregorich, E.G., and M.R. Carter. (Eds.), Soil Quality for Crop Production and Ecosystem Health. Elsevier, New York, pp. 81-114.

Hargreaves, P.R., P.C. Brookes, G.J.S. Ross, and P.R. Poulton. 2003. Evaluating soil microbial biomass carbon as an indicator of long-term environmental change. *Soil Biol. Biochem*. 35, 401-407.

Hosmer, D.W., Lemeshow, S., 1989. *Applied Logistic Regression*. Wiley, New York.

Horwath, W. R., and E. A. Paul. 1994. Microbial biomass. p. 753-773. In R. W. Weaver et al. (ed.) *Methods of soil analysis, Part 2. Microbiological and biochemical properties.* SSSA-ASA. Madison, Wisconsin.

Jackson J.E. 1991. *A User's guide to Principal Components*. Wiles Publishers, New York.

Knoepp, J.D., D.C. Coleman, D.A. Crossley Jr., and J.S. Clark. 2000. Biological indices of soil quality: an ecosystem case study of their use. Forest Ecology and Management 138, 357-368.

Kuo, S. 1996. Phosphorus. In Methods of Soil Analysis. Part 3, Chemical Methods "Availability Indicators" p. 891. J.M. Bigham (ed.), *Soil Science Society of America,* Inc. ASA, Madison, Wisconsin.

Mulvaney, R.L. 1996. Nitrogen-Inorganic Forms. p. 1123. *In Methods of Soil Analysis, Part 3. Chemical Methods,* J. M. Bigham, (ed.) SSSA-ASA. Madison, Wisconsin, USA.

Nelson, D.W. and L.E. Sommers. 1996. Total Carbon and Total Nitrogen. p. 973. *In Methods of Soil Analysis Part 3 Chemical Methods*. J.M. Bigham (ed) SSSA ASA. Madison, Wisconsin.

Peng, C., J. Liu, Q. Dang, X. Zhou, M. Apps. 2002. Developing carbon-based ecological indicators to monitor sustainability of Ontario's forests. Ecological Indicators 1, 235-246.

SAS documentation, version 8.0. 1999. Statistical Analysis Systems, Cary, NC.

Sparling, G.P., C.W. Feltham, J. Reynolds, A.W. West, and P. Singleton. 1990. Estimation of soil microbial C by a fumigation-extraction method: use on soils of high organic matter content and a reassessment of the K_{EC} factor. *Soil Bio.Biochem.* 22:301-307.

U. S. Environmental Protection Agency (USEPA). 1993. *Methods for the determination of inorganic substances in environmental samples*. Environ. Monit. Support Lab., Cincinnati, OH.

Table 1: Landscape Disturbance Classification Scheme

	LOW	**MEDIUM**	**HIGH**
Overstory Vegetation	No disturbance apparent	Minimal disturbance	Extensive tree removal or damage
Understory Vegetation	No or minimal disturbance apparent	Moderate physical impact or removal	Cleared understory
Ground Cover	No or minimal disturbance apparent	Moderate physical impact or patches of bare ground	Extensive bare ground
Soil	No disturbance apparent	Disturbance of O and A horizons, minimal erosion or other soil loss	Extensive soil disturbance, erosion and other soil loss, disruption of natural horizonation
Fire	Minimal (controlled burn, not recent)	Recent controlled burn or more severe burn in recent past	Not used as sole criteria for severe impact

Table2A. Frequency Distribution of Undisturbed Sites in Uplands[8]

Vegetation	Watershed	Bonham	Hal	Ran	Wolf	Shell	Total
Herbaceous/Shrubs		0	0	0	0	0	0
Hardwood/Deciduous		0	0	4	1	5	10
Mixed		6	2	2	4	10	24
Pine/Evergreen		4	0	5	0	1	10
Total		10	2	11	5	16	44

Table2B. Frequency Distribution of Undisturbed Sites in Bottomlands[8].

Vegetation	Watershed	Bonham	Hal	Ran	Wolf	Shell	Total
Herbaceous/Shrubs		0	0	2	0	0	2
Hardwood/Deciduous		9	16	13	4	15	57
Mixed		1	0	0	1	1	3
Pine/Evergreen		2	0	0	0	0	2
Total		12	16	15	5	16	64

Table2C. Frequency Distribution of Disturbed Sites in Uplands[8].

Vegetation	Watershed	Bonham	Hal	Ran	Wolf	Shell	Total
Herbaceous/Shrubs		2	3	4	0	0	9
Hardwood/Deciduous		2	2	4	2	0	10
Mixed		4	20	12	6	6	48
Pine/Evergreen		11	13	4	4	0	32
Total		19	38	24	12	6	99

Vegetation	Watershed	Bonham	Hal	Ran	Wolf	Shell	Total
Herbaceous/Shrubs		0	0	0	0	0	0
Hardwood/Deciduous		3	1	4	1	0	9
Mixed		0	1	0	0	0	1
Pine/Evergreen		0	0	0	0	0	0
Total		3	2	4	1	0	10

Table2D. Frequency Distribution of Disturbed Sites in Bottomlands[8].

8 Watershed Sal was omitted from the analysis due to the missing data.

Table3. Correlation matrix of the 20 Phase 1 biogeochemical variables, Fort Benning, Georgia.

	pH	Ash	TP	WEP	OxalP	MehP	MBP	TC	TN	WEC	MBC	KCl NH_4	MBN	Meh Fe	Meh Al	Meh Ca	Meh Mg	Meh K	Oxal Fe	Oxal Al
pH	1.00																			
Ash	-.02	1.00																		
TP	-.05	-.38	1.00																	
WEP	.01	-.01	.06	1.00																
Oxal P	-.02	-.39	.61	.41	1.00															
Meh P	-.09	-.01	.02	-.02	.08	1.00														
MBP	-.11	-.12	.15	-.18	.10	.20	1.00													
TC	-.20	-.39	.67	.08	.43	-.12	.29	1.00												
TN	-.16	-.45	**.78**	.06	.51	-.13	.29	**.93**	1.0											
WEC	.03	-.46	.08	.31	.34	-.19	.03	.28	.22	1.0										
MBC	-.45	-.35	.47	-.16	.27	.03	.30	.63	.63	.02	1.0									
KClNH_4	0.01	-.46	.42	.40	.58	-.09	-.03	.38	.44	.26	.23	1.0								
MBN	.08	.35	.55	-.30	.23	-.14	.25	.55	.59	.07	.40	.09	1.0							
Meh Fe	-.18	-.37	.48	-.01	.45	-.20	.13	.67	.69	.25	.44	.35	.46	1.0						
Meh Al	-.46	-.23	.38	-.08	.30	.01	.18	.64	.53	.21	.49	.20	.30	.55	1.00					
Meh Ca	.41	-.19	.38	-.22	-.08	.03	.13	.21	.34	-.22	.09	.00	.44	.04	-.20	1.00				
Meh Mg	.23	-.25	.52	-.21	.00	-.01	.16	.34	.47	-.15	.21	.06	.50	.19	-.00	**.91**	1.0			
Meh K	.04	-.29	.58	-.26	.04	.03	.23	.50	.59	-.17	.35	.07	.56	.32	.23	**.78**	**.89**	1.0		
Oxal Fe	.02	-.43	**.71**	.29	.69	-.35	-.03	.55	.66	.25	.31	.55	.41	.59	.21	.22	.33	.32	1.0	
Oxal Al	-08	-.40	.67	-.15	.37	-.04	.07	.54	.56	.03	.38	.28	.46	.34	.50	.30	.42	.52	.55	1.0

Table 4. Mean, Standard Deviation, and PC Loadings of 20 Phase 1 biogeochemical variables

Variable	Mean (of log variable)	Std. Dev (of log variable)	PC 1	PC 2	PC 3	PC 4	PC 5
PH	5.16	**0.55**	**-.0446**	**0.1921**	0.4755	**-.0318**	0.3567
Ash	1.97	**0.06**	**-.2140**	**0.0555**	**-.0923**	**-.1471**	**-.19916**
TP	1.96	**0.30**	.3205	**0.0132**	**0.0810**	**0.1665**	**-.18768**
WEP	-0.94	**0.46**	**-.0079**	-.3308	0.3292	**0.2194**	**-.0918**
Oxal P	1.44	**0.41**	**0.2119**	-.3011	**0.1820**	**0.2969**	**0.0167**
Meh P	-0.01	**0.31**	**-.0405**	**0.0668**	**-.1789**	0.7372	**0.0514**
MBP	0.31	**0.48**	**0.0999**	**0.1006**	**-.2783**	**0.2671**	0.5698
TC	1.10	**0.37**	0.3257	**-.0800**	**-.1187**	**-.0778**	**0.0940**
TN	-.26	**0.42**	0.3489	**-.0357**	**-.0396**	**-.0289**	**0.0542**
WEC	1.76	**0.36**	**0.0772**	**-.2978**	**0.1291**	**-.1912**	0.4865
MBC	2.37	**0.39**	**0.2395**	**-.0387**	-.3238	**0.0517**	**-.06220**
MBN	0.62	**0.39**	**0.2580**	**0.1725**	**-.0373**	**-.1814**	**0.2425**
KCL NH_4	1.49	**0.37**	**0.1846**	**-.2532**	**.2529**	**0.1810**	**-.04849**
Meh Fe	1.70	**0.47**	**0.2656**	**-.1398**	**-.0851**	**-.2454**	**0.0978**
Meh Al	2.43	**0.31**	**0.2116**	**-.1754**	-.3717	**-.0946**	**-.07156**
Meh Ca	1.90	**0.63**	**0.1560**	0.4415	**0.2308**	**0.0712**	**0.0223**
Meh Mg	1.37	**0.66**	**0.2134**	0.3980	**0.1616**	**0.0259**	**-.06289**
Meh K	1.36	**0.42**	**0.2525**	0.3552	**0.0004**	**0.0127**	**-.09660**
Oxal Fe	3.25	**0.52**	**0.2812**	**-.1434**	**0.2825**	**-.1062**	**-.16202**
Oxal Al	3.21	**0.23**	**0.2700**	**0.0522**	**-.0210**	**0.0078**	-.3056
Eigenvalue	**-**	**-**	**7.17**	**3.17**	**2.17**	**1.30**	**1.01**
%Variance explained	**-**	**-**	**35.87**	**15.9**	**10.86**	**6.50**	**5.06**
%Cumulative variance	**-**	**-**	**35.87**	**51.77**	**62.64**	**69.14**	**74.20**

Table 5: Canonical Scores based on Disturbance for the Phase 1 Biogeochemical data

Variable	CAN 1	CAN 2
pH	-0.6160	**0.2104**
Ash	**-0.0111**	**0.3786**
Total Phosphorus	0.4624	**-0.3198**
Water extractable Phosphorus	**0.2442**	**-0.0224**
Oxalate Phosphorus	**0.4170**	**-0.6284**
Mehlich Phosphorus	**-0.2711**	**-0.0068**
Microbial Phosphorus	**0.0619**	**0.2789**
Total Carbon	0.5561	1.8879
Total Nitrogen	**-0.2378**	-1.6925
Water Extractable Carbon	-0.2549	-0.0092
Microbial Carbon	-0.0764	0.2896
KCl extractable Ammonia	-0.2911	0.5234
Microbial Nitrogen	0.3658	-0.0260
Mehlich Iron	-0.0775	0.1166
Mehlich Aluminum	-0.0547	0.6163
Mehlich Calcium	**0.6375**	0.2076
Mehlich Magnesium	-0.2786	0.8354
Mehlich Potassium	0.3415	-0.7149
Oxalate Iron	0.3932	0.3394
Oxalate Aluminum	**-0.6890**	-0.5289

Table 6 Canonical Scores based on Landscape Position for the Phase 1 Biogeochemical data

Variable	**CAN 1**
pH	**-0.4591**
Ash	**0.2010**
Total Phosphorus	**-0.0482**
Water extractable Phosphorus	**-0.1519**
Oxalate Phosphorus	**0.5552**
Mehlich Phosphorus	**-0.1686**
Microbial Phosphorus	0.0211
Total Carbon	**-1.5725**
Total Nitrogen	**2.6686**
Water Extractable Carbon	-0.0143
Microbial Carbon	-0.0580
KCl extractable Ammonia	-0.0793
Microbial Nitrogen	0.1433
Mehlich Iron	0.7076
Mehlich Aluminum	-0.45560
Mehlich Calcium	0.00893
Mehlich Magnesium	0.22314
Mehlich Potassium	-0.1585
Oxalate Iron	-0.0460
Oxalate Aluminum	-0.3331

Table 7. Canonical Scores based on Watershed of origin for the Phase 1 Biogeochemical data

Variable	CAN 1	CAN2	CAN3	CAN4
pH	**-0.0951**	**0.6873**	**0.6575**	**0.1561**
Ash	**0.0651**	-0.2289	**-0.0966**	**0.5367**
Total Phosphorus	**-0.4426**	**-0.2756**	**-0.0312**	**-0.4224**
Water extractable Phosphorus	1.7183	**-0.8161**	**-0.2712**	**-0.6823**
Oxalate Phosphorus	**0.0540**	0.9149	**0.4462**	**1.21610**
Mehlich Phosphorus	**-0.0540**	**-0.1775**	**-0.2360**	**-0.0226**
Microbial Phosphorus	**-0.0031**	**0.26638**	**-0.3285**	**-0.3538**
Total Carbon	**0.4974**	**0.3484**	-0.5967	**1.1193**
Total Nitrogen	**-0.3084**	-0.9242	**0.5687**	**-1.5957**
Water Extractable Carbon	**0.3397**	0.8598	**0.4017**	**-0.3454**
Microbial Carbon	-1.178	1.315	-2.868	**0.548**
KCl extractable Ammonia	**0.3157**	**-0.3447**	**-0.2555**	**1.1976**
Microbial Nitrogen	**-0.1152**	**0.2520**	0.1859	0.0420
Mehlich Iron	**-1.008**	0.4211	0.1880	-0.0976
Mehlich Aluminum	0.3153	0.3472	-0.1707	-0.0147
Mehlich Calcium	**-1.4996**	**-1.2402**	-0.3169	-0.2169
Mehlich Magnesium	1.01180	**1.2984**	0.1366	0.2444
Mehlich Potassium	-0.3438	**-0.9317**	0.7399	0.5737
Oxalate Iron	**1.4156**	-0.6914	-0.1252	-0.3574
Oxalate Aluminum	-0.3713	-0.3164	0.2178	-0.1014

Table 8. Canonical Scores based on Vegetation for the Phase 1 Biogeochemical data

Variable	CAN 1	CAN2
pH	**-0.2536**	**-0.2166**
Ash	0.1957	0.4477
Total Phosphorus	**0.0108**	**0.1654**
Water extractable Phosphorus	**-0.1460**	**-0.166**
Oxalate Phosphorus	**0.3248**	**-0.1017**
Mehlich Phosphorus	**-0.0203**	**-0.5549**
Microbial Phosphorus	**0.1592**	**0.2559**
Total Carbon	-1.4283	3.0180
Total Nitrogen	2.2788	-3.3863
Water Extractable Carbon	-0.1220	-0.50326
Microbial Carbon	0.0284	-0.04500
KCl extractable Ammonia	-0.0837	0.5723
Microbial Nitrogen	-0.1097	0.2673
Mehlich Iron	0.5180	0.3478
Mehlich Aluminum	-0.1293	-0.1953
Mehlich Calcium	-0.1592	0.3915
Mehlich Magnesium	0.5643	0.4240
Mehlich Potassium	-0.2195	-0.3274
Oxalate Iron	0.07016	0.3376
Oxalate Aluminum	-0.3635	-0.0795

Table9. Logistic regression estimates[8].

Parameter	Estimate	Standard Error	p value	Odds ratio
Intercept	-5.762	5.520	0.296	0.00
Uplands	-1.07	0.419	0.10	0.341
Bonham	-1.247	0.745	0.094	0.287
Halloca	-2.482	0.564	<0.0001	0.084
Randal	-0.361	0.501	0.519	0.697
Wolf				
TP	0.857	0.680	0.207	2.358
Oxal P				
	4.202	1.871	0.024	66.82
	6.102	1.551	<0.0001	447.82
Meh P	-3.752	1.257	0.0029	0.023
TC	6.668	2.959	0.024	787.07
TN				
KCl NH_4	-8.371	3.824	0.028	0.00
Meh Ca				
Oxal Al	-3.941	1.118	0.004	0.019
TC*TN				
	1.248	0.578	0.030	3.484
	-6.065	1.905	0.0015	0.002
	2.158	1.682	0.1995	8.654

[8] Watershed Sal was omitted from the analysis due to the missing data.

Table 10. Prediction summary for phase 1 sites8 using logistic regression model at 10%, 50% and 90% cutoff valueφ.

Probability cutoff value	10%	50%	90%
Senstivity[a]	108 (100%)	86 (79.6%)	51 (47.2%)
Specificity[b]	51 (46.7%)	91 (83.4%)	107 (98.16%)
Classification accuracy[c]	52.03%	81.5%	52.29%

8 disturbed class includes both medium and severe; event is the probability of being undisturbed.
number of observations=217; disturbed =108 undisturbed =109
φ the response is predicted to be an event if the estimated value is higher than the cutoff value.
a sensitivity = The proportion of event responses that were predicted correctly.
b specificity =The proportion of non event responses that were predicted correctly.
c classification accuracy = p × sensitivity + (1-p) × specificity, where p is the probability of the event.

Table 11A. The sites that were correctly classified as being disturbed and undisturbed for the watershed Bonham (phase1)8.

	Undisturbed	Disturbed
Uplands	3/10	17/19
Bottomlands	11/12	0/3

Table 11B. The sites that were correctly classified as being disturbed and undisturbed for the watershed Halloca (phase1)8.

	Undisturbed	Disturbed
Uplands	0/2	38/38
Bottomlands	14/16	0/2

Table 11C. The sites that were correctly classified as being disturbed and undisturbed for the watershed Randall (phase 1)8.

	Undisturbed	Disturbed
Uplands	6/11	23/24
Bottomlands	14/15	0/4

Table 11D. The sites that were correctly classified as being disturbed and undisturbed for the watershed Wolf (phase 1)8.

	Undisturbed	Disturbed
Uplands	3/5	11/12
Bottomlands	5/5	0/1

Table 11E. The sites that were correctly classified as being disturbed and undisturbed for the watershed Shell (phase 1)8.

	Undisturbed	Disturbed
Uplands	14/16	2/6
Bottomlands	16/16	0/0

8 **the numerator shows the number of correctly classified sites and the denominator shows the total number of sites in that class.**

Figure 1. Map of Fort Benning Military Installation, Georgia

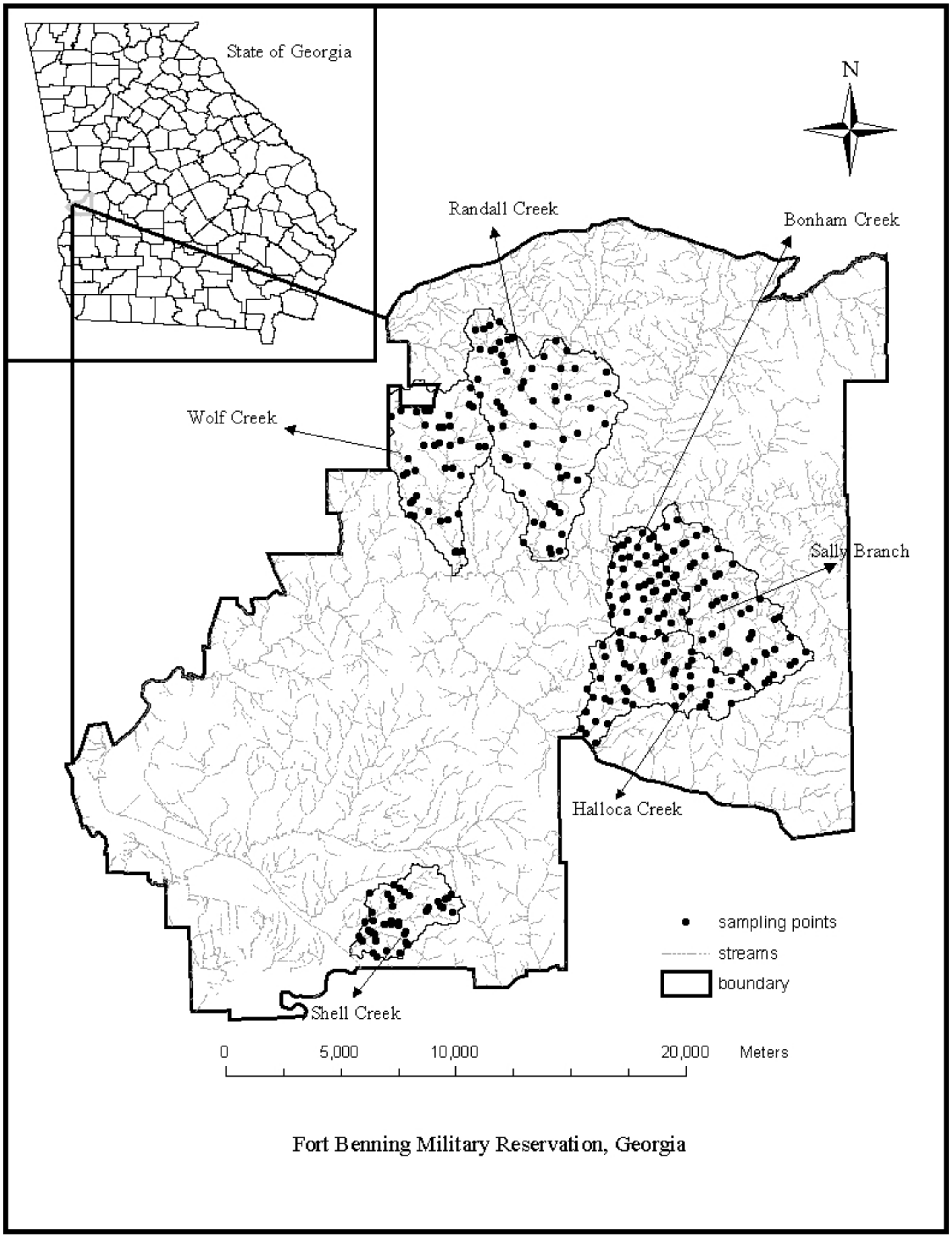


Figure 2. Plot of first and second canonical variables based on disturbance using Phase 1 data, 20 biogeochemical variables

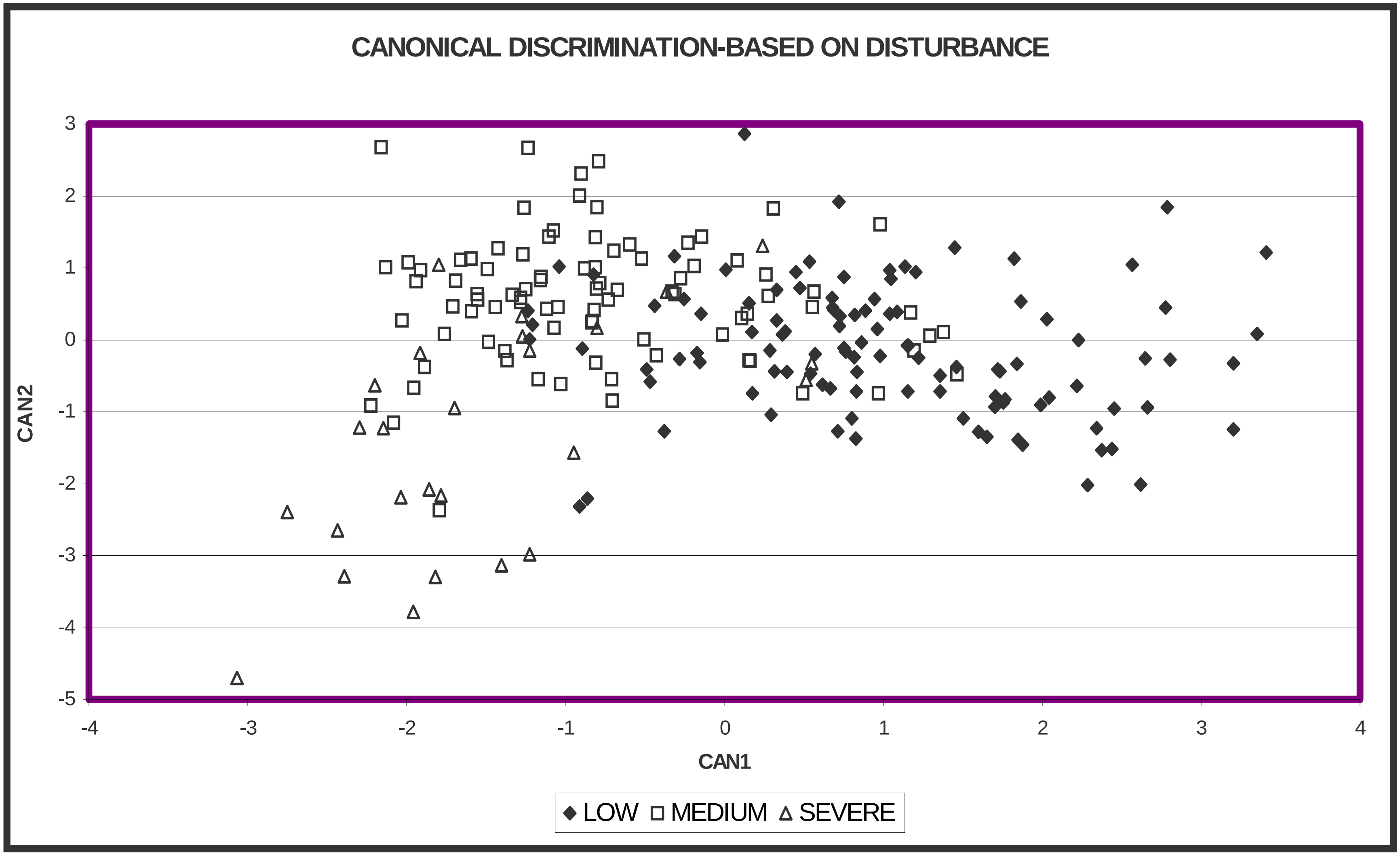


Figure 3. Plot of first canonical variable based on landscape position using Phase 1 data, 20 biogeochemical variables

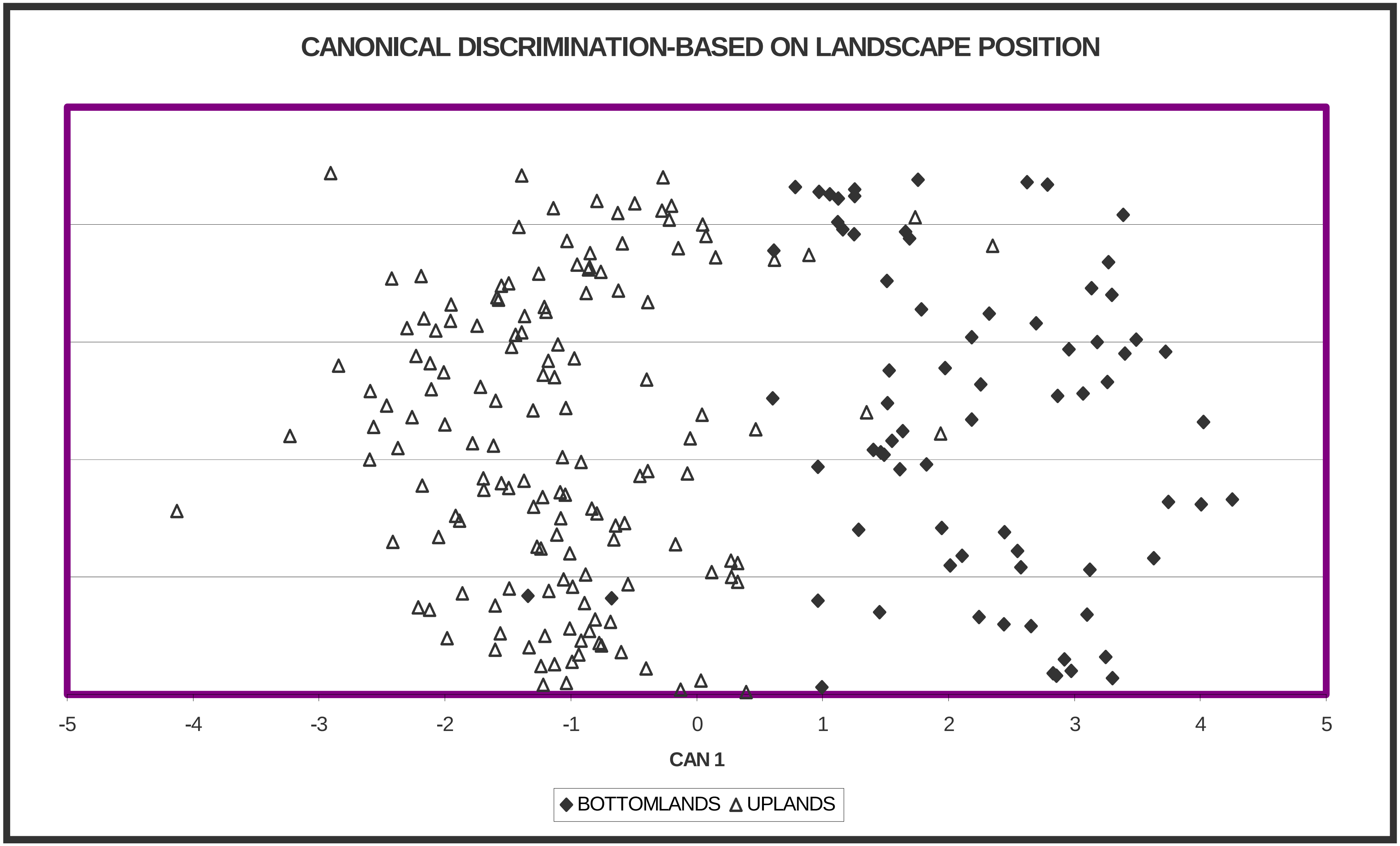


Figure4. Plot of first and second canonical variables based on the watershed using Phase 1 data, 20 soil biogeochemical variables

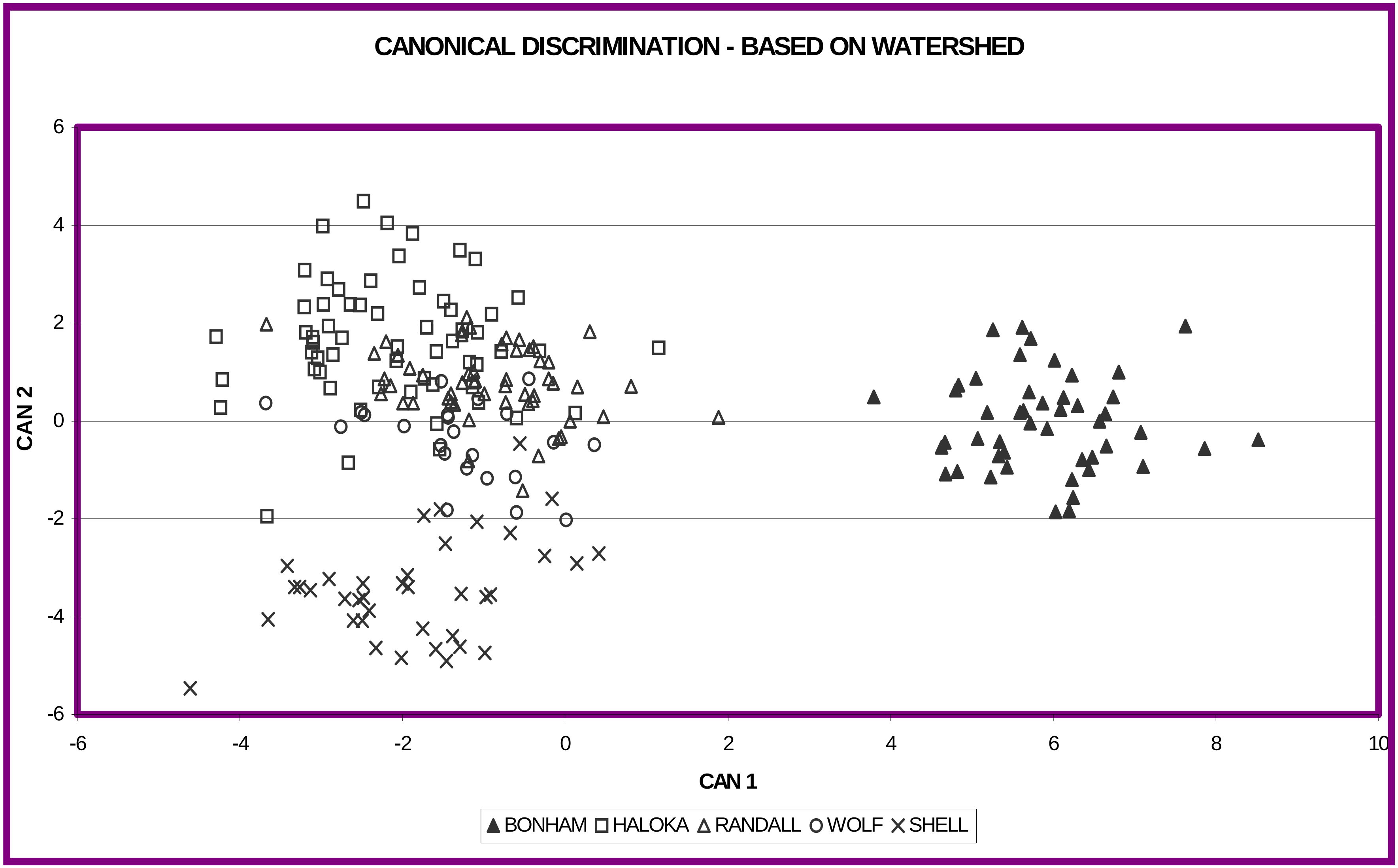


Figure 5. Plot of second and third canonical variables based on the watershed using Phase 1 data, 20 biogeochemical variables

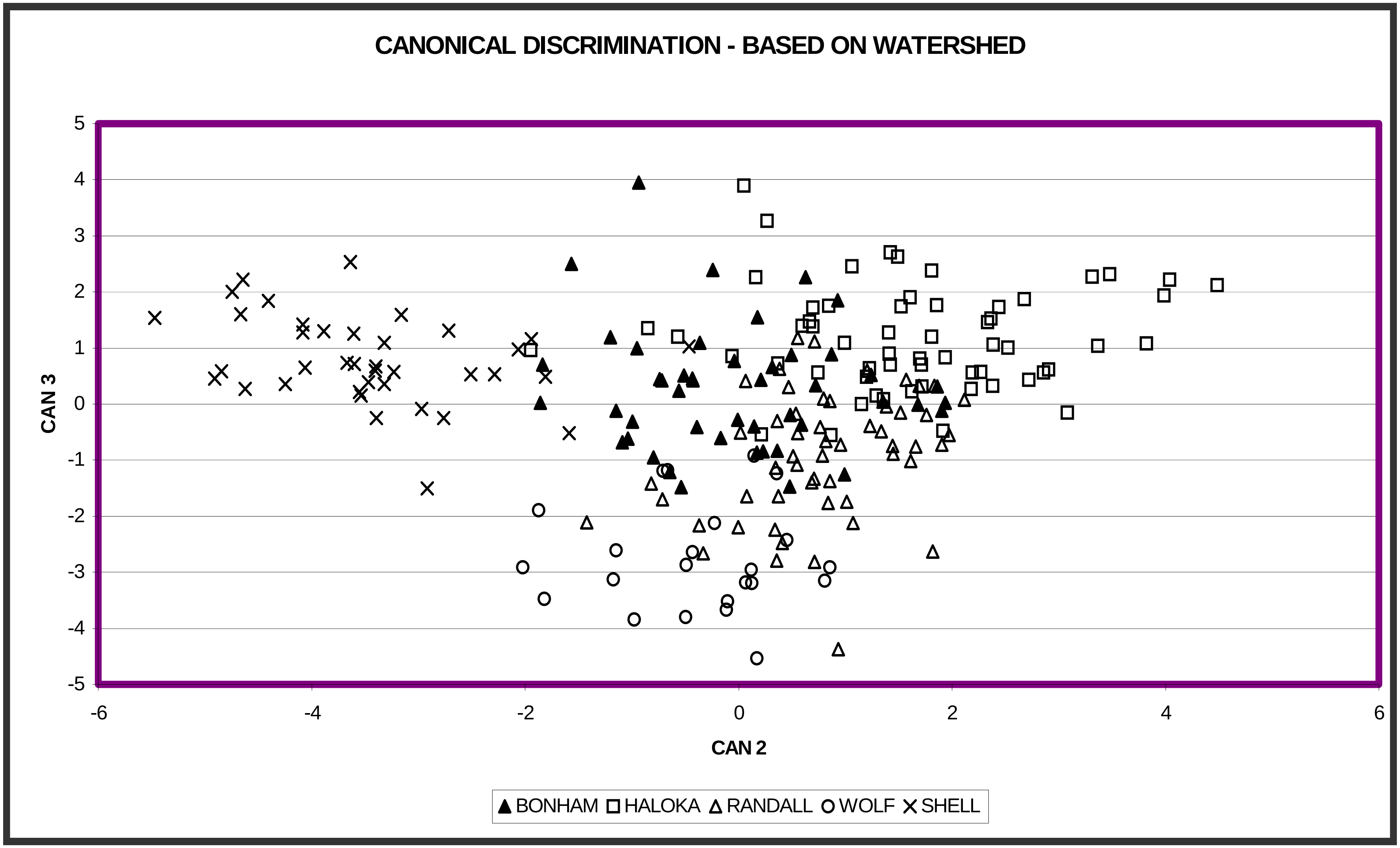
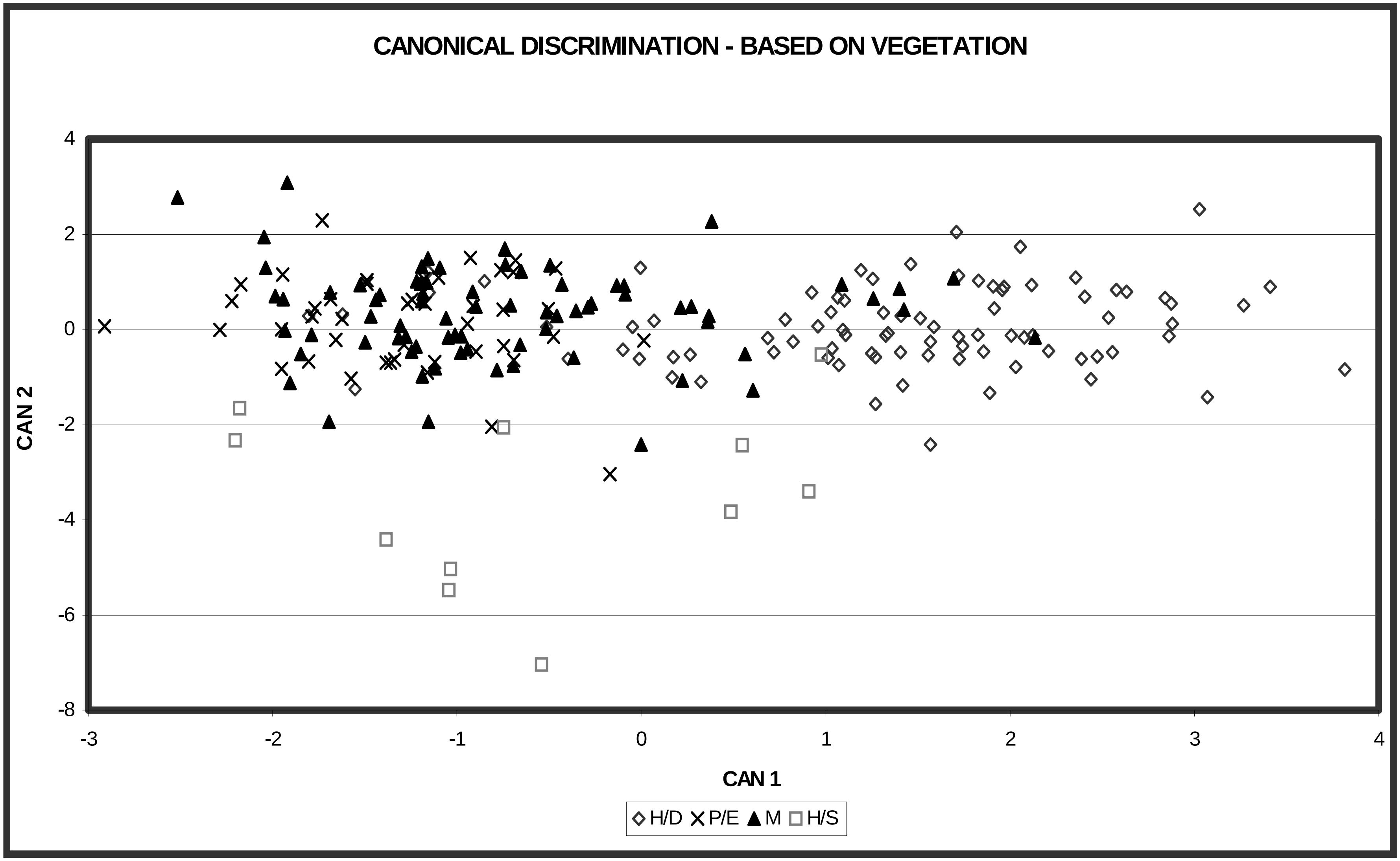


Figure 6. Plot of first and second canonical variables based on vegetation using Phase 1 data, 20 biogeochemical variables



.Figure 7. The probability of being undisturbed at each of the phase 1 sampling sites based on the logistic regression model, Fort Bening Military Reservation, Georgia.

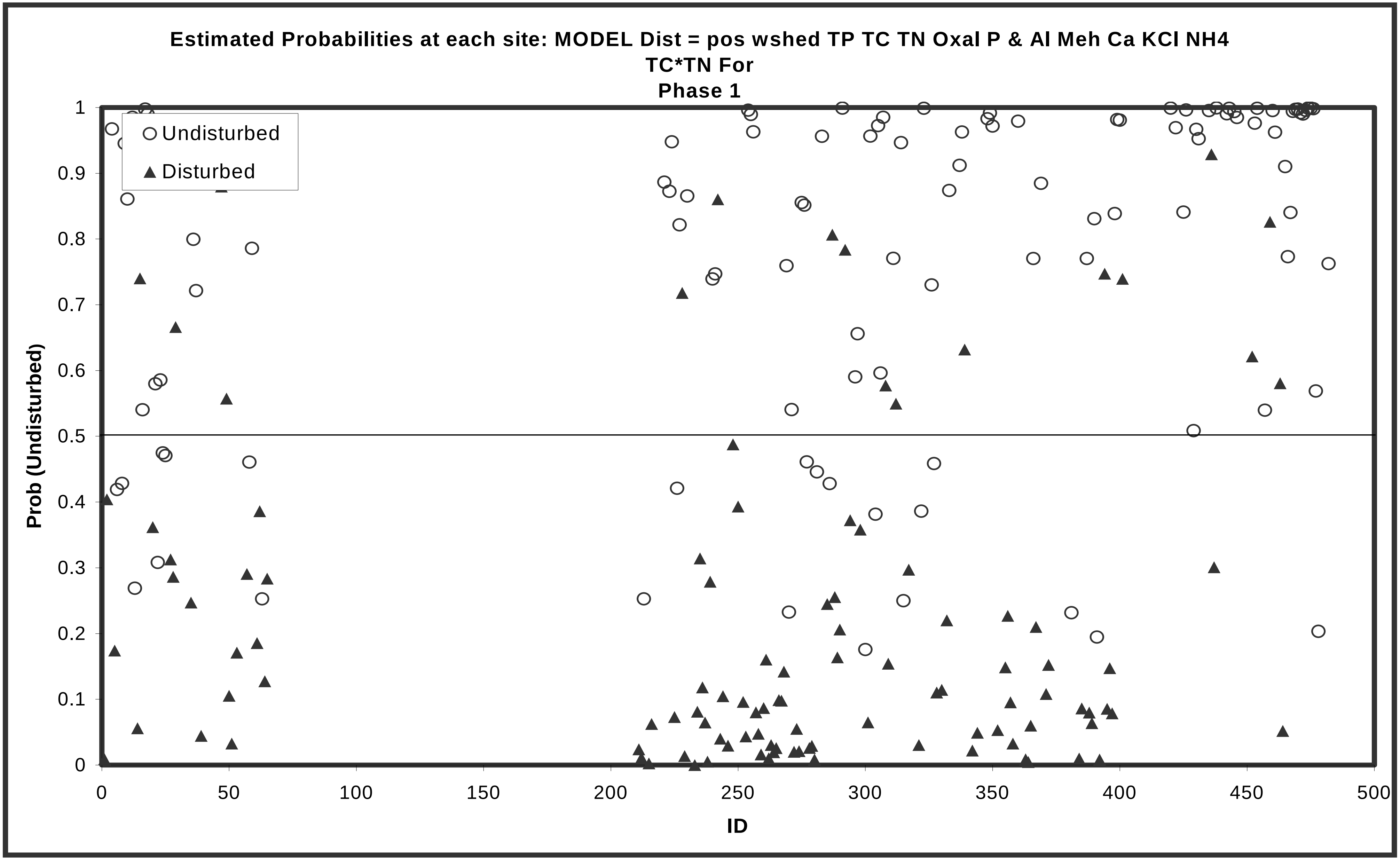
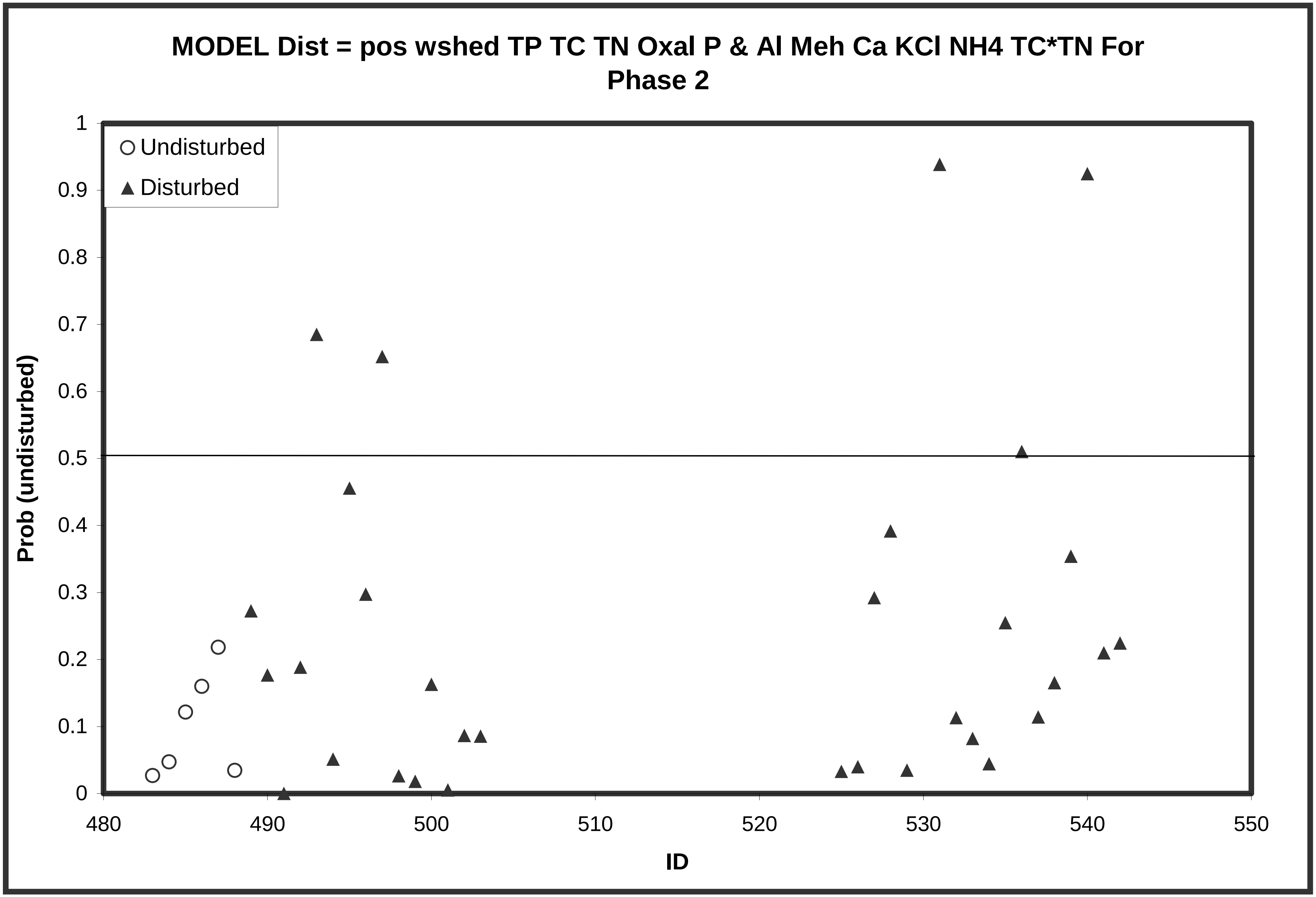


Figure 8. The probability of being undisturbed at each phase2 sampling site based on the Logistic regression model, Fort Benning Military Reservation, Georgia.



3.5.2:

Quantitative analysis of soil nutrient concentrations with near infrared spectroscopy and partial least squares regression. Dabral, S., W. D. Graham, and J.P. Prenger.

ABSTRACT

This paper explores the feasibility of using lab spectral reflectance measurements in the NIR region to predict pH, Ash, Total Phosphorus (TP), Total Carbon (TC), Total Nitrogen (TN), Water-extractable phosphorus (WEP), Water extractable carbon (WEC), Oxalate extractable phosphorus, iron, and aluminum (Oxal P, Oxal Fe, Oxal Al), Microbial biomass Carbon (MBC), KCl extractable ammonium (KCl NH_4), and Mehlich extractable phosphorus, iron, aluminum, calcium, magnesium, and potassium (Meh P, Meh Fe, Meh Al, Meh Ca, Meh Mg, Meh K) using partial least squares regression (PLS). Over 397, topsoil (0-20 cm) samples, taken from a range of disturbance and landscape positions representing a wide range of biogeochemical conditions at Fort Benning, Georgia were used to develop the PLS model (Phase 1). An independent dataset of 155 (Phase 2) soil samples was used to test the predictive ability of NIRS using the model. Out of the 20 soil biogeochemical variables that were measured in this study 10 of the variables showed a forecasting efficiency (EF) of 60% and higher in Phase 1. The most accurately predicted biogeochemical variables for the calibration dataset in the order of forecasting efficiency were: TC (0.796), TN (0.746), MehMg (0.694), TP (0.635), MehK (0.624), MehFe (0.619) and MehCa (0.605). The biogeochemical variables that were most accurately predicted for the independent Phase 2 dataset were: TN (0.888), TC (0.868), OxalFe (0.759), TP (0.769), MBC (0.705) and OxalP (0.61). The results presented in this paper indicate that near-infrared spectroscopy coupled with partial least squares can be very useful to rapidly estimate several soil nutrient properties. Thus near-infrared spectroscopy has the potential to be used as an efficient and cost-effective way to measure soil nutrient status, which can be an indicator of ecosystem integrity.

INTRODUCTION

For an ecosystem to be sustainable it is important that the proper soil quality is maintained and if possible enhanced. Monitoring of function and long-term sustainability of forest ecosystems relies on the use of indicators (Schoenholtz et al., 2000). In the case of soil quality, an indicator is a measurable surrogate of a soil attribute that determines how well a soil functions (Burger and Kelting, 1999). Indices of soil quality which incorporate soil chemical, physical and biological properties are most likely to be adopted if they are sensitive to management-induced changes, easily measured, inexpensive, adaptable for specific ecosystem, relevant across sites and over time and closely linked to measurement of desired values such as productivity or biodiversity (Schoenholtz *et al*. 2000). Assessment of indicators of soil quality is usually done in the lab by measuring soil physical, chemical and biological properties and in general, several soil properties are required to fully characterize a soil quality. With a growing need for the accurate representation of the continuous spatial and temporal variation in soil properties and where more spatially dense analyses are required, in-lab soil extraction procedures alone may not meet these needs (Janik *et al*. 1998). Thus, there is a need for new methods that could complement and possibly substitute for laboratory assessment of soil properties in the future. The use of near-infrared spectroscopy (NIRS) for soil analysis offers one such option.

In recent years, interest in the use of near-infrared spectroscopy (NIRS) (350 to 2500 nm) for prediction of soil properties has greatly increased. NIRS is known for its ability to rapidly, conveniently and accurately analyze many constituents at the same time (Stark *et al.* 1986). Infrared spectroscopy coupled with multivariate analytical methods is emerging as a viable tool to predict many soil properties at a time, for example, clay and/or organic matter content (Sudduth and Hummel 1991, Morra *et al.* 1991, Henderson *et al*. 1992; Ben-Dor *et al* 1995, Wander *et al*. 1996; Chang *et al*. 2001, Dunn *et al*. 2002, Ludwig *et al*. 2002, Groenigen et al. 2003), sensitive soil biomass parameters like microbial carbon and nitrogen (Palmborg *et al.* 1993, 1996, Ludwig *et al.* 2002), soil mineral nitrogen content (Upadhyaya *et al*. 1994, Ehsani *et al.* 1999), total amounts of soil calcium, magnesium, iron, manganese and potassium (Udelhoven *et al.* 2003), and heavy metals in contaminated soils like arsenic, iron, mercury, lead, cadmium, copper, and zinc (Malley and Williams 1997, Kemper *et al.* 2002).

The emergence of robust multivariate analyses techniques like Partial Least Squares (PLS) and principal component regression (PCR) in the last few years has made efficient predictive relationships between lab reflectance measurements and soil properties possible. Near infrared spectral scanning provides a considerable amount of information in contiguous wavelengths, which can be strongly collinear. That is, there may exist a strong linear relationship or co-dependence between the reflectance observed at different wavelengths. When two or more of the explanatory variables (reflectance) co-vary it is very difficult to determine their separate influences on the variable that one is trying to explain (soil biogeochemistry) and therefore, to define a unique regression coefficient for each of these variables. At the same time, it is highly unlikely that all of the data are of explanatory importance; typically, some of the data may be redundant and can be excluded from the analysis.

When the problem of co-linearity in the data exists, an ordinary regression approach is no longer feasible. PLS regression extracts successive linear combinations of the predictors, called latent vectors, that perform simultaneous decomposition of both predictors and responses with the constraint that these components explain as much as possible of the co-variance between the predictors and the responses. A detailed overview of PLS algorithms can be found in Wold 1966, Haaland *et al.* 1988 a and b, deJong 1992 and1993, Garthwaite 1994, and Helland 2001. Several studies have used PLS for soil spectral analysis in the laboratory (Malley and Williams 1997, Ehsani *et al.* 1999, Kemper *et al.* 2002, Ludwig *et al.* 2002, Shepherd and Walsh 2002, Warr 2002, Dunn *et al.* 2002, Groenigen *et al.* 2003, Kooistra *et al.* 2003, Udelhoven *et al.* 2003).

This paper explores the feasibility of using lab spectral reflectance measurements in the NIR region to predict the soil pH, Ash, Total Phosphorus (TP), Total Carbon (TC), Total Nitrogen (TN), Water-extractable phosphorus (WEP), Water extractable carbon (WEC), Oxalate extractable phosphorus, iron, and aluminum (Oxal P, Oxal Fe, Oxal Al), Microbial biomass carbon (MBC), KCl extractable ammonium (KCl NH_4), and Mehlich extractable phosphorus, iron, aluminum, calcium, magnesium, and potassium (Meh P, Meh Fe, Meh Al, Meh Ca, Meh Mg, Meh K) using partial least squares regression. Over 550 topsoil (0-20 cm) soil samples taken from a range of disturbance and landscape positions representing a wide range of biogeochemical conditions at Fort Benning Military installation, Georgia were used in this analysis.

MATERIALS AND METHODS

Study Area

The study area is the Fort Benning Military Reservation, located near Columbus in west-central Georgia (see Figure 1). Fort Benning occupies approximately 73,533 hectares of land area and provides facilities including ranges and maneuver training areas to train soldiers in the science of combined infantry principles, weapon systems, and military tactics. Some 60% of the total land area at Fort Benning is designated as maneuver areas (Department of Army, 1997). Topographic slopes within the study area range from 2 to 15%. A combination of clay beds and weathered coastal plain material and alluvial posits from the Piedmont characterize the soils of Fort Benning. The majority of soils in the Piedmont are classified as Ultisols with Alfisols and Entisols comprising most of the remaining soil types. Ultisols are characterized by weathered minerals and/or subsurface clay horizon. Kaolinite is the most common clay mineral but vermiculite and illite also occur. Sheet and gully erosion is common in this type of landscape. Wetlands and hydric soils are generally restricted to bottomlands along streams and creeks. The type of military training required at Fort Benning affects the nature and extent of the ecological disturbance. Frequent heavy vehicle movement in confined areas can significantly damage ground cover and cause significant soil disturbance and movement, resulting in soil erosion, loss of soil organic matter and stream sedimentation. There is a need for development of "easy to use and easy to measure" indicators that can indicate the status of the ecosystem and help in proper management of the land so that both the military mission and the ecological integrity can be maintained at Fort Benning.

Soil Sampling Scheme

More than 550 soil samples were collected in two phases from the Fort Benning study site. The sampling scheme was designed to capture the full range of spatial variability of soil properties occurring at Fort Benning. Sampling locations were selected to cover a wide range of military and non-military land uses and anthropogenic disturbance regimes. The 397 Phase 1 sampling sites were located along transects which were transverse to the orientation of the main stream channel in fifth and sixth order watersheds, providing approximately uniform coverage of the watersheds. One third of the above mentioned sites were located in bottomlands (wetlands), and two-thirds were located in uplands (side slopes, hilltops and ridges). The 155 Phase 2 sampling sites were designed to characterize well-defined ecological and anthropogenic impact gradients on a smaller scale, and at a greater spatial resolution.

Soil Collection

Soil samples were collected from the upper 20 cm of the surface horizon, using a soil push probe with an inside diameter of 1 inch. Each sample location consisted of a 1-m^2 square plot, within which 5 subsamples were taken at the corners and center of the plot. The individual samples were then composited for analysis as a single sample. Phase 1 sampling was conducted from January to August 2000, while Phase 2 sampling was from December 2000 to June 2001.

Soil Chemical Analyses

Chemical analyses on Phase 1 and Phase 2 soil samples were performed by the University of Florida Wetlands Biogeochemistry Laboratory. Soil pH was measured in 1:1 soil: water slurry in the lab. Ash content was determined from residue after ashing at 550 ^oC (Anderson, 1976). Total phosphorus (TP) analysis was done by dry ashing followed by dissolution in 6M HCl (Anderson, 1976). Total carbon (TC) and total nitrogen (TN) determination was by dry combustion (Nelson and Sommers, 1996). Water-extractable phosphorus (WEP) and water extractable carbon (WEC) were determined by the method of Kuo (1996). Oxalate extractable phosphorus, iron, and aluminum (Oxal P, Oxal Fe, Oxal Al) analyses were performed by the method of Bertsch and Bloom (1996). Determination of microbial biomass carbon (MBC) was by the fumigation-extraction procedure (Horwath and Paul, 1994) as modified by DeBusk and Reddy (1998). KCl extractable ammonium (KCl NH_4) was determined by the method of Mulvaney (1996). Mehlich extractable phosphorus, iron, aluminum, calcium, magnesium, and potassium (Meh P, Meh Fe, Meh Al, Meh Ca, Meh Mg, Meh K) extract were done by the method of Amacher (1996). Phase 1 soil samples were also analyzed for microbial biomass phosphorus (MBP) by the method of Brookes *et al.* (1982) and microbial biomass nitrogen (MBN) by the method of Sparling *et al.* (1990). Nitrogen, phosphorus and carbon levels in extractions and were determined by EPA methods 351.2, 365.1 and 415.1 (1993). Mehlich extractable metals were determined by ICP- EPA method 200.7.

Reflectance Measurements

The reflectance measurements were made using a FieldSpec Pro FR spectroradiometer (Analytical Spectral Devices Inc., Boulder, Colorado) spanning the range from 350 nm to 2500 nm at a sampling interval of 1 nm. A high intensity contact probe with a spot size of 10 mm and a 4.5W internal halogen lamp was used to sense the soil samples. The soil samples were oven-dried at 105 ^{o} C for 24 hours to standardize the effects of moisture content. To eliminate the effects of scattering of light by soil aggregates of variable sizes, soils were ground with a mortar and passed through a 2mm sieve. Oven-dried and ground soil samples were placed in 12-mm deep, 55-mm diameter polystyrene petridishes to a thickness of 1.5 cm. Reflectance measurements taken at four random points inside the petridish were averaged as the reflectance signature of the soil sample. A white reference spectrum was generated between each reading using a calibrated spectralon surface (Lab Sphere, Sutton, NH) with a diffuse reflectance of 99% from 400 nm to 95% at 2500 nm. Soil reflectance was calculated relative to the spectralon (close to 100% reflecting surface). Reflectance measurements were taken on 397 Phase 1soil samples and 155 Phase 2 soil samples. Raw reflectance measurements without any data preprocessing were used for statistical analyses.

Statistical Analyses

The data were analyzed using Partial Least Squares (PLS) regression (SAS version 8.01, Statistical Analysis Systems, Cary, NC). Three hundred ninety-seven Phase 1 reflectance measurements (predictors) and 20 biogeochemical variables (responses) were used to calibrate and validate the model and 155 Phase 2 reflectance measurements were used to independently verify the models' accuracy. In order to select the optimum number of PLS factors, a "leave-one-out" cross-validation technique was used (SAS, 1999). In this technique, each sample is left out sequentially during calibration and then predicted using the calibrated model. This requires a recomputation of the PLS model for every input of observation (SAS, 1999). Usually the number of factors are chosen to minimize the predicted sum of squares (PRESS). However often models with fewer factors have PRESS statistics that are only marginally larger than the minimum value. In this study, a test proposed by van der Voet (1994) was used to compare the predicted residuals from different models. This test ensures that the model chosen has the fewest factors with residuals that are insignificantly larger than the residuals of the model with minimum PRESS (SAS, 1999). No spectral or concentration outliers were removed from the calibration dataset.

The success of the models was evaluated using the following statistical measures: forecasting efficiency (EF), mean error (ME) and standard error of performance (SEP), RPD, RER.

Standard error of performance is the standard deviation of error between the reference and the NIRS estimated values for samples in the validation set (Williams, 1987). Forecasting efficiency (EF) is a relative measure of error and is a statistic to test the goodness of fit of the model Y = X with the restrictions a=0 and b =1 in the model Y = a + bX. EF is equal to one minus the ratio of SEP and SST, where SST is the sum of squares total (squared error for each observation summed across all observations). EF is less than one for any realistic simulation and less than zero if the model predicted values are worse than simply using the measured mean. (Yang et al., 2000). The best calibration is the one with the highest EF and the lowest SEP.

For comparison purposes between calibration of constituents with different ranges of concentrations, RPD and RER are used. RPD is the ratio of the standard deviation of values in the validation set to the SEP and RER is the ratio of the range in the validation set to the SEP (Williams, 1987). In the published literature researchers have used the RPD values to form distinct categories and indicate decreasing reliability of prediction using NIRS (Chang *et al*., 2001, Dunn *et al.*, 2002). In general, RPD>3 is considered acceptable and RPD>5 excellent (Malley *et al*, 1999). RER should be above 10 and is often above 20 (Malley *et al*., 1999). Chang *et al.* (2001) used RPD values to evaluate the ability of NIRS to predict soil properties and classify them into three categories: Category A (RPD >2.0), Category B (RPD between 1.4 and 2.0) and Category C (RPD below 1.4). Chang et al. (2001) argued that the predictions of soil properties in category B could be improved by using different calibration strategies while properties in category C may not be reliably predicted using NIRS. These results were reinforced by Dunn *et al*, 2002 (RPD >2.0 excellent, RPD between 1.6 and 2.0 acceptable and RPD <1.6 poor).

Log-transformed values of the biogeochemical measurements were used in the analyses to re-scale all the variables so that they contained similar ranges of variability, to reduce the magnitude of variation of the variables, and to improve the normality of the distributions. Ash and OxalP data were highly skewed even after the log transformations.

RESULTS

Summary statistics for the soil biogeochemical constituents measured on the Phase 1 soil data are shown in the Table 1, which includes the mean, range, and standard deviation of both the original and log transformed data. The correlation structure of the soil variables was also examined and the correlation matrix of the log variables is presented in Table 2.with the correlations greater than 0.60 highlighted. Total C, N, and P showed a good correlation (sample correlation coefficient>0.60) with other soil variables (OxalFe, OxalAl, MehAl and MehFe) as well as a strong inter-correlation among themselves. Mehlich Ca, Mg, and K showed a strong inter-correlation among themselves and with TP. Microbial biomass C and N were also correlated with TC and TN. A nine factor PLS model was selected for calibration and validation. The amount of prediction and response variation explained by each PLS latent factor is shown in Table 3. The first 9 latent factors explained about 99% of the variance in the reflectance dataset and 43.3% of the variance in the biogeochemical dataset. An absolute minimum PRESS of 0.7814 was achieved with nine extracted factors and with a *p*-value of 0.04.

The first five factor weights for the reflectance dataset are shown in figure 2. Plots of the weights indicate the direction toward which each PLS factor projects. These weights can be considered as indicators of the "correlations" between the property of interest and the infrared frequencies (Janik *et al.* 1998). Those wavelengths with small weights are less important than those with large weights in absolute value. All the factors in Figure 2 show non-zero weightings. The first factor remains mostly positive throughout the entire spectrum range, with high positive weights between 400 and 1100 nm and in the spectrum region greater than 1900 nm. Factor two follows the general shape of the soil spectra between 550 nm to 1900 nm and possibly accounting for variation in the overall average reflectance. Factor 3 has positive weights in the region between 500 nm and 1000 nm indicating variations associated with both organic matter and iron content in the soil. A very strong maxima at 606 nm also occurs in factor 3 that can be attributed to the organic matter in the soil. Higher order factors also showed that the PLS model gave somewhat larger importance to the wavelengths around 1400 nm, 1900 nm and 2100 nm, which are related to the presence of hydroxyl bonds either in soil minerals or in water molecule in the pores.

Table 4 shows the ME, EF, SEP, RPD and RER for the final calibrated model using Phase 1 soil data and reflectance measurements. Out of 20 soil biogeochemical variables that were used in this study seven variables showed a forecasting efficiency of 60% and higher for the calibration set. The most accurately predicted biogeochemical variables for the Phase 1 parameters in the order of forecasting efficiency were: TC (0.796), TN (0.746), MehMg (0.694), TP (0.635), MehK (0.624), MehFe (0.619) and MehCa (0.605). The Phase2 reflectance dataset were used to independently predict soil biogeochemistry using the Phase 1 PLS model. Table 5 summarizes the prediction results. The most accurately predicted Phase 2 biogeochemical variables in order of their forecasting efficiency were: TN (0.888), TC (0.868), OxalFe (0.759), TP (0.769), MBC (0.705) and OxalP (0.61). Scatter plots of actual and PLS predicted soil properties for Phase 1 and 2 dataset are shown in Figure 3 and 4 respectively, along with a 1:1 line.

DISCUSSION

Using the criteria established by Chang *et al.* (2001) with the calibration dataset in our study, group A (RPD>2.0, EF >0.75)) included soil TN and TC; group B (1.4<RPD>2.0, 0.5< EF >0.75) included TP, OxalFe, MehAl, MehFe, MehMg, MehK and MehCa; group C (RPD

<1.4, EF<0.5) included microbial C, P and N, WEC, WEP, KCl NH_4, OxalP and Al, MehP, pH and ash. For the independent dataset, group A included TC, TN, TP, and Oxal Fe; group B included OxalP and MBC while the rest of the soil properties were in group C. These results indicate that TC, TN and TP can be readily and accurately predicted using NIRS for both the calibration and independent data sets. Similar results have been reported in other studies. Chang *et al.* (2001) reported a high prediction for TC (r^2=0.87, RPD = 2.79) and TN (r^2=0.85, RPD = 2.52).

Unlike the success in prediction of soil pH in previous major studies (Chang *et al.* 2001, Dunn *et al.* 2002), the prediction of soil pH was poor in our studies for both the calibration dataset (RPD = 1.06, EF=0.11) and the independent dataset (RPD =0.99, EF=0.0). Chang *et al.* (2001) reported an r^2 of 0.55 and RPD of 1.43, while Dunn *et al.* (2002) reported a predicted r^2 of 0.80 and a RPD of 2.30 for soil pH. Since pH is a secondary soil property (i.e. it does not respond directly to the near-infrared light), its prediction depends on how well it is correlated to primary soil properties like total carbon and nitrogen (Chang *et al.* 2001). One reason for this less than acceptable prediction may be the poor correlation of pH with almost all the other soil constituents and especially TC and TN (Table 3). In contrast to most studies correlating NIRS to soil properties which have focused on relatively homogeneous soils, we have chosen to address soils from a wide range of landscapes and impacts. Properties such as pH would be expected to vary considerably between bottomland and upland soils, thus reducing the overall correlation. Ash content was another soil property that was poorly predicted in this study. We suspect that the narrow range of variability for log ash content, except for a few extreme outliers is responsible for this less than acceptable prediction of ash content. It should be noted that in the Phase 1 data set log-Ash showed little correlation with other soil properties (see Table 2).

Compared to the study done by Ludwig *et al*. 2002, that showed a high correlation coefficient (r) for soil microbial carbon (r =0.83) and microbial nitrogen (r = 0.76), the success in prediction of microbial group properties varied in our study. For the calibration dataset none of the microbial properties was accurately predicted while for the independent dataset microbial carbon showed good prediction (RPD =1.842). The poor prediction of MBC and MBN for the calibration dataset in spite of showing good correlation with TC and TN (0.63 and 0.63 respectively), and the better prediction of MBC for the independent dataset is difficult to explained.

Mehlich extractable cations showed good prediction for the calibration dataset but not for the independent dataset. Predictions for the calibration dataset were MehCa (RPD = 1.59, EF = 0.60), MehMg (RPD = 1.80, EF =0.69), MehK (RPD = 1.63, EF=0.62) and MehFe (RPD = 1.62, EF = 0.61). Chang *et al.* (2001) reported a RPD value of 2.19 for MehCa, 1.78 for MehMg, 1.64 for MehK and 1.66 for MehFe. The calibration dataset showed a very high inter-correlation between MehCa, MehMg and MehK (Table 3). Covariance of these soil properties with other properties exhibiting a primary response in the near-infrared region may be responsible for the good predictions for the calibration dataset.

The success in prediction of Oxalate group soil properties (OxalAl, OxalFe, OxalP) using NIRS varied. Oxalalate extractable Fe showed a RPD value of 1.462 for the calibration dataset and a RPD value of 2.03 for the independent dataset. Also, OxalP showed a RPD value of 1.60 for the independent dataset. Oxalalate extractable Al was more poorly predicted than would be expected from its covariance with primary soil properties.

The prediction of the secondary soil properties is dependent on their association with primary soil properties (TC, TN, TP). The prediction of secondary soil properties for an independent dataset might not be accurate if there is a serious departure from the correlation structure of the calibration dataset. The correlation structure of the calibration (Phase 1)and independent (Phase 2) datasets used in our study are shown in Tables 2 and 6, respectively. The Phase 2dataset showed much stronger correlation among the variables, perhaps because the Phase 2 data was sampled along a few well-defined transects at a much higher spatial resolution and at a smaller scale the Phase 1 data that was sampled to capture the full range of variability of soil properties across the base. Thus it is likely that the soil types varied less within the Phase 2 transects than across the full army base, resulting in stronger correlation among biegeochemical variables. As can be seen from Table 6, in Phase 2 many secondary soil properties showed a higher correlation with TC, TN and TP than they did in Phase 1. However, since the model was built using the Phase 1 calibration dataset there is a very little chance that these secondary soil properties are going to be predicted accurately for the Phase 2 independent dataset.

CONCLUSIONS

Reflectance measurements and 20 soil biogeochemical variables measured on over 550 soil samples were used to develop a robust PLS model for independently predicting TC, TN, and TP of new observations based on the reflectance measurements. Measuring these parameters in the lab can be expensive and time consuming. The results presented in this paper indicate that near-infrared spectroscopy coupled with partial least squares can be a useful and inexpensive alternative. This study reinforces the versatility of NIRS-PLS in successfully predicting soil nutrient content for a diverse range of disturbance regimes and landscape position across a wide range of soil types. Thus near-infrared spectroscopy has the potential to be used as an efficient and cost-effective way to measure soil nutrient status, which can be an indicator of ecosystem integrity.

TC, TN, TP were predicted accurately in both the Phase 1 (calibration) and Phase 2 (independent) datasets even though these data were collected under different sampling strategies. This indicates that using near-infrared spectroscopy provides a robust method to predict these soil properties in a wide variety of conditions. However, since none of the other constituents were successfully predicted in both the calibration and independent datasets further investigations should be conducted before near- infrared spectroscopy is used to predict these constituents at Ft. Benning. Our goal was to derive indicators of disturbance that were generally applicable over a diverse landscape; however, developing calibration datasets based on data stratified by landscape position or watershed of origin may provide better predictive capabilities for the secondary soil properties.

REFERENCES

Amacher, M.C., 1996. Double Acid (Mehlich I) Method. p. 757. *In Methods of Soil Analysis Part 3 Chemical Methods.* J.M. Bigham (ed.), SSSA-ASA. Madison, Wisconsin.

Analytical Spectral Devices, Inc., Boulder, Colorado. (www.asdi.com)

Anderson, J. M., 1976. An ignition method for determination of total phosphorus in lake sediments. *Water Research* 10:329-331.

Ben-Dor, E., and Banin, A. 1995. Near-infrared analysis as a rapid method to simultaneously evaluate several soil properties. *Soil Science Society of America Journal* 59, 364-372.

Bertsch, P.M. and P.R. Bloom, 1996 Ammonium Oxalate Extractable Iron and Aluminum. p. 530. *In Methods of Soil Analysis Part 3 Chemical Methods*. J.M. Bigham (ed), SSSA-ASA. Madison, Wisconsin.

Brookes, P.C., D.S. Powlson, and D.S. Jenkinson. 1982. Measurement of microbial biomass phosphorus in soil. *Soil Bio. Biochem.* 14:319-329.

Chang, W. C., Laird, D. A., Mausbach, M. J. and Hurburgh, C. R. 2001. Near-Infrared Reflectance Spectroscopy – Principal Component Regression Analyses of Soil Properties. *Soil Science Society of America Journal* 65: 480-490.

DeBusk, W.F., and K.R. Reddy. 1998. Turnover of detrital organic carbon in nutrient-impacted Everglades marsh. *Soil Sci. Soc. Am. J.* 62:1460-1468.

de Jong, S. and Kiers, H. 1992, "Principal Covariates Regression," *Chemometrics and Intelligent Laboratory Systems*, 14, 155 -164.

de Jong, S. 1993, "SIMPLS: An Alternative Approach to Partial Least Squares Regression," *Chemometrics and Intelligent Laboratory Systems*, 18, 251 -263.

Dunn, B.W., Beecher, H.G., Batten, G.D., and Ciavarella, S. 2002. The potential of near-infrared reflectance spectroscopy for soil analysis- a case study from the Riverine Plain of south-eastern Australia. *Australian Journal of Experimental Agriculture*. 42, 607-614.

Ehsani, M.R., Upadhyaya, S.K., Slaughter, D., Shafii,S. and Pelletier, M. 1999. A NIR technique for rapid determination of soil mineral nitrogen. *Precision Agriculture.* 1, 217-234.

Garthwaite, P. H. 1994. An interpretation of Partial Least Squares. *Journal of the American Statistical Association.* 89 (425), 122-127.

Groenigen, J. W. V., Mutters, C. S., Horwath, W. R., and Kessel, C. V. 2003. NIR and DRIFT-MIR spectrometry of soils for predicting soil and crop parameters in a flooded field. *Plant and Soil* 250: 155-165.

Haaland, D. M., and Thomas, E. V. 1988a. Partial Least Squares Methods for Spectral Analyses. 1. Relation to other Quantitative Calibration Methods and the Extraction of Qualitative Information. *Anal. Chem.* 60, 1193-1202.

Haaland, D. M., and Thomas, E. V. 1988b. Partial Least Squares Methods for Spectral Analyses. 2. Application to Simulated and Glass Spectral Data. *Anal. Chem.* 60, 1202-1208.

Helland, I. S. 2001. Some theoretical aspects of partial least squares regression. *Chemometrics and Intelligent Laboratory Systems* 58, 97-107.

Henderson, T.L., Baumgardner, M.F., Franzmeier, D.P., Stott, D.E., and Coster, D.C. 1992. High dimensional reflectance analysis of soil organic matter. *Journal of the Soil Science Society of America.* 56, 865-872.

Horwath, W. R., and E. A. Paul. 1994. Microbial biomass. p. 753-773. In R. W. Weaver et al. (ed.) *Methods of soil analysis, Part 2. Microbiological and biochemical properties.* SSSA-ASA. Madison, Wisconsin.

Janik, L. J., Merry, R.H., and Skjemstad, J.O. 1998. Can mid infrared diffuse reflectance analysis replace soil extractions? *Australian Journal of Experimental Agriculture*. 38, 681-696.

Kemper, T., and Sommer, S. 2002. Estimate of Heavy Metal Contamination in Soils after a Mining Accident Using Reflectance Spectroscopy. *Environ. Sci. Technol.* 36, 2742-2747.

Kooistra, L., Wanders, J., Epema, G.F., Leuven, R.S.E.W., Wehrens, R., and Buydens, L.M.C. 2003. The potential of field spectroscopy for the assessment of sediment properties in river floodplains. *Analytica Chimica Acta* 484, 189-200.

Kuo, S. 1996. Phosphorus. In Methods of Soil Analysis. Part 3, Chemical Methods "Availability Indicators" p. 891. J.M. Bigham (ed.), *Soil Science Society of America,* Inc. ASA, Madison, Wisconsin.

Ludwig, B., Khanna, P.K., Bauhus, J., and Hopmans, P. 2002. Near infrared spectroscopy of forest soils to determine chemical and biological properties related to soil sustainability. *Forest Ecology and Management* 171, 121-132.

Malley, D.F., and Williams, P.C. 1997. Use of Near-Infrared Reflectance Spectroscopy in Prediction of Heavy Metals in Freshwater Sediment by Their Association with Organic Matter. *Environ. Sci. Technol.* 31, 3461-3467.

Malley, D.F., Yesmin, L., Wray, D., Edwards, S. 1999. Applications of near-infrared spectroscopy in analysis of soil mineral nutrients. *Communications in Soil Science and Plant Analysis*. 30, 999-1012.

Morra, M.J., Hall, M.H., and Freeborn, L.L. 1991. Carbon and nitrogen analysis of soil fractions using near-infrared reflectance spectroscopy. *Soil Science Society of America Journal* 55, 288-291.

Mulvaney, R.L. 1996. Nitrogen-Inorganic Forms. p. 1123. *In Methods of Soil Analysis, Part 3. Chemical Methods*, J. M. Bigham, (ed.) SSSA-ASA. Madison, Wisconsin, USA.

Nelson, D.W. and L.E. Sommers. 1996. Total Carbon and Total Nitrogen. p. 973. *In Methods of Soil Analysis Part 3 Chemical Methods*. J.M. Bigham (ed) SSSA ASA. Madison, Wisconsin.

Palmborg, C., and Nordgren, A. 1993. Modeling microbial activity and biomass in forest soil with substrate quality measured using near infrared spectroscopy and multivariate statistics. *Soil Biology and Biochemistry* 25, 1713-1718.

Palmborg, C., and Nordgren, A. 1996. Partitioning the variation of microbial measurements in forest soils into heavy metal and substrate quality dependent parts by use of heavy metal and substrate quality dependent parts by use of near infrared spectroscopy and multivariate statistics. *Soil Biology and Biochemistry* 28, 711-720.

Schoenholtz, S.H., Miegroet, H.V., and Burger, J.A. 2000. A review of chemical and physical properties as indicators of forest soil quality: challenges and opportunities. *Forest Ecology and Management* 138, 335-356.

Sparling, G.P., C.W. Feltham, J. Reynolds, A.W. West, and P. Singleton. 1990. Estimation of soil microbial C by a fumigation-extraction method: use on soils of high organic matter content and a reassessment of the K_{EC} factor. *Soil Bio.Biochem.* 22:301-307.

Shepherd, K.D. and M.G. Walsh. 2002. Development of reflectance spectral libraries for characterization of soil properties. *Soil Science Society of America Journal*. 66:988-998.

Stark, E., Luchter, K., Margoshes, M., 1986. Near-infrared analysis (NIRA): a technology for quantitative and qualitative analysis. *Appl. Spec. Rev.* 22, 335-399.

Sudduth, K.A., and Hummel, J.W. 1991. Evaluation of reflectance methods for soil organic matter sensing. *Transactions of the ASAE* 34, 1900-1909.

Upadhyaya, S.K., Shafii, S., and Slaughter, D. 1994. Sensing soil mineral nitrogen for site-specific crop management (SSCM). *ASAE paper* No. 94-1055 (ASAE St. Joseph, MI).
U. S. Environmental Protection Agency (USEPA). 1993. *Methods for the determination of inorganic substances in environmental samples*. Environ. Monit. Support Lab., Cincinnati, OH.
van der Voet, H. 1994, "Comparing the Predictive Accuracy of Models Using a Simple Randomization Test," *Chemometrics and Intelligent Laboratory Systems*, 25, 313 -323.
Wander, M. M., and Traina, S. J. 1996. Organic matter fractions from organically and conventionally managed soils 2. Characterization of composition. *Soil Science Society of America Journal* 60, 1087-1094.
Warr, B. 2002. “Analysis of remote and proximally sensed data for the prediction of soil properties”. *Thesis*. The University of Reading, United Kingdom.
Williams, P.C. 1987. Variables affecting near-infrared reflectance spectroscopy analysis. *In ‘Near-infrared technology in the agricultural and food industries’* (Eds P. Williams, K. Norris) pp 143-167. (American Association of Cereal Chemists Inc.: St. Paul, MN, USA.
Wold, H. 1966, "Estimation of Principal Components and Related Models by Iterative Least Squares," in *Multivariate Analysis*, ed. P. R. Krishnaiah, New York: Academic Press, 391 -420.
Yang, J., Greenwood, D. J., Rowell, D. L., Wadsworth, G. A., and Burns, I. G. 2000. Statistical methods for evaluating a crop nitrogen simulation model, N_ABLE. *Agricultural Systems*. 64, 37-53.

Table 1. Descriptive statistics of the 20 Biogeochemical variables used in this study-Phase 1

	N	Mean8	Meanδ	Max8	Maxδ	Min8	Minδ
pH	397	5.14	-	7.84	-	3.83	-
Ash [%]	397	1.96	91.20	1.45	97.72	1.99	28.18
TC [g/kg]	396	1.09	12.32	-0.08	245.47	2.39	0.831
TP [mg/kg]	397	1.94	87.09	1.73	1.129	2.82	53.70
TN [g/kg]	397	-0.27	0.537	-1.52	13.458	1.129	0.030

Oxal Al [g/kg]	328	3.19	1548.8	2.67	10471.2	4.02	467.73
Oxal Fe [g/kg]	396	3.23	1698.2	2.15	70794.5	4.85	141.25
Oxal P [mg/kg]	348	1.42	26.30	0.85	478.63	2.68	7.079
Meh Al [mg/kg]	328	2.42	263.0	1.58	2570.3	3.41	38.01
Meh Fe [mg/kg]	326	1.68	47.86	0.717	2511.88	3.40	5.211
Meh P [mg/kg]	397	-0.01	0.977	-0.95	33.88	1.53	0.112
MehMg[mg/kg]	328	1.37	23.44	-0.10	691.85	2.84	0.794
Meh P [mg/kg]	328	1.35	22.38	-0.95	33.88	1.53	0.112
Meh Ca [mg/kg]	328	1.89	77.62	0.37	2089.29	3.32	2.344
MBC [mg/kg]	395	2.36	229.08	1.12	3.64	4365.1	13.18
MBN [mg/kg]	301	1.47	29.512	-0.06	2.47	295.12	0.870
MBP [mg/kg]	297	0.30	1.99	-1.71	1.85	70.79	0.019
WEC [mg/kg]	395	1.75	56.23	0.72	2.96	912.01	5.248
WEP [mg/kg]	397	-0.94	0.114	-2.01	0.66	2.01	0.009
KCl ext. NH_4 [mg/kg]	397	0.60	3.98	0.986	2.01	102.3	0.986

8the values are in log space δ the values are in real space.

Table 2. Correlation matrix of the 20 Phase 1 biogeochemical variables, Fort Benning, Georgia.

	pH	Ash	TP	WEP	OxalP	MehP	MBP	TC	TN	WEC	MBC	KCl NH_4	MBN	Meh Fe	Meh Al	Meh Ca	Meh Mg	Meh K	Oxal Fe	Oxal Al
pH	1.00																			
Ash	-.022	1.00																		
TP	-.05	-.38	1.00																	
WEP	0.01	-.01	.06	1.00																
Oxal P	-.027	-.39	.61	.41	1.00															
Meh P	-.098	-.01	.02	-.02	.08	1.00														
MBP	-.118	-.12	.15	-.18	.10	.20	1.00													
TC	-.205	-.39	.67	.08	.43	-.12	.29	1.00												
TN	-.165	-.45	**.78**	.06	.51	-.13	.29	**.93**	1.0											
WEC	.030	-.46	.08	.31	.34	-.19	.03	.28	.22	1.0										
MBC	-.456	-.35	.47	-.16	.27	.03	.30	.63	.63	.02	1.0									
KClNH_4	0.011	-.46	.42	.40	.58	-.09	-.03	.38	.44	.26	.23	1.0								
MBN	.087	.35	.55	-.30	.23	-.14	.25	.55	.59	.07	.40	.09	1.0							
Meh Fe	-.188	-.37	.48	-.01	.45	-.20	.13	.67	.69	.25	.44	.35	.46	1.0						
Meh Al	-.464	-.23	.38	-.08	.30	.01	.18	.64	.53	.21	.49	.20	.30	.55	1.00					
Meh Ca	.414	-.19	.38	-.22	-.08	.03	.13	.21	.34	-.22	.09	.00	.44	.04	-.20	1.00				
Meh Mg	.2306	-.25	.52	-.21	.00	-.01	.16	.34	.47	-.15	.21	.06	.50	.19	-.00	**.91**	1.0			
Meh K	.0481	-.29	.58	-.26	.04	.03	.23	.50	.59	-.17	.35	.07	.56	.32	.23	**.78**	**.89**	1.0		
Oxal Fe	.0224	-.43	**.71**	.29	.69	-.35	-.03	.55	.66	.25	.31	.55	.41	.59	.21	.22	.33	.32	1.0	
Oxal Al	-.086	-.40	.67	-.15	.37	-.04	.07	.54	.56	.03	.38	.28	.46	.34	.50	.30	.42	.52	.55	1.0

Table 4. PLS calibration results from 20 soil biogeochemical variables using phase 1 soil data.

Soil Properties	Number	Mean Error	SEPR	EF^	RPDδ	RERδ
pH	397	0.009	0.510	0.112	1.061	7.847
Ash	397	0.004	0.047	0.220	1.322	11.08
Total Carbon	396	0.000	0.176	**0.746**	**1.984**	**14.02**
Total Phosphorus	397	0.000	0.173	**0.635**	**1.656**	**8.748**
Total Nitrogen	397	-0.010	0.180	**0.796**	**2.216**	**14.66**
Oxalate Al	397	-0.009	0.154	0.480	1.387	8.746
Oxalate Iron	396	0.064	0.348	0.532	1.462	7.7306
Oxalate Phosphorus	348	0.007	0.333	0.267	1.168	5.489
Mehlich Al	328	0.004	0.210	0.516	1.438	8.658
Mehlich Iron	326	0.010	0.276	**0.619**	**1.621**	**9.688**
Mehlich Phosphorus	397	-0.046	0.312	0.001	1.00	7.939
Mehlich Mg	328	-0.028	0.364	**0.694**	**1.808**	**6.560**
Mehlich Potassium	328	-0.018	0.256	**0.624**	**1.631**	**8.194**
Mehlich Calcium	328	-0.026	0.394	**0.605**	**1.591**	**7.466**
Microbial Carbon	395	-0.052	0.311	0.336	1.227	8.079
Microbial Nitrogen	301	-0.029	0.274	0.396	1.287	9.234
Microbial Phosphorus	297	-0.060	0.442	0.157	1.089	8.054
Water ext. Carbon	395	-0.068	0.373	-0.103	0.951	5.994
Water ext. Phosphorus	396	0.007	0.491	-0.158	0.928	5.424
KCl ext. NH_4	397	0.063	0.347	0.150	1.084	7.313

^ Forecasting Efficiency

R Standard Error of Performance

δ Defined in the text.

Table 5 Descriptive statistics of the 18 Biogeochemical variables used in this study-Phase 2

	N	Mean8	Meanδ	Max8	Maxδ	Min8	Minδ
pH	152	5.22	-	7.31	-	4.11	-
Ash [%]	152	1.95	89.12	2.00	99.9	1.64	43.65
TC [g/kg]	152	1.19	15.48	2.46	288.4	0.11	1.28
TP [mg/kg]	151	1.95	89.12	2.82	660.6	1.17	14.79
TN [g/kg]	146	-0.19	0.645	1.10	12.58	-1.4	0.039
Oxal Al [g/kg]	78	3.19	1548.8	3.79	6165.9	2.9	794.32
Oxal Fe [g/kg]	78	3.11	1288.2	4.45	28183.8	2.56	363.07
Oxal P [mg/kg]	78	1.42	26.30	2.46	288.4	0.81	6.45
Meh Al [mg/kg]	78	1.99	97.72	3.36	2290.8	0.77	5.88
Meh Fe [mg/kg]	78	1.49	30.90	2.83	676.08	0.36	2.29
Meh P [mg/kg]	152	0.06	1.148	1.29	19.49	-0.80	0.15
MehMg[mg/kg]	78	0.54	3.46	1.90	79.43	-1.24	0.05
Meh P [mg/kg]	152	0.06	1.14	1.29	19.49	-0.80	0.15
Meh Ca [mg/kg]	78	1.67	46.7	2.80	630.9	0.90	7.94
MBC [mg/kg]	152	2.455	281.8	3.52	3311.3	1.51	32.35
WEC [mg/kg]	152	1.76	57.54	3.02	1047.12	0.20	1.58
WEP [mg/kg]	152	-1.302	0.04	0.20	1.58	-2.09	.0008
KCl ext. NH_4 [mg/kg]	152	0.75	5.62	1.82	66.06	0.15	1.41

8the values are in log space

δ the values are in real space.

Table 6. Correlation matrix of the 20 Phase 2 biogeochemical variables, Fort Benning, Georgia.

	pH	Ash	TP	WEP	OxalP	MehP	MBP	TC	TN	WEC	MBC	KCL NH_4	MBN	Meh Fe	Meh Al	Meh Ca	Meh Mg	Meh K	Oxal Fe	Oxal Al
pH	1.00																			
Ash	-.022	1.00																		
TP	-.05	-.38	1.00																	
WEP	0.01	-.01	.06	1.00																
Oxal P	-.027	-.39	**.61**	.41	1.00															
Meh P	-.098	-.01	.02	-.02	.08	1.00														
MBP	-.118	-.12	.15	-.18	.10	.20	1.00													
TC	-.205	-.39	**.67**	.08	.43	-.12	.29	1.00												
TN	-.165	-.45	**.78**	.06	.51	-.13	.29	**.93**	1.0											
WEC	.030	-.46	.08	.31	.34	-.19	.03	.28	.22	1.0										
MBC	-.456	-.35	.47	-.16	.27	.03	.30	**.63**	**.63**	.02	1.0									
$KCLNH_4$	0.011	-.46	.42	.40	.58	-.09	-.03	.38	.44	.26	.23	1.0								
MBN	.087	.35	.55	-.30	.23	-.14	.25	.55	.59	.07	.40	.09	1.0							
Meh Fe	-.188	-.37	.48	-.01	.45	-.20	.13	**.67**	**.69**	.25	.44	.35	.46	1.0						
Meh Al	-.464	-.23	.38	-.08	.30	.01	.18	**.64**	.53	.21	.49	.20	.30	.55	1.00					
Meh Ca	.414	-.19	.38	-.22	-.08	.03	.13	.21	.34	-.22	.09	.00	.44	.04	-.20	1.00				
Meh Mg	.2306	-.25	.52	-.21	.00	-.01	.16	.34	.47	-.15	.21	.06	.50	.19	-.00	**.91**	1.0			
Meh K	.0481	-.29	.58	-.26	.04	.03	.23	.50	.59	-.17	.35	.07	.56	.32	.23	**.78**	**.89**	1.0		
Oxal Fe	.0224	-.43	**.71**	.29	**.69**	-.35	-.03	.55	**.66**	.25	.31	.55	.41	.59	.21	.22	.33	.32	1.0	
Oxal Al	-.086	-.40	**.67**	-.15	.37	-.04	.07	.54	.56	.03	.38	.28	.46	.34	.50	.30	.42	.52	.55	1.0

Table 7. PLS prediction results for 18 soil biogeochemical variables of phase 2 soil data>.

Soil Properties	Number	Mean Error	SEPRδ	EF^	RPDδ	RERδ
pH	124	0.036	0.457	0.000	0.990	6.973
Ash	124	0.023	0.056	0.101	1.055	5.787
Total Carbon	124	0.078	0.213	**0.868**	**2.760**	**10.98**
Total Phosphorus	123	0.006	0.176	**0.767**	**2.072**	**9.334**
Total Nitrogen	119	0.013	0.212	**0.888**	**2.989**	**11.70**
Oxalate Al	71	0.040	0.184	0.068	1.035	4.791
Oxalate Iron	71	0.062	0.245	**0.759**	**2.037**	**7.632**
Oxalate Phosphorus	71	0.019	0.273	**0.611**	**1.603**	**6.017**
Mehlich Al	71	-0.435	0.626	-0.158	0.928	4.107
Mehlich Iron	71	-0.228	0.461	0.416	1.309	5.319
Mehlich Phosphorus	124	-0.024	0.433	-0.049	0.976	4.826
Mehlich Mg	71	-0.604	0.923	-0.594	0.792	3.377
Mehlich Potassium	71	-0.460	0.694	-0.419	0.839	3.401
Mehlich Calcium	71	-0.036	0.444	-0.597	0.791	4.241
Microbial Carbon	124	-0.020	0.232	**0.705**	**1.842**	**8.596**
Water ext. Carbon	124	-0.180	0.507	0.041	1.021	4.775
Water ext. Phosphorus	124	-0.193	0.708	-0.044	0.978	3.227
KCl ext. NH_4	124	0.139	0.322	-0.305	0.875	4.602

^ Forecasting Efficiency

R Standard Error of Performance

δ Defined in the text.

> MBN and MBP didn't have measured data for the independent dataset.

Figure 1. Map of Fort Benning, Military Installation, Georgia, with sampling sites shown.

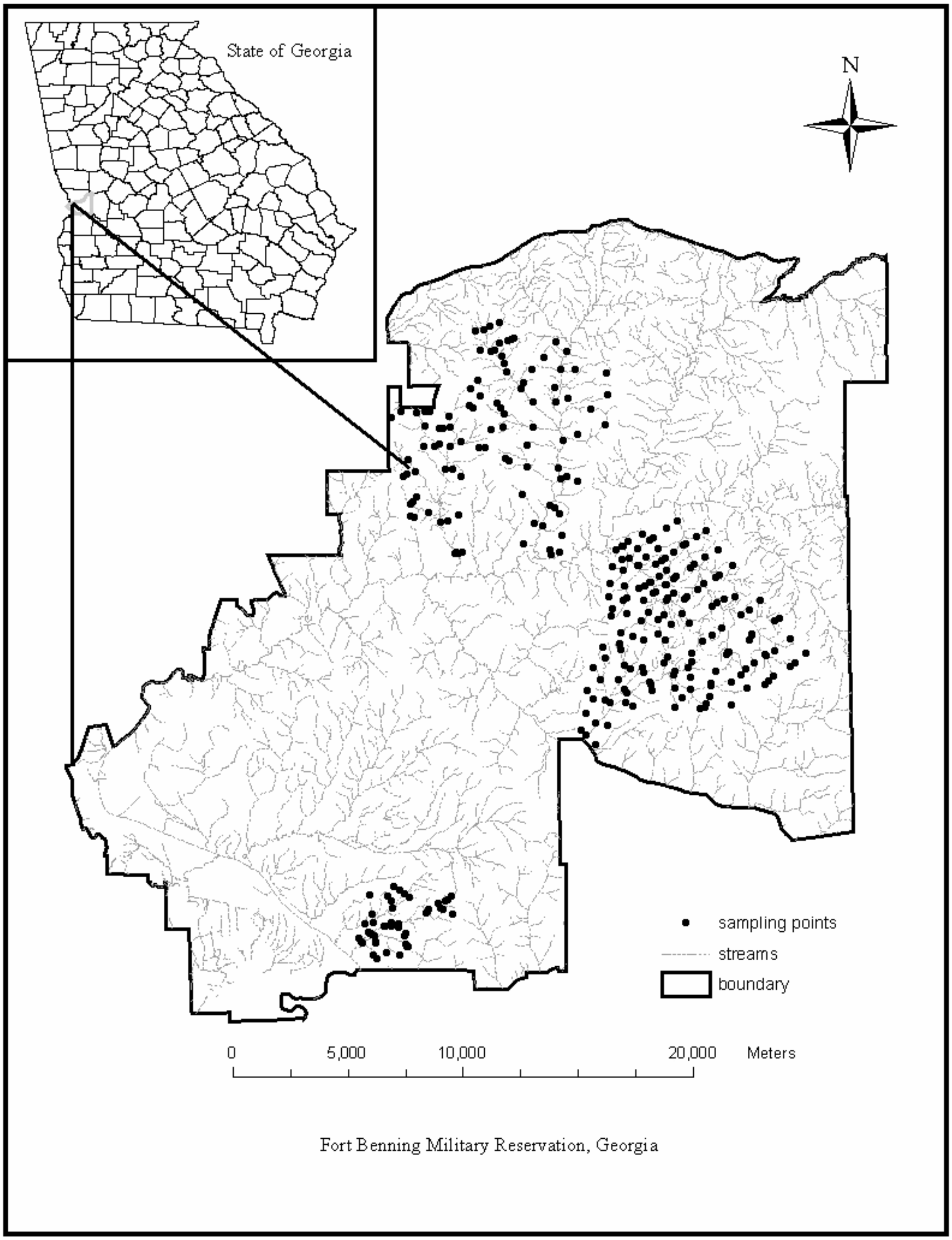
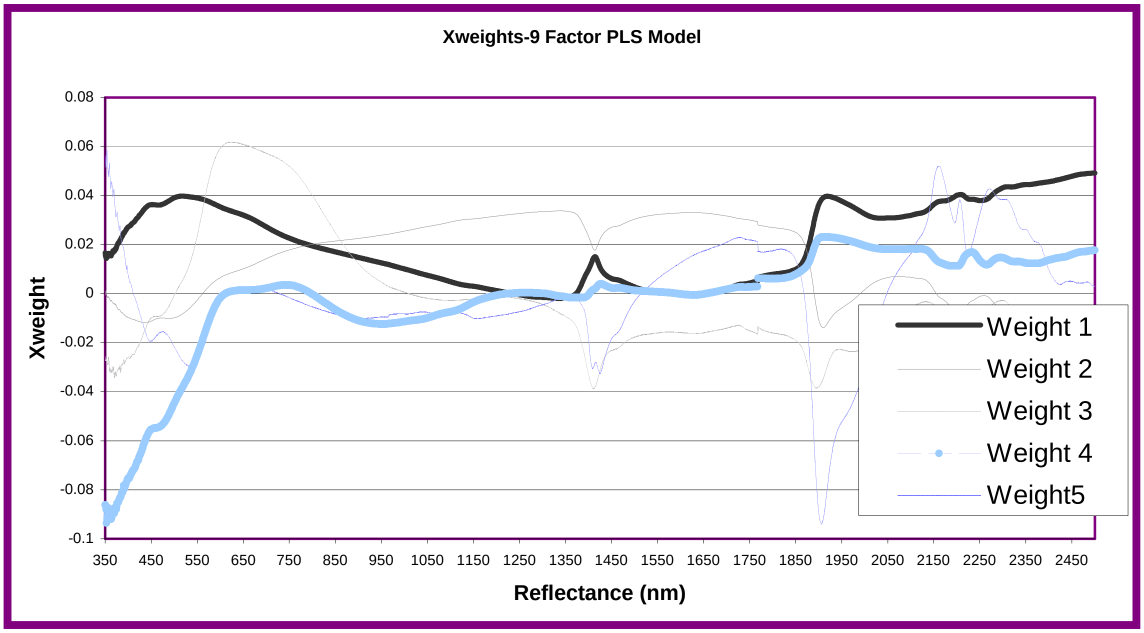


Figure 2. First nine latent factor weights across wavelengths for PLS model.



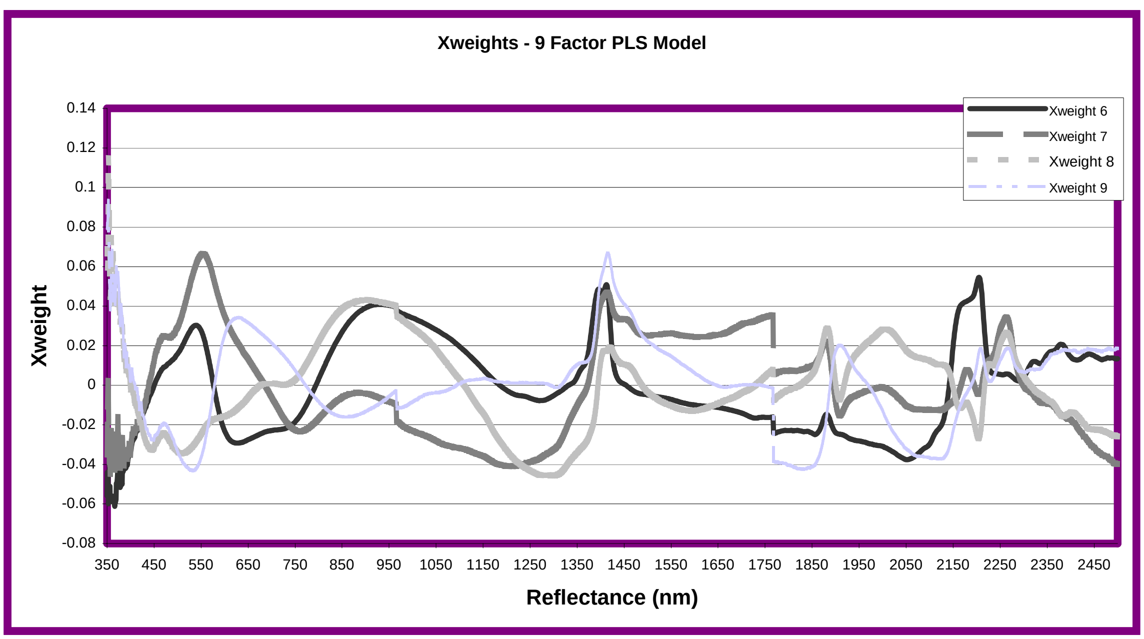
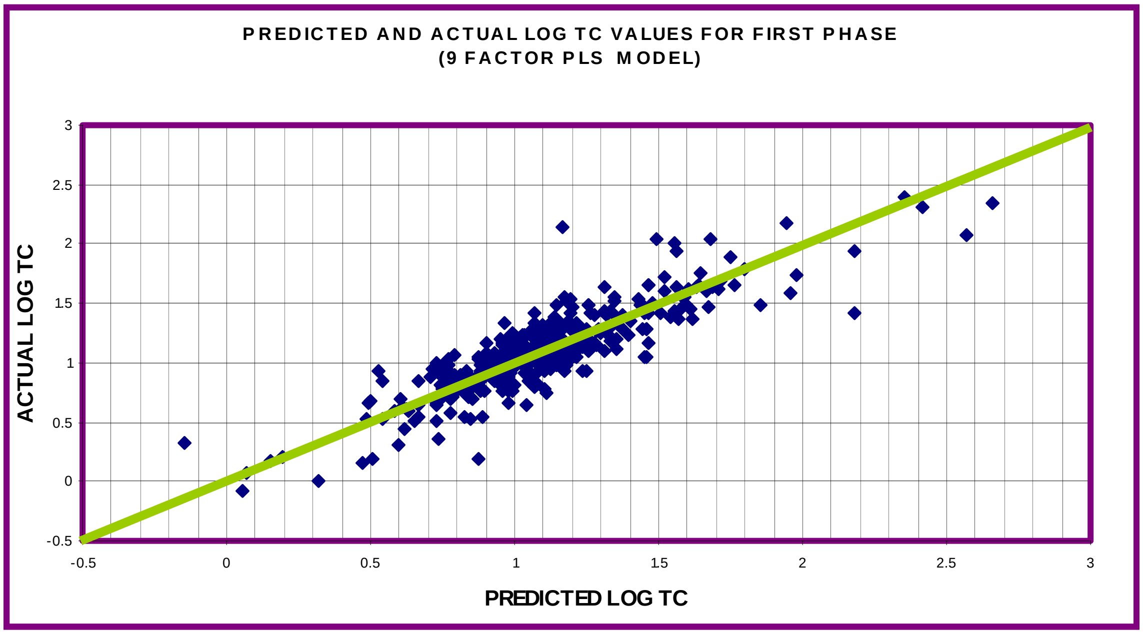
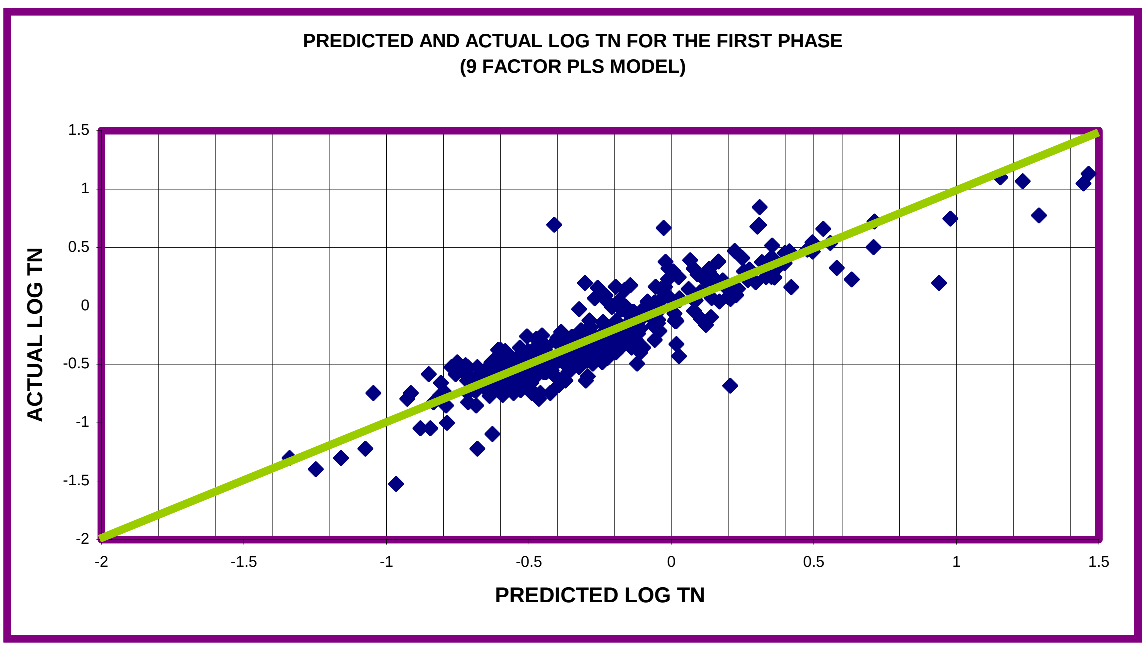
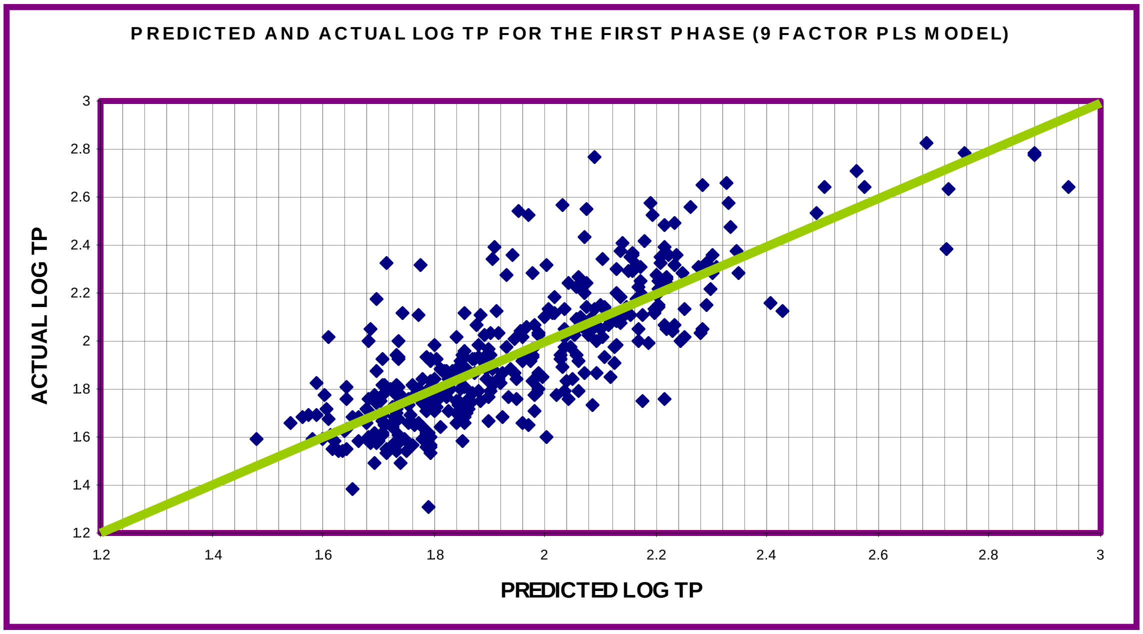
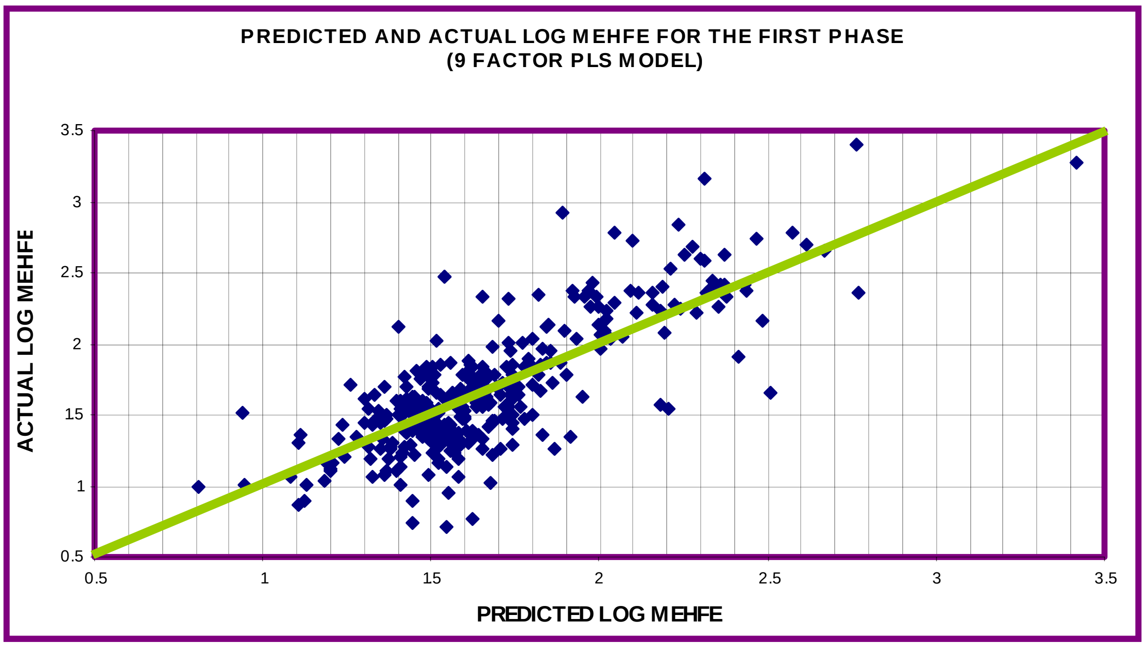


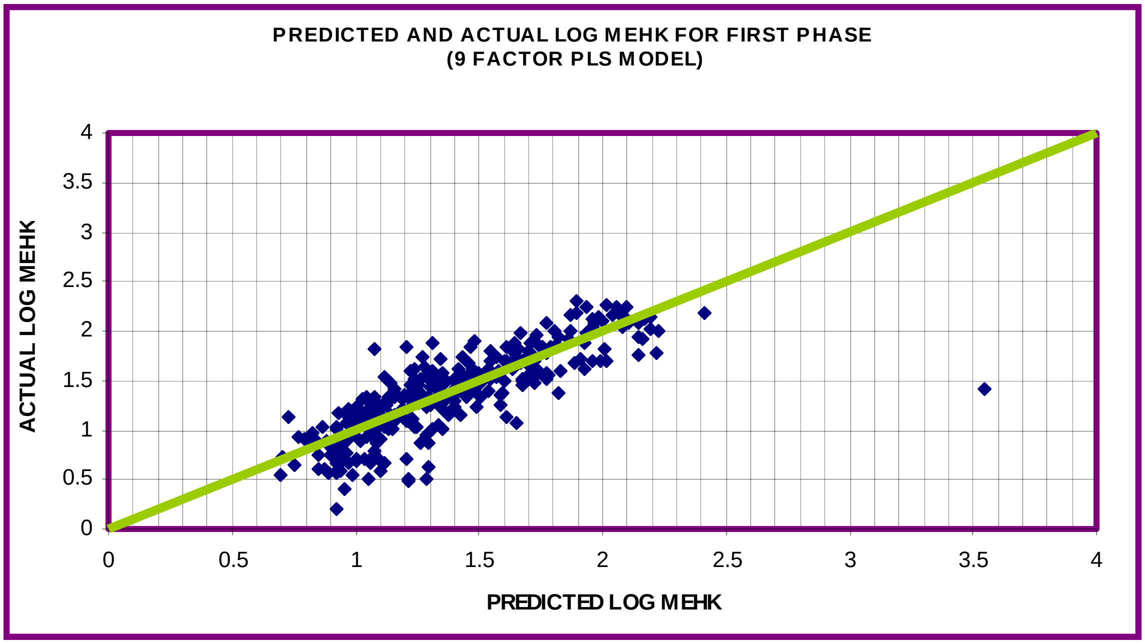
Figure 3. Relationship between actual soil properties as measured by standard lab procedures and lab reflectance predicted values using NIRS-PLS technique for phase1 dataset (Calibrated model).

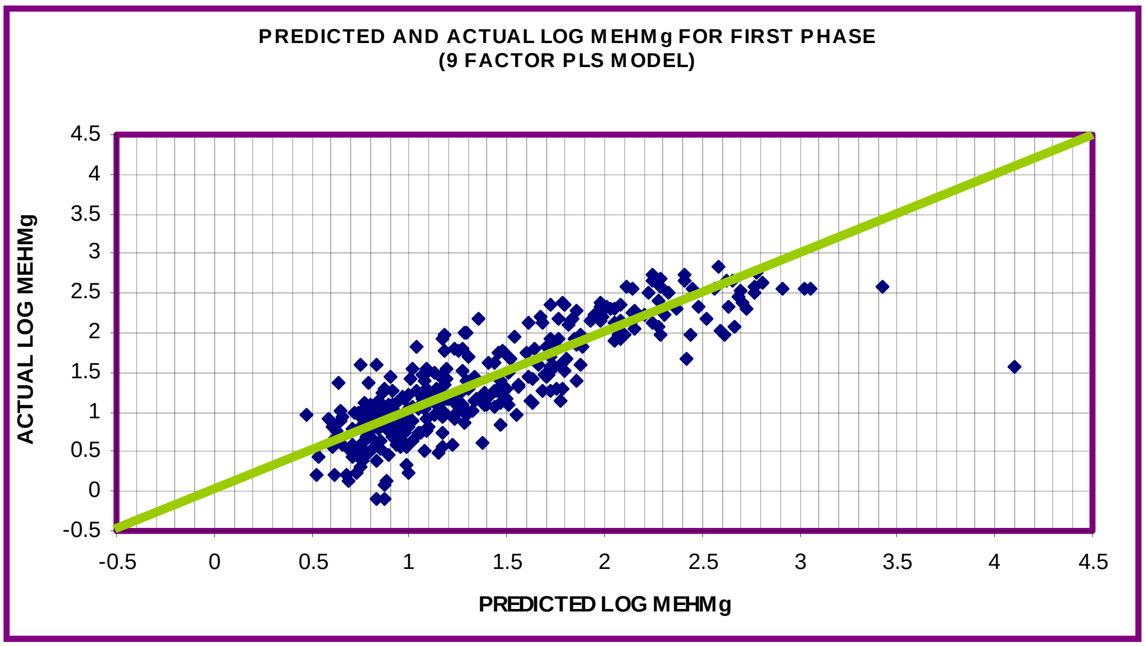


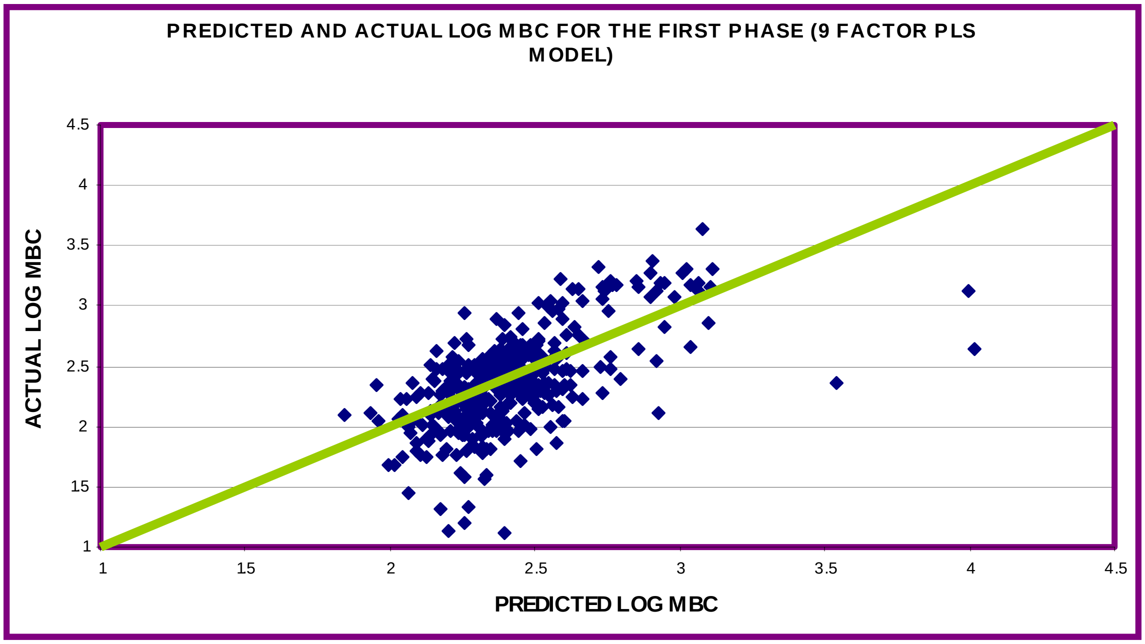



PREDICTED AND ACTUAL LOG TP FOR THE FIRST PHASE (9 FACTOR PLS MODEL)
ACTUAL LOG TP
PREDICTED LOG TP
3
2.8
2.6
2.4
2.2
2
1.8
1.6
1.4
1.2
1.2
1.4
1.6
1.8
2
2.2
2.4
2.6
2.8
3


PREDICTED AND ACTUAL LOG MEHFE FOR THE FIRST PHASE
(9 FACTOR PLS MODEL)
ACTUAL LOG MEHFE
PREDICTED LOG MEHFE
3.5
3
2.5
2
15
1
0.5
0.5
1
15
2
2.5
3
3.5


PREDICTED AND ACTUAL LOG MEHK FOR FIRST PHASE
(9 FACTOR PLS MODEL)
ACTUAL LOG MEHK
4
3.5
3
2.5
2
1.5
1
0.5
0
0
0.5
1
1.5
2
2.5
3
3.5
4
PREDICTED LOG MEHK


PREDICTED AND ACTUAL LOG MEHMg FOR FIRST PHASE
(9 FACTOR PLS MODEL)
ACTUAL LOG MEHMg
4.5
4
3.5
3
2.5
2
1.5
1
0.5
0
-0.5
-0.5
0
0.5
1
1.5
2
2.5
3
3.5
4
4.5
PREDICTED LOG MEHMg


PREDICTED AND ACTUAL LOG MBC FOR THE FIRST PHASE (9 FACTOR PLS MODEL)
ACTUAL LOG MBC
PREDICTED LOG MBC
4.5
4
3.5
3
2.5
2
15
1
1
15
2
2.5
3
3.5
4
4.5

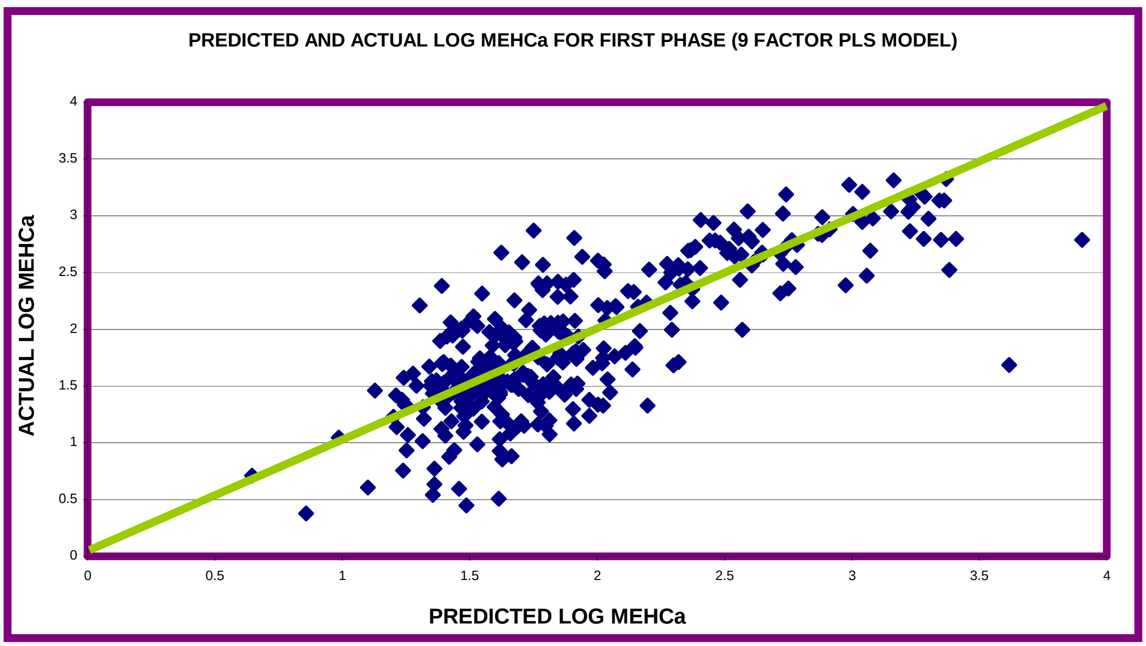
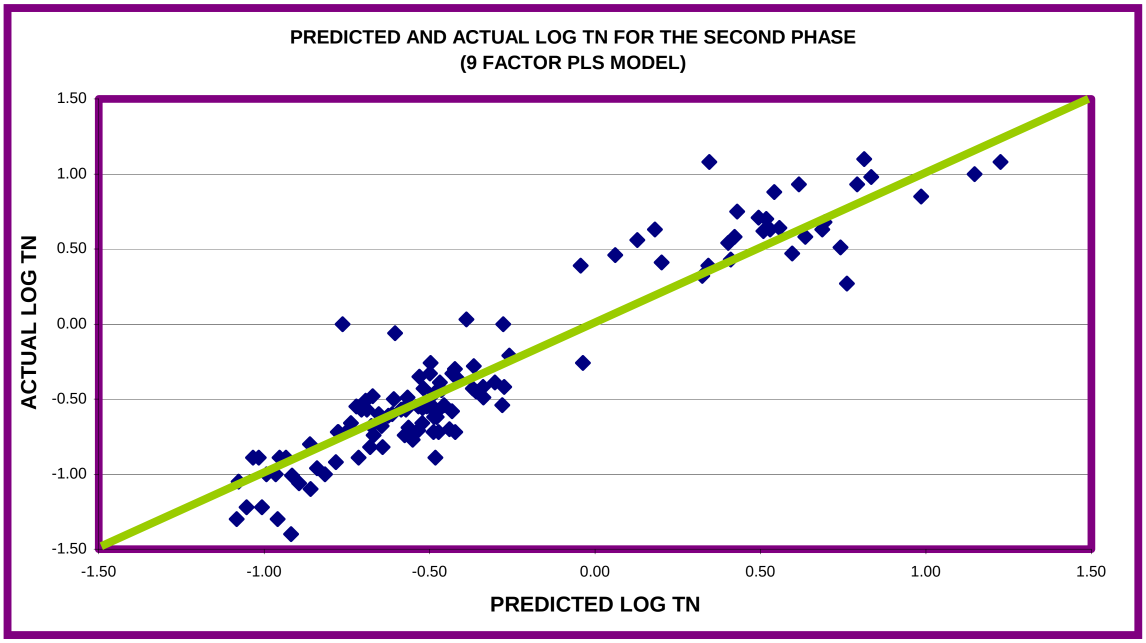
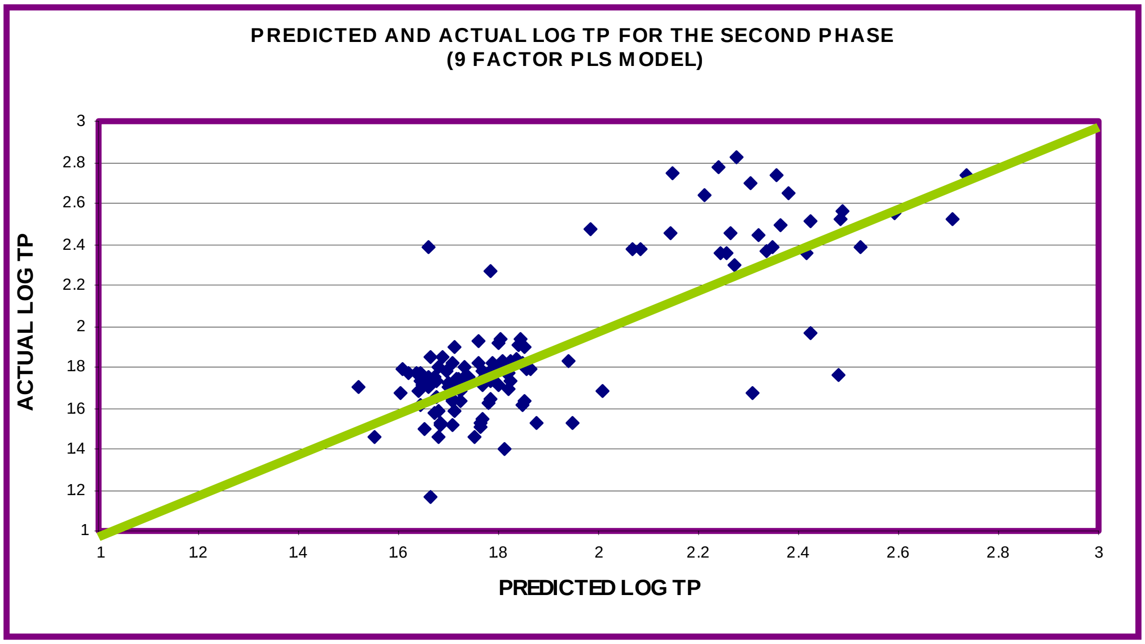
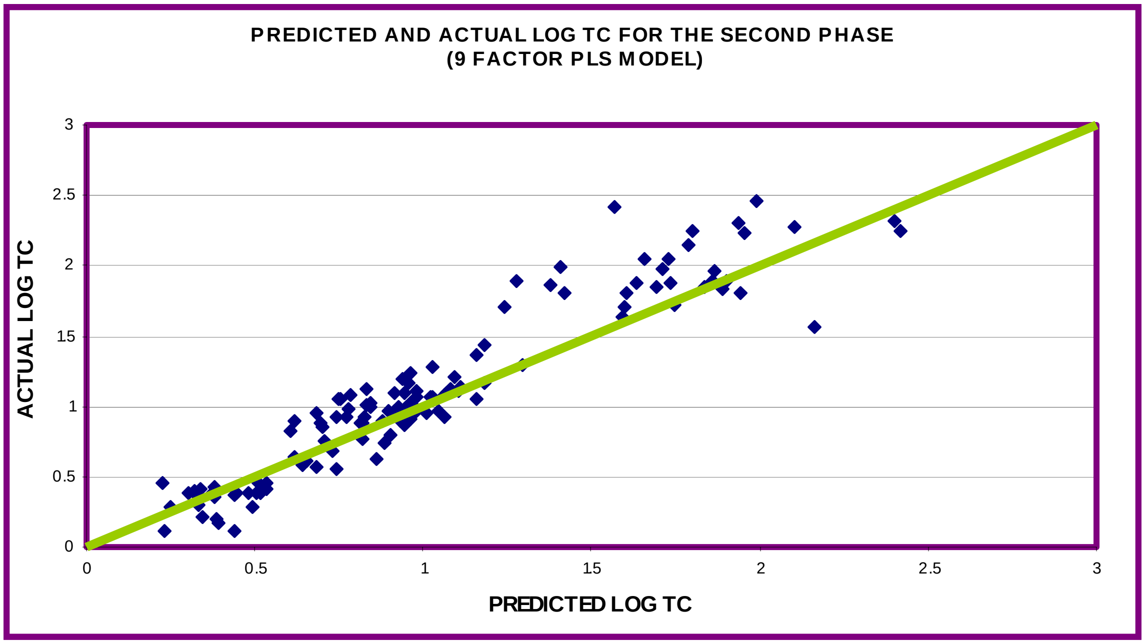
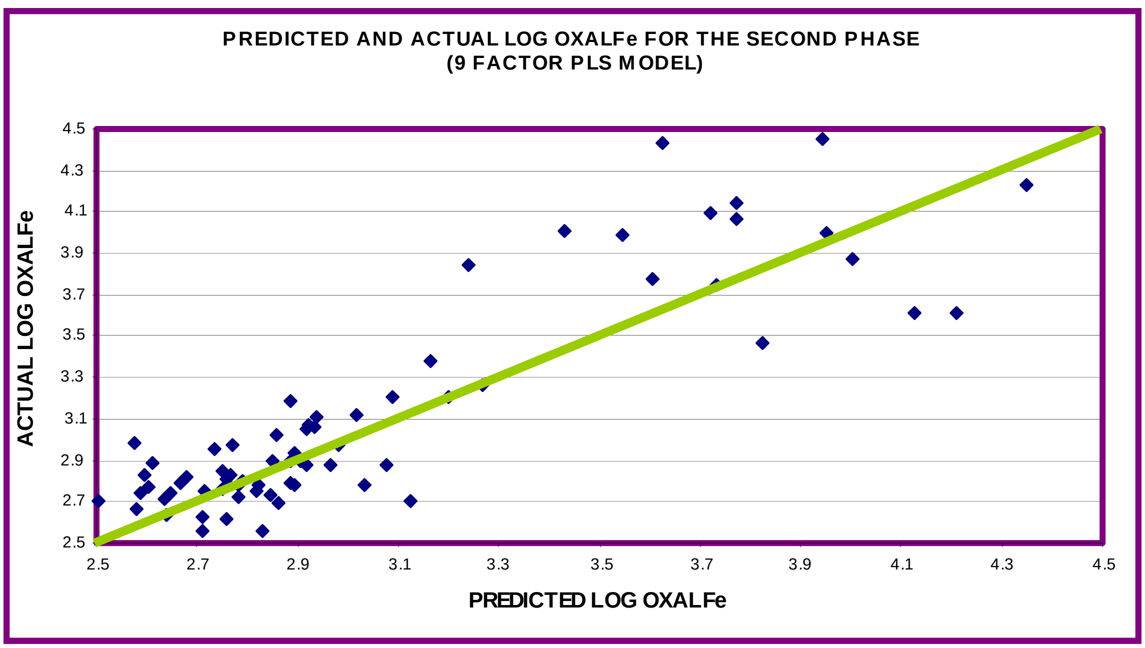

PREDICTED AND ACTUAL LOG MEHCa FOR FIRST PHASE (9 FACTOR PLS MODEL)
ACTUAL LOG MEHCa
PREDICTED LOG MEHCa
4
3.5
3
2.5
2
1.5
1
0.5
0
0
0.5
1
1.5
2
2.5
3
3.5
4

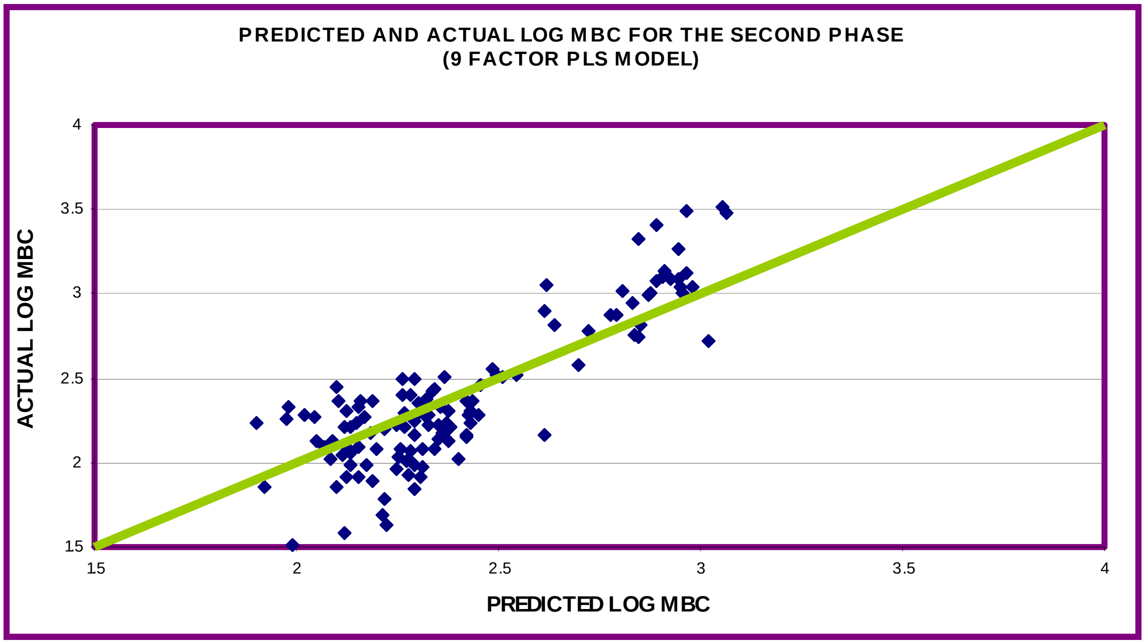
Figure 4. Relationship between actual soil properties as measured by standard lab procedures and lab reflectance predicted values using NIRS-PLS technique for phase 2 dataset (Validation set).

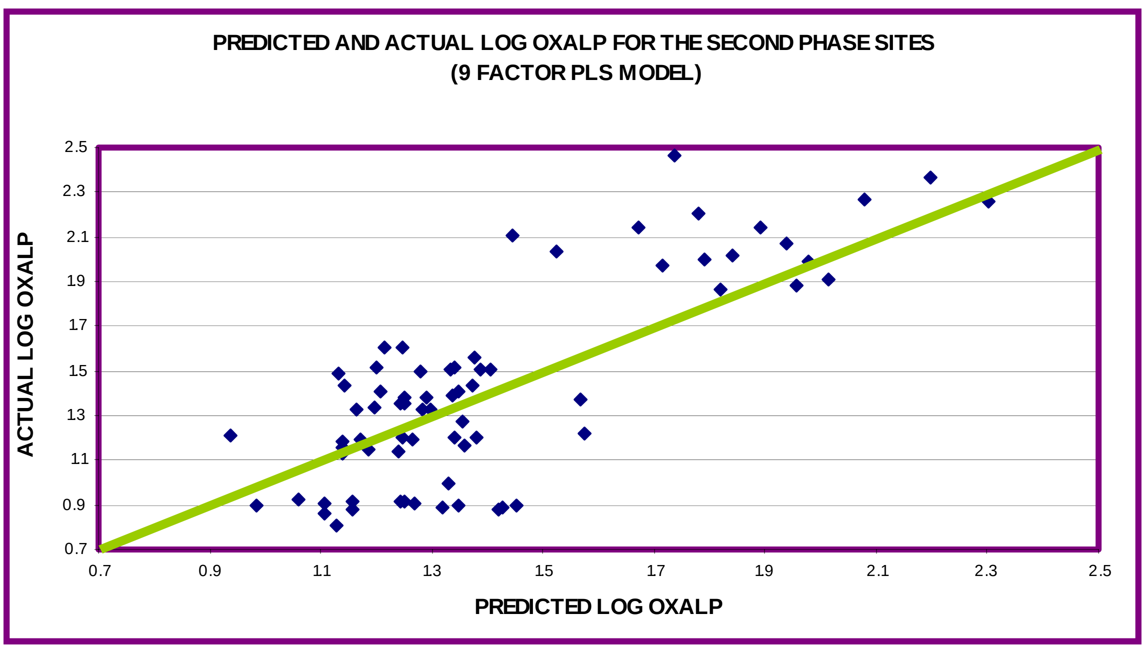





PREDICTED AND ACTUAL LOG TC FOR THE SECOND PHASE
(9 FACTOR PLS MODEL)
ACTUAL LOG TC
PREDICTED LOG TC
0
0.5
1
15
2
2.5
3


PREDICTED AND ACTUAL LOG OXALFe FOR THE SECOND PHASE
(9 FACTOR PLS MODEL)
ACTUAL LOG OXALFe
PREDICTED LOG OXALFe
2.5
2.7
2.9
3.1
3.3
3.5
3.7
3.9
4.1
4.3
4.5


PREDICTED AND ACTUAL LOG MBC FOR THE SECOND PHASE
(9 FACTOR PLS MODEL)
ACTUAL LOG MBC
PREDICTED LOG MBC
4
3.5
3
2.5
2
15
15
2
2.5
3
3.5
4


PREDICTED AND ACTUAL LOG OXALP FOR THE SECOND PHASE SITES
(9 FACTOR PLS MODEL)
ACTUAL LOG OXALP
PREDICTED LOG OXALP
2.5
2.3
2.1
19
17
15
13
11
0.9
0.7
0.7
0.9
11
13
15
17
19
2.1
2.3
2.5

3.6. Summary

Hydrologic, Soil, and Vegetation Indicators of Change in Forested Ecosystem: Synthesis of an Interdisciplinary Project

K. R. Reddy, W. F. DeBusk, W. Graham, J. Jacobs, D. Miller, J. Prenger, P. S. Rao, and G. Tanner

INTRODUCTION

Environmental protection in the U.S. is currently limited by the lack of 1) knowledge of mechanisms that control ecosystem structure and function and the effects of anthropogenic activities, and 2) sound methods to monitor important ecosystem characteristics and the temporal and spatial changes resulting from human perturbations (U.S. Environmental Protection Agency, 1997). These shortcomings point to the need for focused multidisciplinary research that examines ecosystem processes at multiple spatial and temporal scales. A better understanding of ecosystem structure and function is paramount to the development of sound concepts and methodology for ecosystem monitoring. However, given the high cost of environmental monitoring in terms of time, human resources and funding, it is important that we develop simple and efficient, yet scientifically rigorous and ecologically meaningful, methods. One of the most attractive approaches is based on the concept of using physical, chemical or biological properties or processes as indicators of ecosystem integrity, change or response to anthropogenic impacts. These indicators will provide early indications of change associated with (1) natural ecosystem variability and (2) anthropogenic activities, including changes in land management practices. Early indications of change, and an understanding of the likely causes, will improve ecosystem managers' ability to manage activities that are shown to be damaging, and prevent long-term, negative effects.

A suite of variables is needed to measure changes in ecological condition. Two types of indicators that may be useful are (1) variables that inform managers about ecosystem status and (2) variables that signal impending change. In many cases, these indicators may be the same. Both types are needed, but variables that serve as early warnings of impending changes outside the natural range of variation, and variables that are shown to be related to activities affecting the military mission, may be especially valuable.

This paper summarizes the results of an interdisciplinary study that was aimed to address the following objectives: (1) identify physical, chemical and biological variables (properties and processes) associated with soil, surface hydrology and vegetation that may be used as indicators of ecological change, (2) evaluate potential ecological indicators based on sensitivity, selectivity, ease of measurement and cost effectiveness, (3) select indicators that most effectively show a high correlation with a certain state in a specific ecosystem, provide early warning of impending change, and differentiate between natural ecological variation and anthropogenic negative impacts, and (4) determine the likely range of natural variation for indicator variables, and compare with the range of values under anthropogenic, especially mission-related, influences. Details of specific studies are reported in a series of papers published or currently in review (Archer and Miller, 2004; Bhat et al., 2004; Bryant et al., 2004; Dabrel et al., 2004a, b; Perkins et al. 2004a, b; Prenger et al. 2004a, b; Silveira et al., 2004a,b).

CONCEPTUAL FRAMEWORK

The conceptual framework for our study was based on a watershed approach, which recognizes discrete hydrologic units within which soil properties and vegetation community characteristics are shaped by flow and storage of water (Fig. 1). Mass flow of water (via overland flow, stream flow and groundwater flow) and entrained solutes and particulate matter (soil/sediment, dissolved and particulate organic matter and minerals) provides the mechanism for transfer of organic matter and mineral nutrients from uplands (ridgetops and slopes) to bottomlands (riparian zone, wetlands) in the watershed. Elevation gradients, in combination with variation in slope and aspect, give rise to soil moisture gradients and vegetation community and productivity gradients. Wetlands serve as sinks for matter (particulate and dissolved compounds) and energy (organic C) in the watershed. Wetland biogeochemical properties typically reflect and integrate land use and disturbance characteristics in the watershed. For example, wetlands in ecologically impacted watersheds may exhibit increased sedimentation from accelerated erosion in the watershed, increased productivity and eutrophication from nutrient loading, or altered hydroperiod from short-circuiting of subsurface flow and soil water storage due to gulley formation.

Our overall objective was to characterize "baseline conditions" of watershed hydrology; soil hydraulic, chemical and biological properties; vegetation species composition and density; then evaluate the response of related variables (potential indicators of ecological condition) across a wide range of disturbance due to military and related anthropogenic activities. In general, the soil hydrologic and biogeochemical parameters relate to changes in soil physical and chemical characteristics, and the response of soil microbial population and plant communities. Cause and effect relationships developed between environmental changes, due to both natural variability and anthropogenic perturbation, and soil and vegetation responses, primarily as they relate to nutrient storage, nutrient turnover and population dynamics, can aid in determining anthropogenic impacts on the ecosystem. Our conceptual approach focuses on four ecological components of a watershed: 1) upland soils, 2) vegetation, 3) surface hydrology and 4) wetlands and streams (Fig. 2). The interactions and linkages among these components may exert significant influence on habitat quality in the watershed, and on downstream water quality. Our selection of ecological indicators was based in part on their ability to accurately reflect these ecological processes.

SITE DESCRIPTION

The study area is located at the Ft. Benning military installation in west-central Georgia (Fig. 3). This area lies immediately to the south of the fall line, within the fall line hills district of the Coastal Plain physiographic province and the Carolina and Georgia sand hills major land resource area (USDA, 1997). The climate is characterized by hot summers and mild winters, with an average annual rainfall of about 52 inches. The topography of this area is nearly level to gently sloping ridgetops, moderately steep and steep hillsides, and nearly level valleys along stream channels and other tributaries. Upland soils in the area are primarily well to excessively drained Ultisols and Entisols, supporting forests of slash (*Pinus elliottii* var. elliotti), longleaf (*Pinus palustris*), and loblolly pine (*Pinus taeda*). Troup loamy sand (Grossarenic Kandiudults) is the most widespread soil type in uplands of the central and northern portion of Fort Benning. "Loam hills," dominated by Nankin sandy clay loam (Typic Kanhapludults), occur in a broad band across the southern portion of the installation. Sandhill communities are associated with excessively- drained ridgetops in the central and northern portion of Ft. Benning. They are underlain by the Lakeland series (Typic Quartzipsamments) and feature longleaf pine, turkey oak (*Quercus laevis*), blackjack oak (*Quercus* *marilandica*), and post oak (*Quercus stellata*). Wetlands and hydric soils (principally Bibb sandy loam [Typic Fluvaquents]) are restricted to foot and toeslope bottomlands along streams and creeks.

Natural resource management includes forest management for timber harvest, prescribed burning (3 year rotation), and wildlife management. Forest impacts at Fort Benning, besides timber harvesting and planting, also include mechanical and human military training. Prescribed burning, of variable frequency and intensity, is used to improve or maintain habitat and control hardwood regeneration, reduce forest fuel loads, and prepare the site for regeneration.

MATERIALS AND METHODS

Analyses of soil biogeochemical properties and vegetation community composition were conducted in two phases: (1) landscape level assessment, to establish the range and spatial distribution of parameter values in the study area (Phase 1), and (2) spatially-intensive measurement of selected parameters across topographic, ecological and disturbance gradients (Phase 2). Evaluations of soil hydraulic properties and watershed hydrology were focused on paired watersheds representative of highly disturbed and minimally disturbed (baseline) conditions within the Fort Benning study area.

Landscape Assessment of Soil Biogeochemistry and Vegetation

Surface soil samples were obtained within 6 watersheds in the Fort Benning installation: Sally Branch, Bonham Creek, Halloca Creek, Randall Creek, Wolf Creek, and Shell Creek (Fig 3). Soil sampling sites (approximately 300) were located along transects spanning individual watersheds and perpendicular to the main stream channel, providing uniform coverage of the watersheds and capturing the hydrologic gradient in each. Sampling site locations were adjusted along the transects to provide equal representation of three landscape positions, i.e., ridge or hill top, side slope, and bottomland (wetland) areas within each watershed. Thus, approximately two thirds of the sites were located in upland areas and one third in bottomlands. Each sampling site was also classified according to the level of disturbance to soil and vegetation resulting from military training or other land use / land management activities (e.g., timber harvest). On-site inspection, augmented by aerial photography and other documentation of activity, was used to assess site condition and assign each site to one of three disturbance levels: low, moderate or severe impact.

One composite soil sample was obtained at each site, consisting of five 20-cm deep sub-samples taken by 1-inch diameter soil probe within a 1 m^2 quadrat. Triplicate samples were taken at approximately 20% of the sites, to provide an estimate of within-site variability. Structural and compositional parameters of the vegetation were measured at the same sampling sites. Understory woody plants canopy cover (< 2m tall) along three 5-m transects and overstory canopy cover (densiometer) were measured at all locations. Understory composition and cover and biomass, including litter were measured within three 1-m^2 quadrats at the triplicate soil testing sites within the watersheds (n= 56).

A suite of chemical and biological properties, related to nutrient storage and cycling, was measured for each soil sample. These included Bulk density, pH, total nitrogen (N), carbon (C), and phosphorus (P), ammonium (NH_4^+), water extractable C and P, Mehlich-1 extractable iron (Fe), aluminum (Al), calcium (Ca), magnesium (Mg), potassium (K) and P; oxalate extractable Fe, Al, and P; microbial biomass C, soil (microbial) respiration, and microbial enzyme activity (dehydrogenase, peptidase, acid phosphatase, and beta glucosidase).

Spatially Intensive Measurement of Soil Biogeochemical and Vegetation Parameters

Variability of soil biogeochemical properties across land use and land disturbance gradients:

Based on the results of initial sampling and data evaluation, additional soil samples were obtained from well-defined impact gradients on a smaller scale in select watersheds. Upland transects ranging from 200 to 400 m in length were established in upland areas of severe military impact (2 transects), managed forest (3), and low impact (1). Transects of varying length were also established in wetlands downslope from each of these areas. Soils along upland transects were sampled at 20 m intervals, to a depth of 20 cm (composite of 5 subsamples), while wetland soils were sampled at 5 to 20 m intervals based on overall transect length, to a depth of 5 cm (composite of 3 subsamples). The transect-based sampling design facilitated comparison of indicator response in areas of high and low disturbance and, simultaneously, an evaluation of local, within-site variability. The soil characteristics and properties evaluated for this phase of the study were total C, N and P, pH, organic matter, exchangeable NH_4^+, potentially mineralizable N, microbial biomass C and N, soil respiration, Mehlich 1 and 3 extractable P, HCl and ammonium oxalate extractable P, Fe and Al, and microbial enzyme activities (acid phosphatase, beta glucosidase and dehydrogenase).

Response of vegetation ground cover to disturbance associated with clear cutting

A chronosequence study focusing on recovery of ground cover vegetation after clear cutting was conducted in 2001/2002. Ground cover vegetation was assessed within two major soil groups (loamy vs sandy soils) and four time intervals after logging for a total of 32 sites. Military activity for these sites was low to moderate. Identification of pattern and rate of ground cover recovery following clear cutting will aid in identification of sensitivity and rate of return of herbaceous species following low to moderate levels of disturbance and further separate natural variation from variation attributed to anthropogenic disturbance.

Within each soil type, 4 sites were selected from each of the following categories representing time since last clear-cut: 0-3, 8-10, 18-20, and >30 years. Potential sites were subjected to the same logging techniques, which included roller chopping and burning but no herbicides and had similar fire histories and slope (0-6%). While all sites were clear-cut, only those with > 30 yr. old stands were thinned. All pine stands within the installation that fit these criteria were compiled into a list from which study sites were randomly chosen. The 0-3 yr. sites were longleaf plantations, with no overstory and generally high ground cover. The 8-10 year sites were either longleaf or loblolly plantations (all plantations were loblolly before 1996) with no overstory above 10 feet. While overstory cover increased and ground cover decreased for 15-20 year sites, the highest canopy cover was found on the oldest sites (>30 years).

Five random subplots were selected at each of the 32 sampling sites. Each subplot was categorized as: skid trail/road, low disturbance or unknown based on a visual assessment of the disturbance. Overstory canopy cover was measured with a concave spherical densiometer by averaging the readings of the four cardinal directions from the center point of the subplot. Radiating from the center point, three-meter transects were established at 0°, 120°, and 240°. Along each transect, woody (<2m in height) cover by species was measured. Aerial herbaceous vegetation cover by species was estimated using foliar ocular observation in 1 m^2 quadrats at the center point and at the terminus of the 240° transect.

An undisturbed soil core was taken adjacent to each herbaceous quadrat for laboratory bulk density determination. Additional soil samples were collected at the terminus and centerpoint of each transect and at each center point for a total of 4 samples; 20 overall for the site. These samples were combined in a single container for each subplot. Texture, pH, organic matter, total nitrogen and total carbon were measured for these composite samples.

Watershed Hydrology

Throughfall

Throughfall was measured simultaneously at Fort Benning in western GA for the five forest communities that are characteristic of the region: mature pine, 13 year-old pine plantation, lowland hardwood, upland hardwood, and mixed upland hardwood/pine. Canopy parameters, climatic variables, and interception components were measured throughout the study period. The measured data include precipitation, throughfall, stemflow, atmospheric conditions, and canopy cover.

A rectangular plot was established in each forest community. The plots were randomly selected within areas having vegetation that is consistent with the average vegetation density and distribution for the respective forest community (Fig. 4). The dimensions of each plot ranged from 10 x 40 m in wetland area in the riparian corridor to 30 x 30 m in upland areas. The plot in the riparian corridor was designated as Wetland (WET). Similarly, the plots in the upland area were designated as Pine (PIN), Pine Plantation (PIP), Hardwood (HRD), and Mixed (MXD). Each plot was subdivided into four sampling grids of equal size. Although these sampling grids consist of similar tree species, there are differences in number of trees per grid, percent of total number by tree types, average tree height and diameter, and trees per hectare. Each grid was outfitted with four throughfall collectors and one tipping bucket rain gauge for a total of 20 sampling points per plot. The throughfall collectors and tipping buckets were randomly placed on the ground within the confines of the grid. As the average separation distance necessary to ensure independent measurements likely would have extended plots beyond a single forest type and density (Loescher et al., 2002), each instrument was relocated randomly within the grid after one set of data were collected to reduce the standard error of estimation (Lloyd and Marques, 1988). The throughfall data were collected using 203.2 mm diameter tipping bucket rain gauges (model RG-100a, RainWise®) and 152.4 mm diameter throughfall collectors on a bi-weekly basis. The tipping bucket measurements were aggregated to the same biweekly periods as the throughfall collectors. Canopy cover was determined by direct measurement with a Model-A spherical densiometer using the method outlined by Lemmon (1956).

Infiltration

In situ steady-state infiltration rate was measured at most of the SWRC sampling sites, using a single-ring, constant-head infiltrometer (Perkins et al., 2004). Infiltration rate was calculated using the method described by Elrick and Reynolds (1992), which considers corrections for a 3-dimensional (or "bulb") wetting front underneath the single-ring infiltrometer. Saturated hydraulic conductivity (K_{sat}) was estimated using the field measurements, and compared with K_{sat} estimates based on constant-flux infiltration and constant negative head infiltration. Differences in K_{sat} values measured using these three methods were explained on the basis of sampling support (wetted volume) and variations with depth.

Undisturbed soil cores were sampled to a depth of 15 cm at sites within several sub-watersheds at Fort Benning. A total of 33 cores were collected from low-impact training sites and 15 from mechanized training areas (high impact), within the Bonham Creek and Sally Branch watersheds. Predominant soil types at the sampling sites were Troup (loamy, siliceous, thermic, Grossarenic Kandiudults) and Lakeland (thermic, coated Typic Quartzipsamments) series. Upland soils in the sampled areas have similar A and E horizons, and differ primarily in the depth to the diagnostic horizon. Soil water retention characteristics (SWRC) of the soil cores were determined under laboratory conditions (Perkins et al., 2004). Soil-water retention curves were measured for each soil core, using the TEMPE cell method (Flint and Flint, 2002). Using a physically based scaling approach (Kosugi and Hopmans, 1998), soil water retention data were fitted to a water retention function, which was used to estimate a scaling factor that is directly proportional to the median pore size of the soil (Perkins et al., 2004). Following measurement of SWRCs, soil cores were oven-dried at 105 °C for a minimum of 24 hours, and bulk density was determined from mass and volume of soil samples.

Streamwater Flow and Quality

Relationships among watershed physical characteristics and stream flow quantity and quality were analyzed at a watershed scale in Fort Benning. For the streamflow hydrograph analysis, the Fort Benning study watersheds, Bonham-1 and Bonham-2, Bonham, Little Pine Knot, and Sally Branch (named for the creek/stream that drains the watershed), were selected to represent a range of the region's soils, topography, land use, and vegetation communities. In addition to standard watershed characteristics, a dimensionless military disturbance parameter was calculated. The parameter, disturbance index (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., 2004).

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.

Both annual and storm-based analyses were used to identify ecological indicators. 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, the hydrologic indices are adapted from the procedure outlined in the Indicators of Hydrologic Alteration (IHA) method (Richter et al., 1996). The adaptation uses only non-redundant yet biologically significant hydrologic indices as per the recommendations of Olden and Poff (2003). In this study, the annual-based indices are used to contrast adjacent 2nd order sub-basins with similar soil and vegetation: a minimally-impacted watershed ("Bonham-1") and a moderately- to highly-impacted watershed ("Bonham-2"). Bonham-2 encompasses a portion of the Rowan Hill tank training area, while Bonham-1 is not affected by mechanized training.

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. Four groups of hydrologic characteristics, based on ecological function, that are relevant to storm-based hydrologic indices having a total of 18 specific storm-based ecologically relevant hydrologic indices were used to characterize variation in water condition in individual watersheds. 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.

For the water quality analysis, Bonham-1 and Bonham-2, Bonham, Little Pine Knot, Sally, Oswichee, and Randall (named for the creek which drains the watershed) watershed were selected to represent a range of region's soils, topography, land use, and vegetation communities. 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).

Soil-water storage

Soil-water content and storage dynamics play a dominant role in determining hydrologic (e.g., infiltration and runoff) and biological processes (e.g., biogeochemical rates; plant-water stress) in watersheds. Spatial distribution of soil hydraulic properties was measured in a low impact sub-watershed of the Bonham Creek mixed-impact watershed (Perkins et al., 2004). Every two months for a one-year period (June 2001 – June 2002), point water content measurements were obtained in the Bonham-1 watershed using the Delta-T® TH2O Soil Moisture Meter. Sample locations were predetermined at relatively regular intervals over the 95.1 Ha (~0.3 sq. miles) watershed using 50-meter contour lines as references. Measurements were used to estimate the total water storage and spatial moments of water content within the catchment. Near-stream spatial soil saturation limits were recorded to compare previous near-stream saturation delineation. At each sample location, water content measurements were taken at the soil surface as well as depths of 15, 30, 45, 60, and 75 centimeters by first digging with a soil auger to the desired depth, inserting the probe into the soil, and then obtaining a water content reading. This process was repeated at all sampling locations on a given sampling campaign (about 50).

Each depth was treated as a horizontal cross-section of the watershed and was analyzed separately for estimating soil-water storage. In order to interpolate water content between measured points, a statistical distribution of water content was computed for each depth to eliminate potential outliers. Then variograms were computed and used to develop spatial water content models by ordinary kriging. GEO-EAS® software (EPA software) was used to calculate and assign unbiased water content values over the Bonham-1 watershed for each depth. Maps of soil-water storage distribution were generated from the GEO-EAS grid output using ARCVIEW®.

Surface hydrology measurements were concentrated in the Bonham Creek watershed during Phase 1. Adjacent sub-basins (2nd order) with similar soil and vegetation were selected for initial studies: a minimally-impacted watershed ("Bonham-1") and a moderately- to highly-impacted watershed ("Bonham-2"). Bonham-2 encompasses a portion of the Rowan Hill tank training area, while Bonham-1 is not affected by mechanized training. Canopy throughfall collectors were installed at 68 sites in Bonham-1 and Bonham-2 watersheds. These, along with precipitation collectors, were monitored regularly to provide calibration data for throughfall and other hydrologic models. Streamflow measurement (stage) stations were established near the discharge points of the Bonham tributaries that drain the 2 sub-basins under study.

RESULTS AND DISCUSSION

Soil Biogeochemistry

Landscape evaluation of soil properties

Landscape evaluation of soil chemical and biological characteristics at 300 sampling sites revealed a high level of spatial variability for all parameters measured. Among the major factors contributing to landscape- or watershed-scale variability in soil chemistry were differences among underlying geological features, soil types, and current/historical land management practices. At the local (e.g., less than 1 km^2) scale, variability was due primarily to landscape position and anthropogenic disturbance (e.g., intensive military training and logging). Among low-impact sites, which were used to establish a background or "reference" condition for evaluation of military impacts, significant differences in soil chemistry were observed between upland and bottomland areas. These differences were largely related to natural accretion of organic matter in wetlands, due to higher productivity and lower decomposition rates relative to uplands, as well as allochthonous inputs of silt and clay fractions from the ubiquitous erosional and leaching processes in upland soils of this region. For example, within low-impact areas, soil pH was slightly lower in bottomlands than in uplands; and concentrations of total C, N and P, ammonium, water extractable C and P, and Mehlich-1 and oxalate extractable Fe, Al, and P were higher in bottomlands than in uplands (Table 1). In contrast, Mehlich-1 Ca, Mg and K concentrations were higher in upland soils than in bottomlands. Movement of Fe and Al from upland soils into bottomlands may be closely associated with erosion and overland (downslope) transport of the clay fraction. Entrainment of Fe- and Al- phosphates in the clay fraction may account for mass transport of P from uplands to bottomlands.

Military training impacts, as well as other significant soil/vegetation disturbances, have accelerated natural erosional processes, resulting in loss of topsoil in many upland areas and, consequently, depletion of soil organic matter. Accordingly, thickness of the soil A horizon , measured at selected sites across a military impact gradient within the Fort Benning study area, decreased with increasing soil disturbance (Fig. 5). Field assessments and soil chemistry analyses indicated that, although widespread sedimentation of wetlands with clay, silt and occasionally sand, had occurred, there was no evidence that soil organic matter from uplands had been deposited in wetlands. Thus, it is likely that this organic material was flushed out of the watershed in streamflow, resulting in a decrease in organic content of disturbed wetland soils through "dilution" by inorganic soil material.

In general, for both wetlands and uplands, soil total C (roughly equivalent to organic C in this low-carbonate system) and other chemical and biological parameters associated with soil organic matter tended to decrease with increasing site disturbance (Fig. 6). The ratio of soil microbial biomass to total C (MBC:TC) increased with increasing soil disturbance, which likely reflects the relative availability, or lability, of organic C to heterotrophic microorganisms in the soil. It appears that the loss of soil organic matter near the soil surface through topsoil erosion in uplands or sedimentation in wetlands results in a higher proportion of freshly-deposited organic material in the soil organic matter pool, thus stimulating microbial growth. Soil enzyme levels were generally related to soil total C content, due to their association with organic C utilization by soil microorganisms. In particular, β-glucosidase, an enzyme involved in the degradation of complex carbon compounds to monosaccharides, best distinguished between low-, moderate- and severe-impact areas.

Spatially-intensive evaluation of soil properties

Phase 2 data, which was relatively site-specific compared to Phase 1 data, revealed similar trends in soil C and microbial biomass in response to site disturbance (Fig. 7). For this spatially-intensive phase of the study, all upland transects were located within soil map units dominated by the Troup series, and wetland transects were located within the Bibb series, thus the "background" variability in soil physical and chemical properties was substantially reduced. A portion of this study was conducted along transects spanning military and non-military disturbance/land use gradients at paired (severe- and low-impact) sites. Total C and total N decreased with the level of impact, as shown in Phase 1 of the study. Microbial biomass C (MBC) was also significantly lower in high impact areas, symptomatic of mineralizable C limitations to microbial activity. As observed in Phase 1 of the study, the contribution of the microbial biomass to the overall C pool (MBC:TC) increased with level of impact, indicating a shift from storage of stable humic materials to bioaccumulation of C in response to disturbance (Insam & Domsch, 1988). Accordingly, the ratio of labile C to total C increased with level of impact, indicating that SOM at highly-impacted (eroded) sites was predominantly "new" organic matter with a relatively high turnover rate. The metabolic quotient, defined as microbial respiration:biomass C ratio (Insam & Domsch, 1988), decreased at high impact sites, indicating that microbes were more efficient in converting a higher proportion of C into biomass than in low impact sites.

Multivariate statistical analyses were performed on the Phase 1 soil biogeochemical data set. Cannonical Discriminant Analysis was used as to reduce the dimensionality of the multivariate data set while maximizing the separation between specific categories of data. Discriminant Function Analysis was used to classify observations into groups on the basis of the biogeochemical data set. Results indicate that canonical variable 1 provides relatively good separation among sites designated as low and moderate, while canonical variable 2 primarily provides separation of severe-disturbance sites from those with low to moderate disturbance. Results of Discriminant Function Analysis indicate that the Phase 1 soil biogeochemistry data "predict", to a large extent the degree of site disturbance.

Near Infrared Reflectance Spectroscopy (NIRS) analysis

NIRS analysis was conducted on soil samples taken from the Ft. Benning Installation in Phase I in order to determine whether soil sample spectral signatures can be used to discriminate ecological impact, and to determine the relationship between biogeochemistry and spectral reflectance for soil samples. The reflectance signatures of soil samples were analyzed using multivariate statistical methods. Principal Components Analysis was performed to achieve reduction of the dimensionality of data (2000+ variables of wavelengths) into a few important variables. Canonical Discrimination and Discriminant Function Analysis were conducted to determine whether spectral signatures can be used to discriminate soils taken from bottomlands and uplands and also from low, medium and highly disturbed sites. Canonical Correlation and Partial Least Squares were carried out to relate spectral signatures to soil biogeochemistry.

Discrimination on the basis of landscape position using NIRS data was successful using one canonical variable, and results were comparable to Cannonical Discrimination Analysis results found using biogeochemistry data directly. Cannonical Discrimination on the basis on disturbance was not as successful as that obtained using 20 biogeochemical variables, but comparable to that obtained using 4 variables.Results of the Discriminant Function Analysis for landscape position based on the reflectance data are slightly less accurate than those obtained using 18 biogeochemical variables, but provide approximately the same accuracy as those obtained using 4 biogeochemical variables. Results of the Discriminant Function Analysis for disturbance based on reflectance data are slightly less accurate than those obtained using 18 biogeochemical variables, but provide approximately the same accuracy as those obtained using 4 biogeochemical variables.

Vegetation

Landscape evaluation of vegetation characteristics

Structural and compositional parameters of vegetation were measured at the Phase I soil biogeochemical sites. A total of 113 woody and 110 herbaceous species were encountered. Cannonical Correspondence Analysis (CCA) of relative woody plant cover with environmental variables indicates a separation of low disturbance sites from moderate and severe sites, but no marked separation between moderate and severe disturbance sites. Severe disturbance was most closely associated with upland, sandy clay soils. Increased overstory canopy cover as estimated by densiometer measurements were associated with low disturbance sites. These associations have some statistical strength given the significance of the first eigenvalue, however, the lack of a major decrease of the sequential eigenvalues from Axis 1 through Axis 4 indicates a lack of close association among the variables.

Severe disturbance sites were areas of active heavy military equipment training (tanks and Bradley personnel carriers). Within this classification there was a gradient of disturbance from a condition of virtual absence of woody plants to a condition of scattered larger trees (*Pinus palustris, Quercus arkansana, Pinus taeda*) and remnant shrubs and vines that could withstand, or be spread by, repeated vehicular trampling (*Opuntia* sp., *Ipomea* sp., *Vaccinium* sp., *Viburnum rufidulum*, *Crataegus* sp.). There appeared to be a relationship between the cover of a subset of the herbaceous species and sites of severe disturbance.

Response of vegetation ground cover to disturbance associated with clear cutting

The vegetation chronosequence study focused on recovery of ground cover vegetation following soil/vegetation disturbance associated with clear cutting. A total of 47 woody species were encountered in the study plots. Species richness in the 0-3, 8-10, 15-18, and 30-80 year age classes (time since last clear cut) was 36, 31, 37, and 32, respectively. The most abundant and frequent species in all age classes was *Rubus* sp . Indicator analysis identified *Gaylussacia mosieri* (indicator value = 43.7; p=0.028) and *Cary* spp.(indicator value = 31.1; p = 0.072) as the only significant indicators of age class. *Gaylussacia mosieri* occurred most frequently with highest cover in the 30-80 year old age class but also occurred infrequently in younger age classes. Carya sp. were indicators of the 15-18 year age class.

One hundred and fifty eight herbaceous ground cover species were encountered. Species richness for age classes from most recently clear-cut to oldest sites was 80, 61, 79, and 71, respectively. Many species were rare with a total of 57 (approximately 36% of all herbaceous sp.) occurring once in 32 sites, 22 (~ 14%) twice, and 12 (~ 8%) three times). Cover and frequency for herbaceous species differed across age classes (Table 2). After removal of a single outlier (site with low sand content 52%), indicator analysis identified several species representative of each of the 4 age classes. Analysis based on 31 sites with % sand ranging from 67-91% identified *Cyperus croceaus* and *Bulbostylis barbata* as indicators of the 0-3 age class (Table 3). *Andropogon virginicus*, *Dichanthelium* sp., *Sporobolus junceaus* and *Sphagnum sp.* were significant indicators of the 8-10 year age class. *Andropogon virginicus* occurred almost exclusively in the 8-10 year age class. *Pityopsis species* and *Tridens flavus* were indicators of 15-20 yr class. *Andropogon ternarius,, Schizacharium scoparium, Desmodium sp., Hieracium sp, Rhynchosia tomentosa* (marginally significant*)* were indicators of 30-80 yr old sites. *Schizacharium scoparium* and *Andropogon ternarius* are difficult to differentiate in field sampling when floral parts are unavailable. Therefore, values for these two species and those that could not be differentiated as either were summed. Indicator analysis found this complex to be a significant indicator for 30-80 yr age class.

Our results suggest herbaceous species' composition and cover is more indicative of recovery time than woody species. Herbaceous species may be more sensitive than trees and shrubs to local edaphic variation (Drewa et al. 2002), and thus possibly to disturbances that alter soil characteristics. Generally, compared to herbaceous species, woody species are more broadly distributed, animal dispersed, and have underground root systems that facilitate rapid aboveground regrowth and vegetative spread. This allows greater adaptation to disturbance and thus less responsiveness to change (Olson and Platt 1995, Gile et al. 1997). The important environmental gradients shaping herbaceous species composition were age class (8-10 yr. and 0-3 yr.) and bulk density, although species-environment correlation (less than 40%) was lower.

Generally, only a few species stood out as possible indicators of recovery after silvicultural disturbance. Several studies have found successful herbaceous understory indicators of pine tree establishment and growth (Strong et al. 1991, Dibble et al. 1999), although there is the question of whether successful pine growth can be a proxy for overall landscape health. One of the criteria for a successful indicator is for the species to have low variability in response to change in environmental conditions (Dale and Beyeler 2001). Further studies incorporating more sites would help to clarify the validity of these species as indicators.

Watershed Hydrology

Throughfall

During the study period, 140 discrete storm events generated 752.8 mm of precipitation. The events ranged in intensity from 0.3 to 14.4 mm hr^{-1} with an average intensity of 1.8 mm hr^{-1}. Total precipitation accumulation for each event ranged from 0.3 to 73.2 mm with an average of 5.4 mm. Approximately 46% of all storms deposited less than 1 mm. The duration of each event ranged from 0.5 to 34 hours with 50% of all events being one hour or less. The precipitation required to saturate the canopy ranged from 1.14 mm for the wetland plot to 4.00 mm for the pine plantation plot.

The total throughfall measurements ranged from 553.8 mm in the mixed plot to 614.5 mm in the wetland plot. Throughfall plus stemflow accounted for 77.7 to 82.5% of incident precipitation for mature pine and hardwood forests, respectively. Interception losses were largest in the mature pine forest (22.3%) and smallest in the hardwood forest (17.4%). A comparison of average canopy cover and actual interception losses showed that interception losses were very consistent, within 2%, for all forest communities except the mature pine. The pine losses were approximately 5% greater than the other communities. Community specific characteristics were found to control the water input to the soil surface. The relative ratio of throughfall to precipitation was observed to vary seasonally with the largest ratio observed in the fall and winter. The seasonal canopy cover values, as opposed to annual average values, are required to capture the distinct patterns of throughfall corresponding to forest dynamics. In addition, knowledge of canopy cover by landuse is necessary to characterize the increasing throughfall observed for sparser canopies. Variations in tree species and understory composition among forest communities were also identified as having a significant impact on model parameters and subsequent interception prediction.

Infiltration

In wetland/riparian areas, measured K_{sat} values were <5 cm/hr, while in training areas the K_{sat} values ranged from 3 to 36 cm/hr. The range of K_{sat} measured in disturbed areas was lower than that measured in undisturbed areas (6 to 54 cm/hr), but not as low as had been expected since field observations suggest significant soil loss from runoff and erosion events while the undisturbed areas show no evidence of erosion processes. Observed soil losses in the training areas are therefore attributed to a loss of protective vegetative cover (pine/oak forest; understory vegetation; litter layer) and decrease in soil organic matter that otherwise protect against sediment detachment and transport. Thus, during high-intensity rainfall events with wet antecedent soil-water conditions, surface runoff and erosion occur only on the disturbed ridge tops and along roads. Storm hydrograph analyses support this observation.

The mean steady-state infiltration rate of training site soils (12.0 cm hr^{-1}) is less than half that of the non-training sites (26.8 cm hr^{-1}) (Table 4), but is greater than the maximum 100-yr 24-hr rainfall intensity of 10 cm yr^{-1}. Thus, infiltration rate alone does not adequately explain the observed increases in runoff and erosion at highly-impacted training sites. However, the effects of the combined land and soil disturbance features may synergistically increase the runoff and erosion potential. The loss of canopy vegetation and absence of litter and a duff layer intensify the effect of raindrop impact, leading to increased soil detachment. Bare soil surface also promotes surface sealing which, in turn, results in reduced infiltration rates and a more rapid runoff response. The effects of a surface seal would not be manifest in the infiltration measurements conducted in this study since the insertion of the infiltrometer disturbs and crust formation on the soil surface. Consequently, the actual infiltration rates at the training sites may be much lower than measured values.

Streamwater Flow and Quality

Annual-based analysis results for Bonham-1 (reference) and Bonham-2 (impacted) watersheds revealed differences across the range of flow conditions. The average flow conditions revealed that the mean annual flow is higher in the impacted watershed as compared to the reference. The impacted watershed also maintained higher baseflow values. The reference watershed is characterized by higher flow variability for all metrics considered in annual and critical month (December) mean values, storm event timing, and periodic low flows. While the 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 during events. During high flow conditions, the reference watershed crosses a threshold of seven times the median annual daily flow volume for about 2-3 days a year. In the impacted watershed, these floods never occurred. High flood pulses and the flood frequency were higher in the reference watershed.

ANOVA tests indicated that the mean values of storm-based indices, except the time of rise, differ among watersheds 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 indicates that the baseflow and peak discharge in the further confirmed that the mean values of 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 Bonham-1 watershed is characterized by a relatively low baseflow index with significantly higher and more variable peak discharge. During events, 90% of the total events produced greater than bankfull discharge in the reference watershed indicating a highly connected system as compared to 2-47% in impacted watersheds. The storm flows consistently lasted longer and responded faster to rain events in the reference watershed as compared to the impacted ones. Once the stream responded, the time of rise 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 as compared to impacted watersheds.

7 key storm-based hydrologic indices are significantly related to military land management. Increased military land extent, bare land, and 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. Result also shows that the 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, however, identified for forestry management practices. Based on stepwise multiple correlations that characterized the response of storm-based indices to military impacts, the three management variables that impacted storm responses were military land extent, road density, and the number of road crossing streams. The greatest impact of land management was found for indices corresponding to storm duration (time base, response 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. Stream pH, temperature, and conductivity were positively correlated with total length of all roads within the watershed. Stream TKN was correlated negatively with disturbance index. TP was negatively correlated with road density whereas positively correlated with number of road crossing streams. Chloride showed a positive correlation with the total length of all roads within the watershed. TOC was negatively correlated with military land and a disturbance index based on percentage bare land on slopes greater than 3%. This study identified strong relationships among selected watershed physical characteristics that are more susceptible to human induced disturbances and water quality parameters. Regression results suggested that 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. lag, and time of rise).

Water quality parameters in the study watersheds varied over the sampling period and among watersheds. 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). 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. 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. TKN, TP, and Cl 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.

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. Stream pH, temperature, and conductivity were positively correlated with total length of all roads within the watershed. Stream TKN was correlated negatively with disturbance index. TP was negatively correlated with road density whereas positively correlated with number of road crossing streams. Chloride showed a positive correlation with the total length of all roads within the watershed. TOC was negatively correlated with military land and a disturbance index based on percentage bare land on slopes greater than 3%. This study identified strong relationships among selected watershed physical characteristics that are more susceptible to human induced disturbances and water quality parameters.

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. Watersheds with more roads crossing streams tended to produce more TP. 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. TSS variability, on the other hand, as may be expected was captured by the percent of bare land within a watershed. In summary, the water quality results suggested 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.

Soil-water content and storage dynamics

Bimonthly measurement of distributed soil moisture content in the Bonham Creek sub-watershed indicated relatively dry upland soils with increasing water content on the hill slopes (Fig. 8). The majority of the water storage is confined to the areas immediately adjacent to the stream channel. Our hypothesis that water content is spatially dependent on landscape features such as slope and elevation was not confirmed by statistical analysis. However, when compared to volumes estimated from precipitation and hydrograph data, our estimated soil-water storage appear to account for the expected volume of precipitation minus hydrograph volume. A one-year cycle of soil-water storage monitoring provided a temporal pattern of total soil-water storage in the watershed (Fig. 9). Total volume of soil-water storage measured during drier months (June-August) was consistently lower than the levels observed during wetter months (September – May).

Comparative sampling of upland soils in areas that have experienced severe soil and vegetation degradation from tank training and other military exercises revealed higher soil bulk density and lower porosity than in low-impact or non-training areas. This implies a rearrangement of finer soil particles, resulting in a loss of larger pore-size fractions and decreased saturated hydraulic conductivity. Consequently, upland soils of this severely-impacted watershed possess a much greater runoff potential, which increases the probability of continued erosion.

Similar-media scaling approaches were implemented as a compact way to assess the differences between the hydraulic properties of soil cores taken at training and non-training sites. Scaling factors of the training site cores were clustered between 0.70 and 0.90, while the scaling factors of the non-training site cores were more uniformly distributed between 0.50 and 2.0. The mean value for scaling factors at non-training sites (1.11) was significantly greater than the mean for training sites (0.83), and the variance of scaling factors at non-training sites was three times higher than for training sites (Table 4). While the scaling factors themselves weakly correlated with landscape features such as slope or elevation, the scaling factors reveal a greater variance (more dispersed distribution) for the soil cores taken at the training versus non-training sites. Geostatistical analyses also show that scaling factors for training sites have less spatial correlation than those of the non-training sites.

Additional distinctions found for the cores taken at the training sites included larger bulk densities from soil compaction (Table 4), and lower soil organic carbon content (from loss of surface soil and vegetative cover) as indicated by spectral response. Localized training events denude hilltops and mix soil surface and subsurface horizons, leaving a highly variable and fragmented pattern in soil hydrologic properties, which, in turn, contributes to increased erosion potential. Because soil erosion impacts are event driven (i.e., occur at a specific point in time), an impending disturbance level at the training location cannot be easily appraised with the indicator concept. However, ecological indicators may be a viable approach for monitoring the spread of ridge-top disturbance to lower physiographic positions in the watersheds, such as the impacts of sediment deposition in the riparian wetlands and in-stream processes.

SUMMARY

Military impact at Fort Benning is patchy; severe disturbance is concentrated in a few areas of the base, primarily along sandy ridge tops. Many areas experience minimal disturbance from military training, since they serve as buffer zones or are used only to accommodate training of infantry foot soldiers. Severe impacts to soil, vegetation and hydrologic processes are associated with mechanized training involving tracked (tanks and Bradley) vehicles. Commonly observed impacts of mechanized training on soil and vegetation included (1) disturbance or destruction of vegetation communities, including ground cover, understory and canopy vegetation, (2) disruption of soil A horizon and effective burial or dilution of biologically-active topsoil with organic-poor lower horizons, (3) compaction of subsoil, reducing soil permeability and increasing runoff and erosion potential, (4) loss of A and E horizons in severely-impacted upland areas, rendering soil unsuitable for supporting native plant communities, (5) gulley erosion in downslope areas, with significant sedimentation in wetlands and streams, and (6) short-circuiting of watershed flow paths with increased surface runoff and decreased subsurface detention in uplands, creating hydrologic and ecological imbalances in wetlands and streams.

Moderate to severe impacts also occur in several areas of non-military land use, primarily due to forest clear-cutting activities. In addition to canopy removal and severe reduction of understory vegetation and ground cover, clear cutting and related activities (e.g., creation of access roads) in this region can cause significant soil disturbance, compaction, and erosion. Hydrologic and ecological impacts observed in wetlands and streams downslope from clear-cut upland areas were similar in nature to those observed in association with severe military disturbance.

Not surprisingly, the soil, vegetation and hydrologic parameters (potential indicators) that were most closely correlated with pre-determined site disturbance levels (low, moderate, severe) were those that reflected loss of vegetation biomass and community structure, disruption and/or compaction of soil, and loss of soil A horizon (and soil organic matter) in uplands; and accelerated sedimentation of clay and sand in wetlands. The most promising soil biogeochemical indicators for upland areas were directly related to soil organic matter content and quality (biodegradability): total C, microbial biomass, beta-glucosidase activity, soil respiration, and ratio of microbial respiration to biomass.

Vegetative indicators that most accurately reflected the impacts of military training were relatively straightforward, i.e., percent cover of herbaceous vegetation (ground cover), or in cases of more severe impacts, mid-story or canopy cover. In general, discernable reduction in plant cover or biomass occurred only in association with severe site impacts, where the use of ecological indices or other metrics is less critical. However, studies of sites in various stages of recovery from severe disturbance yielded a list of indicator plant species that may be useful in tracking the progress of restoration efforts in highly-impacted areas. It should be noted that use of indicators related to vegetation community composition in moderately or less impacted sites was often confounded by residual effects of prior soil disturbance related to agricultural land uses. Thus, reliance on indicator species to assess ecological condition may require a re-evaluation of “natural” or reference conditions prior to their use.

In wetland areas downslope from impacted uplands, relationships between soil biogeochemical indicators and upland impacts were less clearly defined. However, indicators that directly related to wetland soil organic matter content (and “dilution” by clay or sand) were useful in identifying sediment-impacted wetlands located below severely-disturbed upland areas. The potential value of wetland soil biogeochemical properties as indicators of nutrient loading in uplands (e.g., from excessive fertilization or waste disposal) was not realized at the Fort Benning study areas, due to the nature of the ecological impacts in upland areas.

Stream TOC and TKN concentration decreased with increasing soil and vegetation disturbance (proportion of bare ground) in the watershed, reflecting (1) depletion of soil organic matter and detritus (in the form of forest litter and duff layers) in uplands and (2) reduced leaching in soils due to short-circuited flow paths (gulleys) from uplands to streams. Hydrologic indicators are, potentially, of significant value for analysis of disturbance or recovery on a watershed scale. Of primary utility would be the analysis of hydrographs, which clearly reflect hydrologic imbalances resulting from soil and vegetation disturbance in uplands. On a smaller scale, soil physical parameters, such as bulk density, porosity, texture (grain-size distribution) and possibly saturated hydraulic conductivity are potentially useful indicators of ecological condition at specified sites.

CONCLUSIONS

General Conclusions:

13. Approximately 2-15% of throughfall shows up as stream flow. Median value is approximately 6%. Time to peak discharge is approximately 3 hours.
14. Storm intensities are usually $<K_{sat}$ at most places, except severely disturbed areas.
15. Soil cover plays an important role in determining the potential runoff and may be more important than K_{sat} of surface soil.
16. Biogeochemical cycling in soils and vegetation are influenced by soil-water content.
17. Soil organic matter and its cycling is an important biogeochemical indicator.
18. Spectral analysis shows excellent promise to determine soil nutrient status.
19. Understory vegetation species composition correlates with disturbance. Clear indicators generally observed only at heavily impacted sites.
20. Nutrient and sediment loads in “low” and “medium” impact sites are not too large. Sediment may be the most important water quality attribute for “severe” impact sites.
21. Water quality measurements revealed low levels of most nutrients.
22. Decreased canopy cover in wetlands and hardwood communities of impacted areas increase the nutrient load to streams.
23. Riparian zones play an important role in determining water quality.
24. Multivariate Analysis, Principal Component Analysis, and Canonical Correspondence Analysis yielded combinations of factors that are useful in identifying impacts.

REFERENCES:

American Public Health Association (APHA). 1992. Standard methods for the examination of water and wastewater. 18th edn. American Public Health Association, Washington D.C.

Anderson, T.-H., and K. H. Domsch. 1993. The metabolic quotient for CO_2/qCO_2 as a specific activity parameter to assess the effects of environmental conditions, such as pH, on the microbial biomass of forest soils. Soil Biol. Biochem. 25:393-395.

Berryman, D., B. Bobee , D.Cluis, and J. Haemmerlin. 1988. Nonparametric tests for trend detection in water quality time series. Water Resources Bull. 24(3):545-556.

Blackburn, W.H., F.B. Pierson, C.L. Hanson, T.L. Thurow, and A.L. Hanson. 1992. The spatial and temporal influence of vegetation on surface soil factors in semiarid rangelands. Transactions of the ASAE 35:479-486.

Bonham, C. D. 1989. Measurements for terrestrial plants. John Wiley & Sons. New York. 338 p.

Bundy, L.G. and J.J. Meisinger 1994. Nitrogen availability indices. *In* Methods of Soil Analysis Part 2. Microbiological and Biochemical Properties. R.W. Weaver et al. (eds.) Soil Sci. Soc. America. Book Series No. 5. pp. 951-984.

Burgess, T.M., and R. Webster. 1980a. Optimal interpolation and isarithmic mapping of soil properties. I. The semi-variogram and punctual kriging. J. Soil Sci. 31:315-331.

Burgess, T.M., and R. Webster. 1980b. Optimal interpolation and isarithmic mapping of soil properties. II. Block kriging. J. Soil Sci. 31:333-341.

Clewell, A. F. 1989. Natural history of wiregrass (Aristida stricta Michx graminae). Natural Areas Journal 9:223-233.

Christensen, N.L. 1977. Fire and soil-plant nutrient relations in a pine-wiregrass savanna on the Coastal Plain of North Carolina. Oecologia 27:27-44.

Christensen, N.L. 1993. The effects of fire on nutrient cycles in longleaf pine ecosystems. Pages 205-214 in S.M. Hermann, ed. Proc. of the Tall Timbers Fire Ecology Conference, No. 18, The Longleaf Pine Ecosystem: Ecology, Restoration, and Management. Tall Timbers Research Station, Tallahassee, FL.

D'Angelo, E. M. and K. R. Reddy. 1994b. Diagenesis of organic matter in a wetland receiving hypereutrophic lake water. II. Role of inorganic electron acceptors in nutrient release. J. Environ. Qual. 23:928-936.

D’Angelo, E.M. and K.R. Reddy. 1998. Influence of electron acceptors on biogeochemical properties of wetland soils: Microbial activity and organic carbon mineralization. Soil Biol. Biochem. (In review).

DeBusk, W.F., K.R. Reddy, M.S. Koch and Y. Wang. 1994. Spatial distribution of soil nutrients in the northern Everglades. Soil Sci. Soc. Am. J. 58:543-552.

DeBusk W.F. and K.R. Reddy. 1998. Turnover of detrital organic carbon in a nutrient-impacted Everglades marsh. Soil Sci. Soc. Am J. 62:1460-1468.

Dick, R.P. 1994. Soil enzyme activities as indicators of soil quality. *In* Defining Soil Quality for a sustainable environment SSSA Special publ. No. 35. J.W. Doran, D.C. Coleman, D.F. Bezdick, and B.A. Stewart (eds.) Soil Sci. Soc. Am. Madison, Wis. pp. 107-124.

Dingman, S.L., 1994. Physical Hydrology. New Jersey: Prentice Hall, 1994.

Doran, J.W., and T.B. Parkin. 1994. Defining and assessing soil quality. p. 3-21. In J. W. Doran et al. (ed.) Defining soil quality for a sustainable environment. SSSA Spec. Publ. 35. SSSA and ASA, Madison, WI.

Elliot, E.T., D.C. Coleman, R.E. Engham, and J.A. Trofymow. 1984. Carbon and energy flow through microflora and microfauna in the soil subsystem of terrestrial ecosystems. p. 424-

433. in Klug, M.J., and C.A. Reddy (ed.) Current perspectives in microbial ecology. American Society of Microbiology, Washington.
Findlay, R.H., M.B. Trexler, J.B. Guckert, D.C. White. 1990. Laboratory study of disturbance in marine sediments: response of a microbial community. Marine Ecology Progress Series. 62:121-133.
Frostegard, A., E. Baath and A. Tunlid. 1993. Shifts in the structure of soil microbial communities in limed forests as revealed by phospholipid fatty acid analysis. Soil Biol. Biochem. 25:723-730.
Goovaerts, P. 1997. Geostatistics for natural resources evaluation. Oxford University Press, New York, NY.
Graham, W.D. and C.D. Tankersley. 1993. Forecasting Piezometric Head Levels in the Floridan Aquifer: A Kalman Filtering Approach. Water Resources Research. 29(11):3791-3800.
Grover, N.C., and Harrington, A.W. 1966. Stream Flow: Measurements, Records and their uses. New York: Dover.
Hamlett, J.M., R. Horton, and N.A.C. Cressie. 1986. Resistant and exploratory techniques for use in semivariogram analyses. Soil Sci. Soc. Am. J. 50:868-875.
Hardesty, J.L., D.R. Gordon, K. Poiani, and L. Provencher. 1997. Monitoring ecological condition in a northwest Florida sandhill matrix ecosystem. Final report to Natural Resources Branch, Eglin Air Force Base, Florida. The Nature Conservancy, Gainesville, FL.
Hart, S.C., J.M. Stark, E.A. Davidson and M.K. Firestone. 1994. Nitrogen mineralization, immobilization and nitrification. *In* Methods of Soil Analysis Part 2. Microbiological and Biochemical Properties. R.W. Weaver et al. (eds.) Soil Sci. Soc. America. Book Series No. 5. pp. 985-1018.
Helvey, J.D., and J.H. Patric. 1965. Design criteria for interception studies. Wallingford, Oxon., U.K.: International Association of Hydrologic Sciences, Publication No. 67:131-137.
Hester, J.W., T.L. Thurow, and C.A. Taylor, Jr. 1997. Hydrologic characteristics of vegetation types as affected by prescribed burning. J. Range Manage. 50:199-204.
Horwarth, W.R. and E. A. Paul. 1994. Microbial biomass. *In* Methods of Soil Analysis Part 2. Microbiological and Biochemical Properties. R.W. Weaver et al. (eds.) Soil Sci. Soc. America. Book Series No. 5. pp. 753-774.
Isaaks, E.H., and R.M. Srivastava. 1989. An introduction to applied geostatistics. Oxford University Press, New York, NY.
Journel, A.G., and C. Huijbregts. 1978. Mining geostatistics. Academic Press, New York.
Klute, A. 1986. Methods of Soil Analysis. Part I. Physical and mineralogical methods. 2nd ed. Soil Sci. Soc. Amer. Madison, WIS.
Kowalchuk GA, Stephen JR, De Boer W, Prosser JI, Embley TM, Woldendorp JW. 1997. Analysis of ammonia-oxidizing bacteria of the beta subdivision of the class Proteobacteria in coastal sand dunes by DGGE and sequencing of PCR-amplified 16S rDNA fragments. Appl. Environ. Microbiol. 63:1489-1497.
Lemon, p.c. 1949. Successional responses to herbs in the longleaf-slash pine forest after a fire. Ecology 30:135-145.
Lodge, D. J., W. H. McDowell, and C. P. McSwiney. 1994. The importance of nutrient pulses in tropical forests. Tree 9:384-387.
Marx, D. and K. Thompson. 1987. Practical aspects of agricultural kriging. Univ. of Arkansas, Bulletin 903.

Means, D.B. and G.Grow. 1985. The endangered longleaf pine community. ENFO 85:1-12.
Melillo, J.M., R.J. Naiman, J.D. Aber, and A.E. Linkins. 1984. Factors controlling mass loss and nitrogen dynamics of plant litter decaying in northern streams. Bull. Mar. Sci. 35:341-356.
Myers, R.L. 1990. Scrub and high pine. Pp. 150-193 in R.L. Myers and J.J. Ewel, eds. Ecosystems of Florida. University of Central Florida Press, Orlando, FL.
Muyzer, G., E. de Waal, and A Uitterlinden. 1993. Profiling of complex microbial populations by DGGE of PCR amplified genes coding for 16S rRNA. Appl. Environ. Microbiol. 59:695-700.
Newman, S., and K.R. Reddy. 1993. Alkaline phosphatase activity in the sediment-water column of a hypereutrophic lake. J. Environ. Qual. 22:832-838.
Newman, S., K.R. Reddy, W.F. DeBusk and Y. Wang. 1997. Spatial distribution of soil nutrient in a northern Everglades marsh: Water Conservation Area 1. 61:1275-1283.
Ohtonen, R. 1994. Accumulation of organic matter along a pollution gradient: application of Odum's theory of ecosystem energetics. Microb. Ecol. 27:43-55.
Platt, W.J., G.W. Evans, and M.M. Davis. 1988. Effects of fire season on flowering of forbs and shrubs in longleaf pine forests. Oecologia 76:353-363.
Pitman, J.I., 1989. Rainfall interception by bracken litter-relationship between biomass, storage, and drainage rate. Journal of Hydrology 111:281-291.
Provencher, L., D.R.Gordon, K.E.M. Galley, J.L. Hardesty, H.L. Rodgers, J. Sheehan, E.Levine, G.W. Tanner, L.A. Brennan, and K.W. Blandford. 1996. Pre-restoration analysis of plants, invertebrates, and birds in sandhill systems at Eglin Air Force Base, Florida. Annual Report to Natural Resources Division, Eglin Air Force Base, Niceville, FL. Public Lands Program, The Nature Conservancy, Gainesville, FL.
Reddy, K.R., O.A. Diaz, L.J. Scinto, and M. Agami. 1995. Phosphorus dynamics in selected wetlands and streams of the Lake Okeechobee Basin. Ecol. Eng. 5:183-207.
Reddy, K. R., J. R. White, A. Wright, and T. Chua. 1998. Influence of phosphorus loading on microbial processes in soil and water column of wetlands. In Phosphorus Biogeochemistry in Subtropical Ecosystems: Florida as a case example. K. R. Reddy, G. A. O'Connor, and C.L. Schelske (eds). CRC/Lewis Publ. (in press).
Rouhani, S., and H. Wackernagel. 1990. Multivariate geostatistocal approach to space-time data analysis. Water Resources Research. 26(4):585-592.
Stephen, J.R., Mccaig, A.E., Smith, Z., Prosser, J.I. and Embley, T.M.1996. Molecular diversity of soil and marine 16s rRNA gene sequences related to -subgroup ammonia-oxidizing bacteria. Appl. Environ.Microbiol. 62:4147-4154.
Swift, M.J. 1982. Microbial succession during the decomposition of organic matter. p. 164-177. In Burns, R.G. and J.H. Slater (ed.) Experimental Microbial Ecology. Blackwell Scientific, Oxford.
Tabatabai, M.A. 1994. Soil enzymes. *In* Methods of Soil Analysis Part 2. Microbiological and Biochemical Properties. R.W. Weaver et al. (eds.) Soil Sci. Soc. America. Book Series No. 5. pp. 775-834.
Tiedje, J.M. 1982. Denitrification. p. 1011-1026. *In* A.L. Page, R.H. Miller and D.R. Keeney (eds.) Methods of Soil Analysis, Part 2. Chemical and Microbiological Properties (2^{nd} ed.)
Torstensson, L., M. Pell, and B. Stenberg. 1998. Need of a strategy for evaluation of arable soil quality. Ambio 27:4-8.

Trangmar, B.B., R.S. Yost, and G. Uehara. 1985. Application of geostatistics to spatial studies of soil properties. Adv. in Agronomy 38:45-94. Academic Press, London.

USDA, 1997. Soil survey of Chattahoochee and Marion counties, Georgia. Natural Resources Conservation Service, U.S. Department of Agriculture.

U.S. Environmental Protection Agency. 1997. Environmental monitoring and assessment program (EMAP) research strategy. U.S. Environmental Protection Agency, Office of Research and Development.

Van Veen, J.A., J.N. Ladd, and M.J. Frissel. 1984. Modeling C and N turnover through the microbial biomass in soil. Plant and Soil 76: 257-274.

Wardle, D. A. 1993. Changes in the microbial biomass and metabolic quotient during leaf litter succession in some New Zealand forest and scrubland ecosystems. Functional Ecology 7:346-355.

Table 1. Summary comparison (mean, 10th percentile and 90th percentile values) of soil chemical properties among bottomland, mid-slope, and upland/ridge sites in the Fort Benning study area.

	Bottomland (n=98)			**Mid-Slope (n=101)**			**Upland/Ridge (n=101)**		
	Mean	**10th pct**	**90th pct**	**Mean**	**10th pct**	**90th pct**	**Mean**	**10th pct**	**90th pct**
pH	4.99	4.19	5.85	5.30	4.71	5.85	5.26	4.76	5.71
Total C (g kg^{-1})	38.3	7.0	100.6	11.8	6.5	18.7	10.0	3.5	18.5
DOC (mg kg^{-1})	121.3	21.3	272.7	70.8	20.2	135.3	60.1	15.8	151.3
MBC (mg kg^{-1})	561.2	122.0	1435.0	232.3	79.0	425.2	205.2	64.9	372.7
Total N (g kg^{-1})	2.03	0.33	4.92	0.45	0.23	0.75	0.39	0.17	0.67
Exchange. NH4-N (mg kg^{-1})	10.70	2.00	22.40	4.30	1.20	7.70	5.00	1.10	9.60
Total P (mg kg^{-1})	200.8	59.5	435.5	80.3	42.9	124.8	79.7	39.3	137.8
Mehlich-P (mg kg^{-1})	1.32	0.35	1.81	1.20	0.54	1.93	2.09	0.48	3.60
Mehlich-Fe (mg kg^{-1})	306.0	35.1	722.6	40.0	17.7	64.1	34.1	13.3	57.6
Mehlich-Al (mg kg^{-1})	603.1	94.6	1444.7	258.0	108.1	414.1	233.3	113.0	354.8
Mehlich-Ca (mg kg^{-1})	232.3	21.1	534.7	220.5	15.7	666.3	166.7	11.6	626.4
Mehlich-Mg (mg kg^{-1})	66.4	11.8	185.4	79.5	2.6	268.7	64.6	2.5	301.7

TABLE 2: Mean aerial cover (%) and frequency (%) (based on 80 subplots per age class) for herbaceous species in four age classes indicating years post clear cut.

Herbaceous Species (Code)	**0-3**		**8-10**		**15-20**		**30-80**	
	Cover	Freq.	Cover	Freq.	Cover	Freq.	Cover	Freq.
Acalypha gracilens	0.5	12	1.3	26	0.4	14	1.3	23
Acanthaceae fam	0	0	0	0	0	0	t	1
Agalinis setacea	0.3	2	t*	1	0	0	0.2	2
Agrimonia microcarpa	0	0	0	0	0	0	0.2	1
Andropogon gerardii	0.1	2	t	1	t	1	0	0
Andropogon gyrans	0	0	0	0	0	0	0.2	1
Andropogon virginicus	0.2	1	7.0	27	0	0	1.0	4
Apiaceae fam	0.1	2	t	2	0.2	7	0.3	3
Aristida purpurascens	0.1	1	t	1	0	0	0	0
Aristida sp.	0.7	6	5.3	39	1.4	14	2.2	27
Arundinaria gigantea	0	0	0	0	0.2	2	0	0
Aster concolor	0.4	5	0.5	4	t	1	0.3	4
Aster dumosus	1.9	14	0.3	14	1.75	20	2.3	23
Aster linariifolius	0	0	0	0	0	0	0.3	2
Aster patens	0	0	0	0	0.2	3	0.1	1
Aster paternus	0	0	0.1	1	0.2	2	0	0
Aster solidagineus	0.2	3	0	0	0	0	0.1	2
Aster sp.	t	1	0.1	3	0.1	3	0.5	9
Aster tortifolius	0.4	4	0.3	7	0.2	6	0.6	10
Bulbostylis barbata	1.2	7	t	1	0	0	0	0
Centrosema virginianum	0.2	2	0	0	0.3	5	0.3	5
Cercis Canadensis	0	0	0	0	t	1	0	0
Chamaecrista fasciculata	0.6	13	0.2	6	1.2	21	1.1	17
Chasmanthium laxum var. sessiliflorum	0.6	6	0	0	0.2	2	0	0
Chrysopsis mariana	0	0	0	0	t	1	0.1	1
Cirsium sp.	0	0	0	0	0	0	0.1	1
Conyza Canadensis	1.2	21	0.9	10	0.2	3	0	0
Coreopsis sp.	1.0	23	3.5	42	2.7	35	3	30
Crotalaria rotundifolia	0	0	0.2	3	0	0	0	0
Crotonopsis linearis	0	0	t	1	0	0	0	0
Cyperus croceus	0.4	11	0	0	t	1	t	2
Dalea sp.	0	0	0	0	t	1	0	0
Desmodium rotundifolium	0	0	0	0	t	3	0.1	1
Desmodium sp.	0.4	6	0	0	0.4	4	1.2	10
Dichanthelium sp.	7.7	65	12.7	74	5.8	59	4.9	46
Digitaria cognata	0.1	2	0.1	3	t	1	0	0
Digitaria filiformis var. filiformis	1.1	4	0.5	5	0	0	0	0
Diodia teres	0.8	3	0	0	0	0	0	0
Elephantopus elatus	0.2	1	0	0	0	0	0	0
Eragrostis hirsute	0.1	3	0.6	10	0.9	6	0.1	2
Eupatorium aromaticum	0.1	2	0	0	0.3	3	0.2	4
Eupatorium capillifolium	3.7	24	3.9	28	2.2	17	0.7	7
Eupatorium mohrii	0.3	1	0	0	0	0	0	0
Eupatorium rotundifolium	t	1	0	0	0	0	t	1
Euthamia caroliniana	0	0	0	0	t	1	0.9	4
Fabaceae fam	0.5	4	0	0	0.8	16	0.5	5

Florichia floridana	0.1	1	0	0	0	0	0	0
Galactia microphylla	0.1	2	0	0	0	0	0	0
Galactia sp.	0.2	5	0	0	t	2	t	1
Galium pilosum	1.1	6	0.2	5	0.2	6	0.1	3
Gnaphalium obtusifolium	t	1	0	0	0	0	0	0
Gnaphalium sp.	0.2	7	0.2	9	0.1	5	0.2	5
Gymnopogon ambiguus	0.5	8	0.2	6	0.5	7	0.9	7
Haplopappus divaricatus	0	0	0	0	0.1	3	0.1	1
Hedyotis procumbens	0	0	0	0	t	1	t	1
Helianthemum corymbosum	t	1	0	0	0	0	t	1
Helianthus floridanus	0	0	0.1	1	0	0	0	0
Heterotheca subaxillaris	t	1	0	0	0	0	0	0
Hieracium sp.	0.1	2	0	0	0	0	0.1	5
Hypericum gentianoides	0	0	t	1	0	0	0	0
Ipomoea sp.	t	1	0	0	0	0	0	0
Juncus dichotomus	0	0	0	0	t	1	0	0
Kummerowia striata	0	0	t	2	T	1	0.1	1
Lechea minor	0.1	3	0.2	3	0	0	0.3	4
Lechea mucronata	0	0	0.1	2	0	0	0	0
Lechea sp.	0.4	7	0.3	8	T	1	0.1	3
Lespedeza hirta	0.1	1	t	2	0.1	1	0	0
Lespedeza stuevei	2.3	22	1.0	9	0.7	7	1.8	23
Liatris elegans	0.2	5	0.1	2	0.1	1	t	1
Liatris tenuifolia	0.4	4	0	0	0	0	0.1	2
Liatrus sp.	t	1	0	0	0	0	0	0
Lobelia puberula	0	0	0	0	t	2	0	0
Ludwigia sp.	0	0	0	0	t	1	0	0
Mollugo verticillata	0.1	2	0	0	0	0	0	0
Opuntia humifusa	0	0	0.1	2	0.3	6	0	0
Oxalis corniculata	0.1	3	0.1	4	0	0	0	0
Panicum anceps	0	0	0	0	t	1	0	0
Panicum rigidulum	0	0	0	0	0.2	3	0	0
Panicum verrucosum	0	0	0	0	0.2	3	0	0
Panicum virgatum	0	0	t	1	0.1	1	0.1	2
Paspalum notatum	1.1	8	0.4	8	0	0	0.1	2
Paspalum setaceum	0	0	t	1	0	0	0	0
Phlox nivalis	0.3	1	0	0	0.2	1	0	0
Piriqueta caroliniana	0	0	0.1	2	0	0	t	1
Pityopsis sp.	4.4	29	4.6	39	13.2	52	7.5	44
Poaceae fam	0	0	t	1	0	0	2.0	16
Polygala grandiflora	t	1	0	0	0	0	0	0
Polypremum procumbens	0.8	10	0.4	5	0.1	1	0.1	2
Pteridium aquilinum	1.1	8	0	0	0.5	4	0.4	3
Rhexia mariana	0	0	0	0	0.1	3	0	0
Rhus copallinum	0	0	t	1	0	0	0	0
Rhynchosia reniformis	0.4	4	t	1	0.1	2	t	1
Rhynchosia tomentosa	0.3	4	0	0	0.2	4	0.5	13
Rudbeckia fulgida	0	0	0	0	0.1	2	0	0
Ruellia caroliniensis	0	0	0	0	0	0	t	1
Saccharum alopecuroides	0	0	0	0	0	0	0.3	2

Andropogon ternariues/Schizacharium scopairium	0.5	7	0.7	10	0.6	5	3.8	29
Schizacharium scoparium	4.0	24	3.0	27	7.5	36	14.0	57
Andropogon ternarius	1.0	5	0.7	5	0.8	7	2.7	30
Scleria sp.	1.0	12	0.1	5	0.2	4	1.3	10
Seymeria pectinata	0	0	0	0	0.1	1	0	0
Silphium compositum	0	0	0	0	0.3	2	0.1	1
Solidago fistulosa	0	0	0	0	0.1	2	0.4	1
Solidago latissimifolia	0	0	0	0	0	0	0.3	2
Solidago nemoralis	2.9	24	0.4	12	1.9	22	3.0	30
Solidago odora	0	0	0	0	0.6	4	0	0
Solidago sp.	0	0	0	0	0.1	2	0	0
Sorghastrum secundum	0	0	0.5	6	0.3	2	0	0
Sphagnum sp.	0	0	1.6	24	0.1	5	0.1	2
Sporobolus junceus	0	0	1.5	9	0	0	0	0
Stylisma patens	t	1	0.1	2	0	0	0	0
Stylodon carneum	0	0	0	0	0.2	1	0.1	1
Tephrosia florida	0.2	2	0	0	0	0	0	0
Tephrosia sp.	0	0	0	0	t	1	0	0
Tephrosia virginiana	t	1	0	0	0	0	0	0
Tragia urens	t	1	t	1	0	0	t	1
Trichostema dichotomum	0	0	0.5	3	0	0	0	0
Trichostema setaceum	0.3	1	0.3	4	0.1	5	0	0
Tridens carolinianus	0	0	t	1	0	0	0	0
Tridens flavus	0.2	6	0	0	1.0	11	0	0
16 Unkown herbaceous	t-1.0	t-2	t	0	t	t-1	t	t
Urtica sp.	0.1	3	0	0	T	1	0	0
Vicia sp.	t	1	0	0	0	0	0	0
Viola palmate var. triloba	t	1	0	0	0	0	0	0
Viola primulifolia	0	0	t	1	T	1	0	0
Wahlenbergia marginata	0	0	0.3	1	0	0	t	1

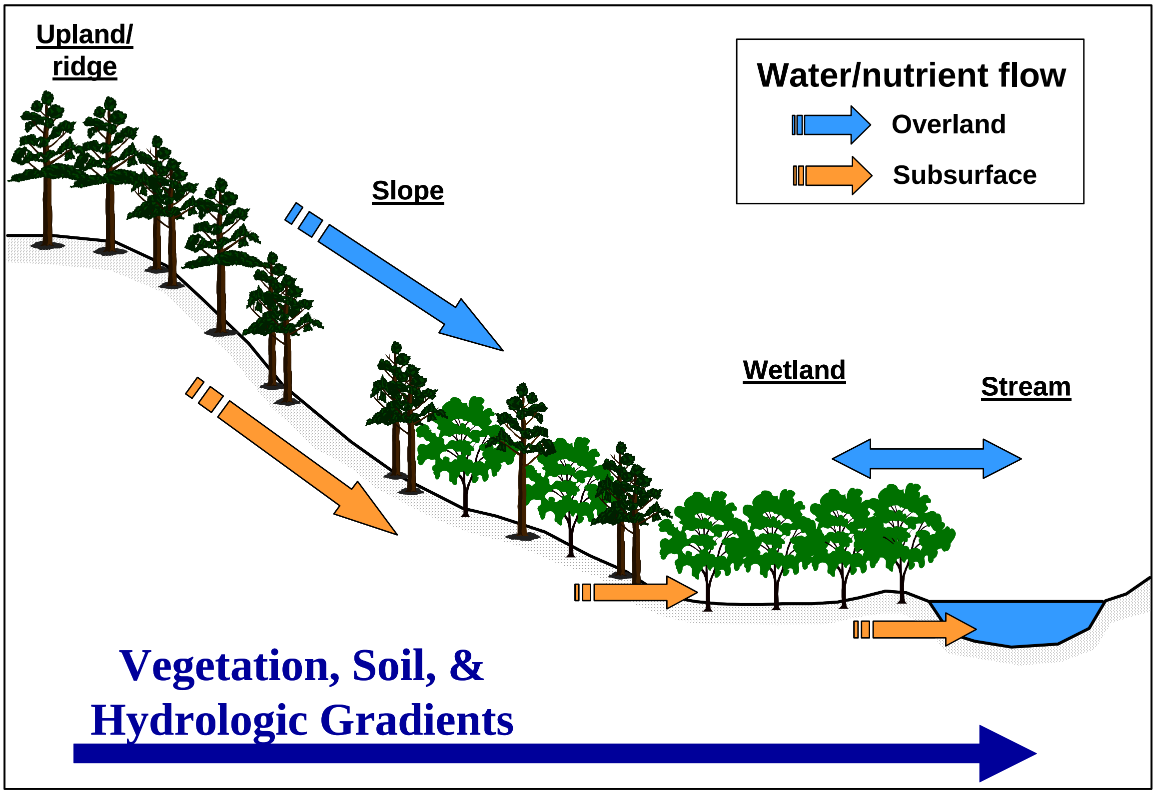
* =trace = < 0.1

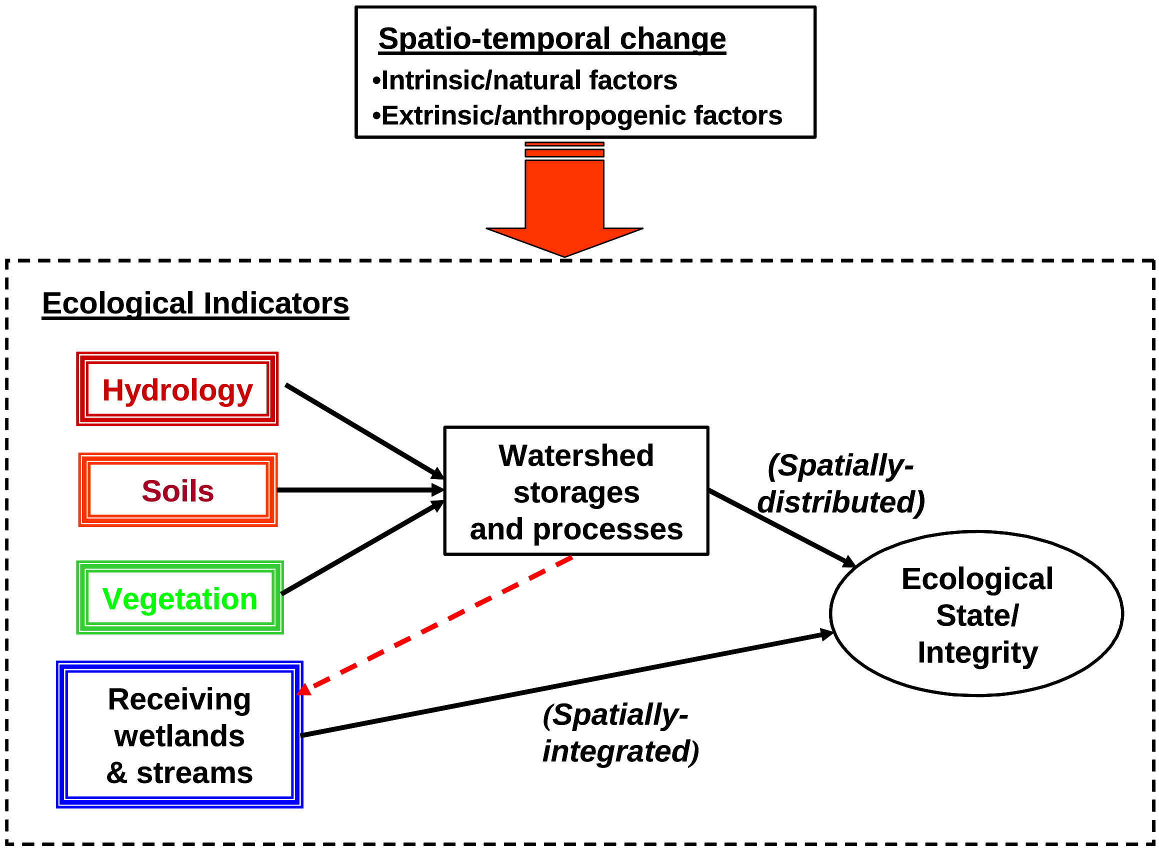
Table 3. Post clear cut age class, indicator value and significance for species identified as indicators.

Species	Post Clear Cut Age Class (years)	Indicator Value	p-value
Bulbostylus barbata	0-3	36.2	0.063
Cyperus croceus	0-3	43.7	0.034
Andropogon virginicus	8-10	62.1	0.002
Dichanthelium species	8-10	41.1	0.012
Sphagnum species	8-10	50.3	0.008
Sporobolus junceus	8-10	28.6	0.044
Pityopsis species	15-20	44.5	0.020
Tridens flavus	15-20	40.2	0.048
Desmodium species	15-20	39.8	0.034
Andropogon ternarius	30-80	36.3	0.028
Schizacharium scoparium	30-80	49.0	0.011
Schizacharium/Andropogon ternarius Complex	30-80	47.1	0.019
Hieracium species	30-80	36.7	0.024
Rhynchosia tomentosa	30-80	32.9	0.088

Table 4. Summary statistics for soil scaling factors, bulk density and infiltration rate for training and non-training areas in the Ft. Benning study area.

1.	**2. Scaling Factor**		**3. Bulk Density (g cm^{-3})**		**4. Infiltration Rate (cm h^{-1})**	
5. Statistic	**6. Training**	**7. Non-training**	**8. Training**	**9. Non-training**	**10. Training**	**11. Non-training**
12. Mean	13. 0.83	14. 1.11	15. 1.54	16. 1.38	17. 12.0	18. 26.8
19. Min	20. 0.50	21. 0.42	22. 1.40	23. 1.07	24. 1.1	25. 1.2
26. Max	27. 1.44	28. 1.95	29. 1.75	30. 1.6	31. 35.8	32. 53.0
33. Variance	34. 0.05	35. 0.15	36. 0.01	37. 0.02	38. 86.5	39. 154


Upland/
ridge
Water/nutrient flow
Overland
Subsurface
Slope
Wetland
Stream
Vegetation, Soil, &
Hydrologic Gradients
Figure 1



Spatio-temporal change
•Intrinsic/natural factors
•Extrinsic/anthropogenic factors
Ecological Indicators
Hydrology
Soils
Vegetation
Watershed storages and processes
(Spatially-distributed)
Ecological State/ Integrity
Receiving wetlands & streams
(Spatially-integrated)

Figure 2

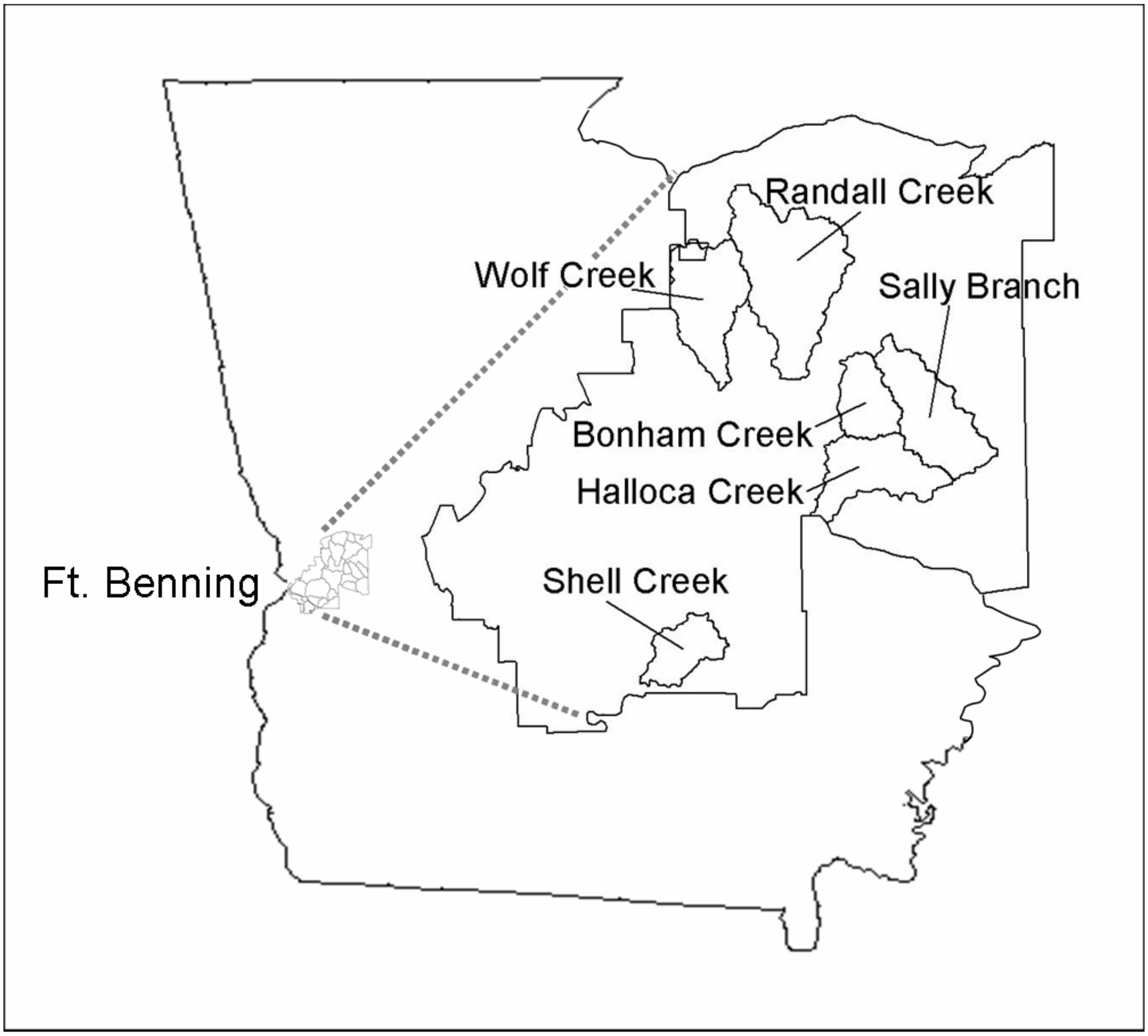

Randall Creek
Wolf Creek
Sally Branch
Bonham Creek
Halloca Creek
Ft. Benning
Shell Creek

Figure 3.

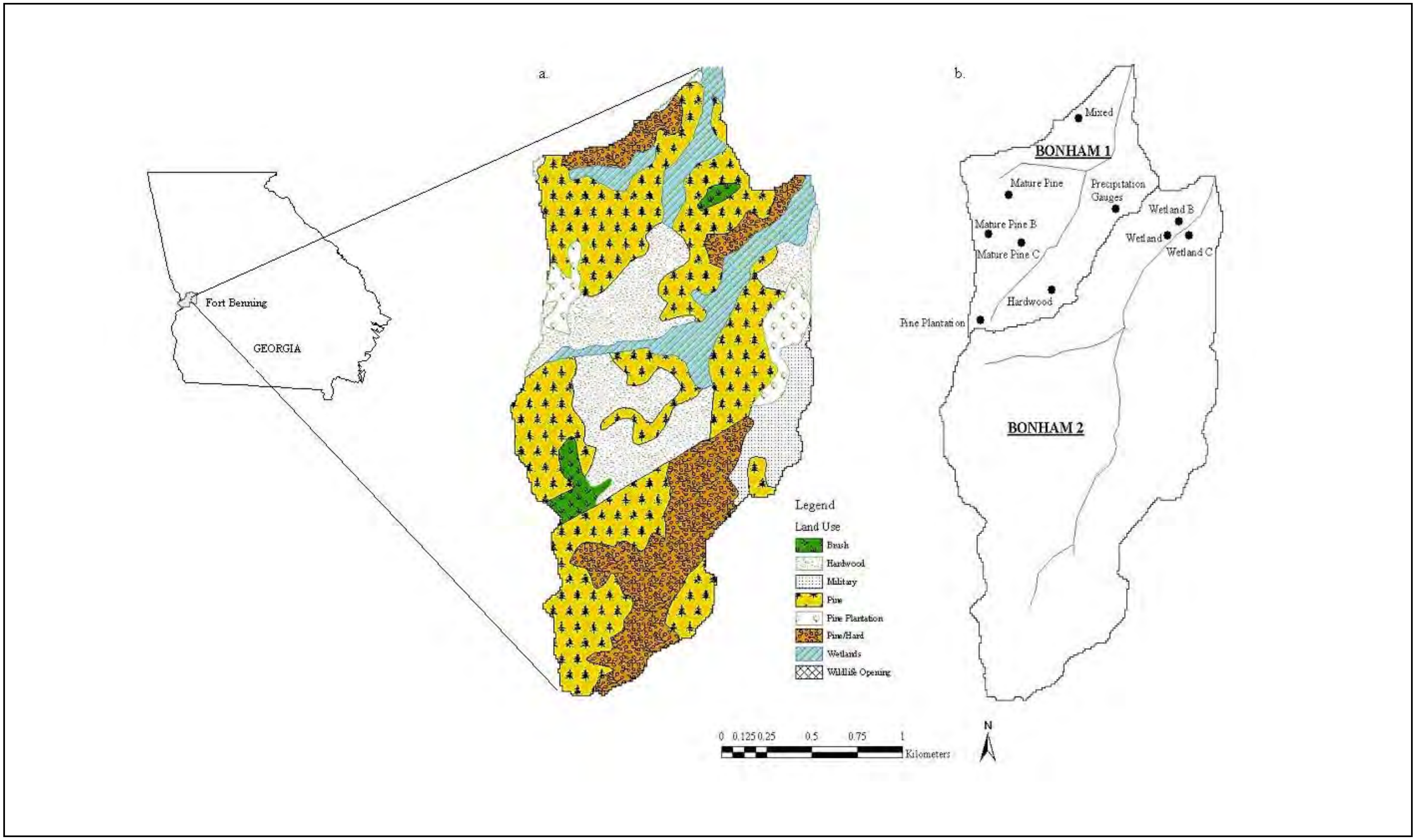

a.
b.
Fort Benning
GEORGIA
Mixed
BONHAM 1
Mature Pine
Precipitation Gauges
Wetland B
Mature Pine B
Wetland
Wetland C
Mature Pine C
Hardwood
Pine Plantation
BONHAM 2
Legend
Land Use
Brush
Hardwood
Military
Pine
Pine Plantation
Pine/Hard
Wetlands
Wildlife Opening
0 0.125 0.25 0.5 0.75 1
Kilometers
N

Figure 4.

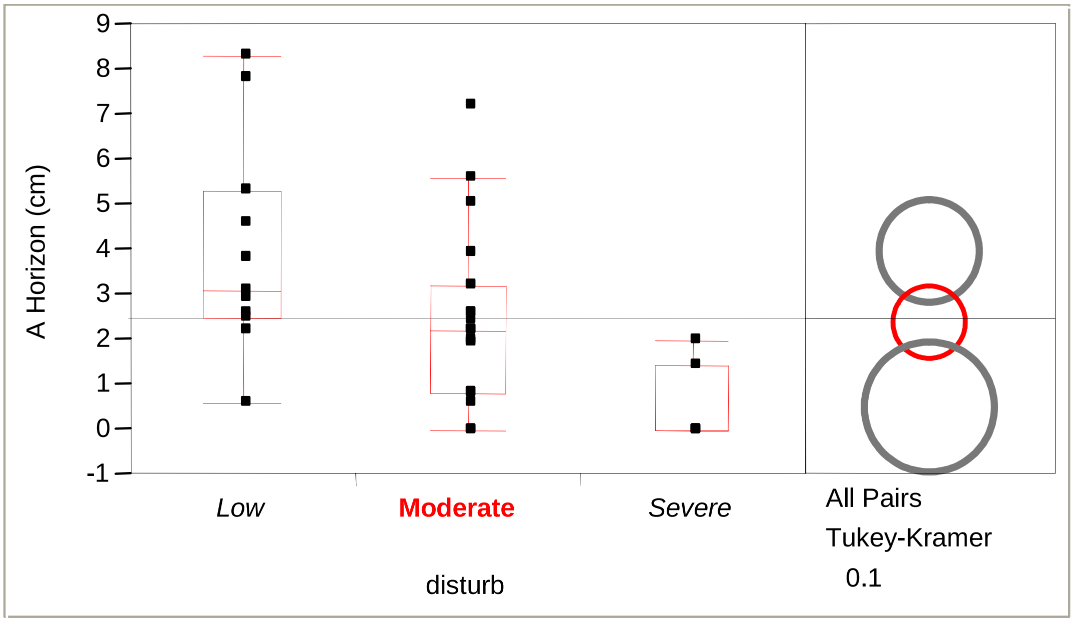

A Horizon (cm)
9
8
7
6
5
4
3
2
1
0
-1
Low
Moderate
Severe
disturb
All Pairs
Tukey-Kramer
0.1

Figure 5.

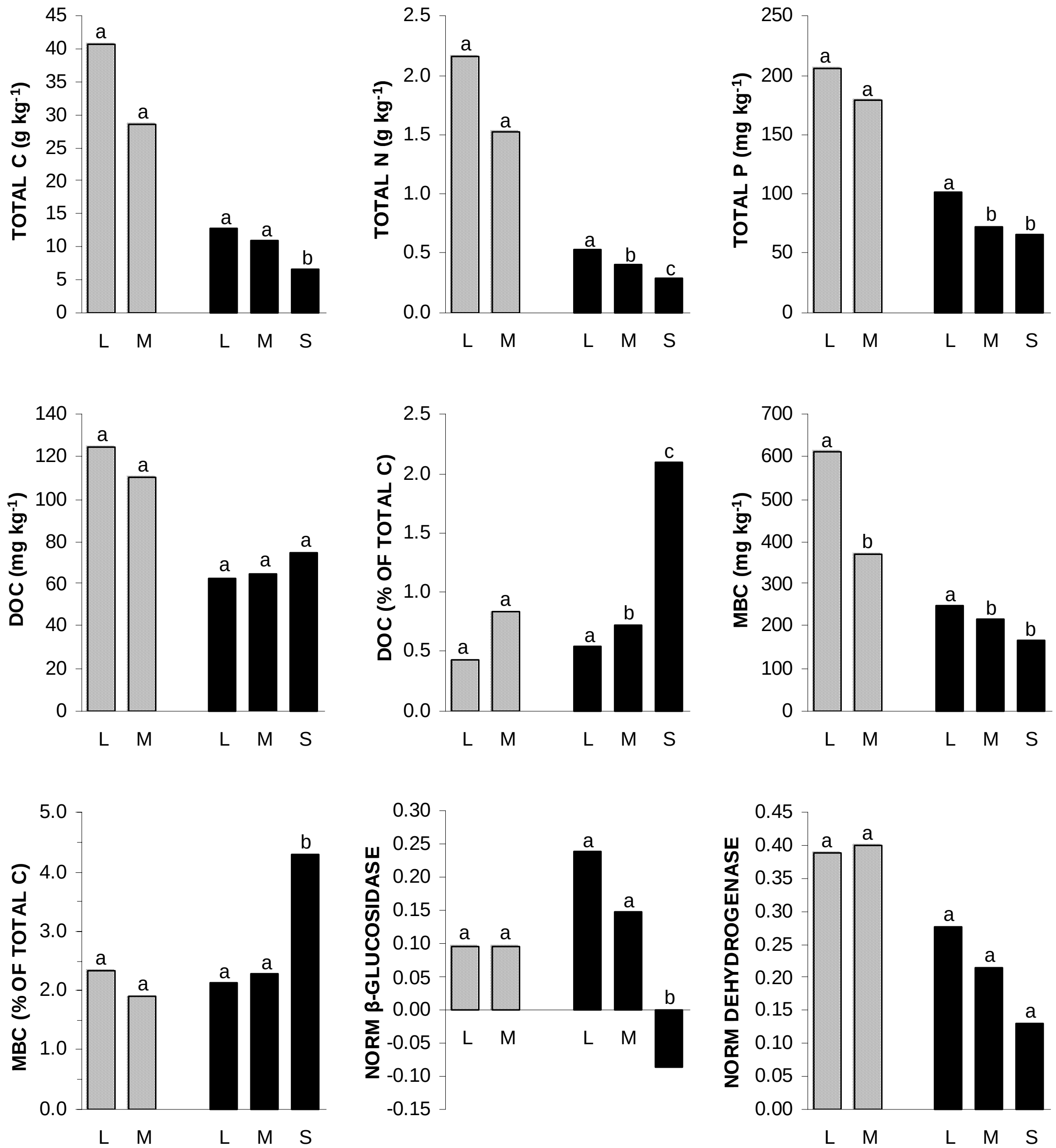


Figure 6. Summary of Phase 1 analysis (300 sites) of selected soil properties associated with organic matter storage and turnover, for low- (L), moderate- (M) and severe- (S) disturbance sites in uplands (solid bars) and bottomlands (hatched bars). Data from ridge-top and mid-slope sites were grouped into a single upland category. Mean values are presented; significantly different values (P<0.05) within each landscape category are indicated by different letters. All wetland sites sampled during Phase 1 were classified as either "low" or "moderate" disturbance, hence there is no "severe" class shown for wetlands.

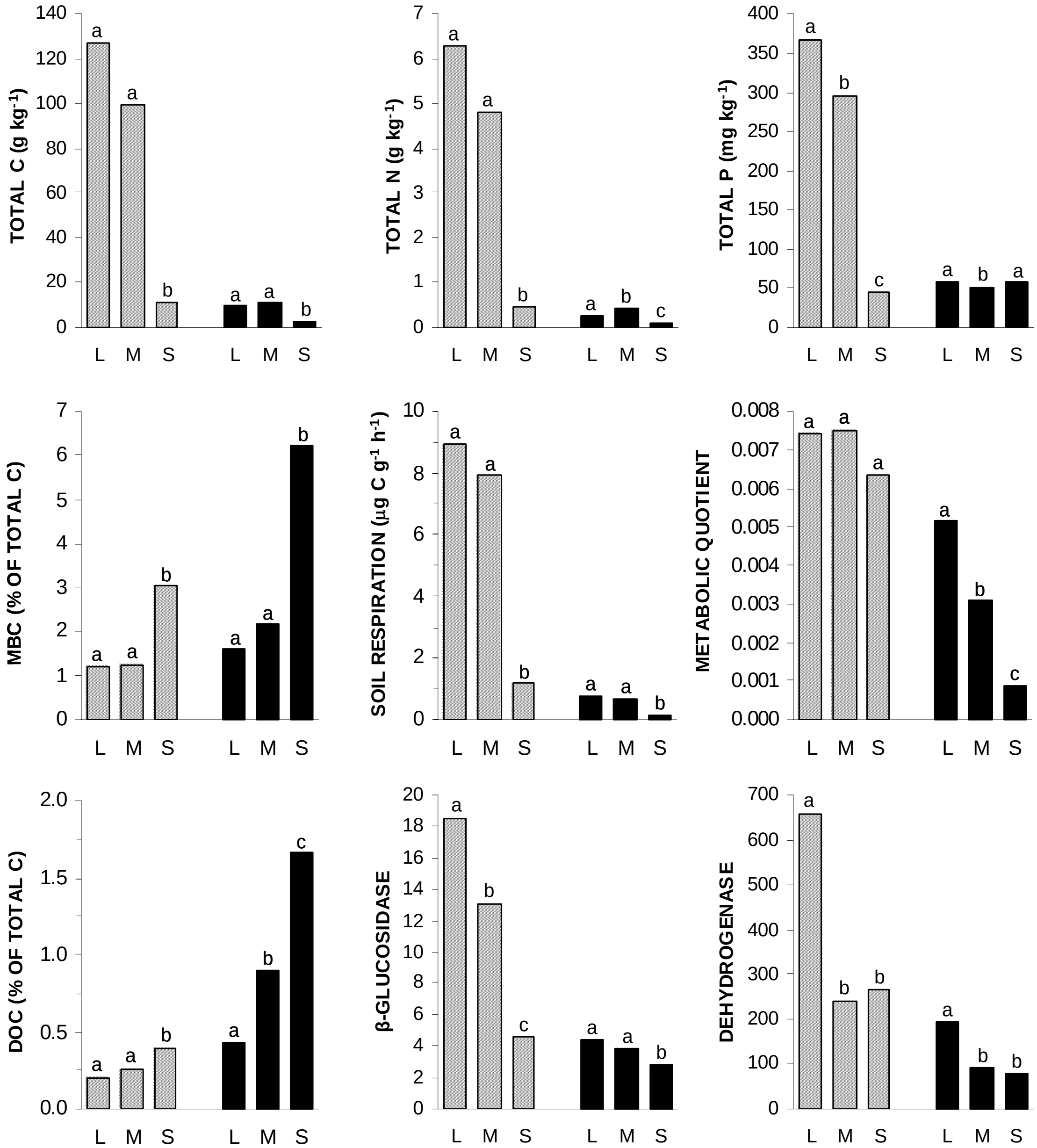


Figure 7. Summary of Phase 2 (transects) analysis of selected soil properties associated with organic matter storage and turnover, for low- (L), moderate- (M) and severe- (S) disturbance sites in uplands (solid bars) and bottomlands (hatched bars). Mean values are presented; significantly different values (P<0.05) within each landscape category are indicated by different letters.

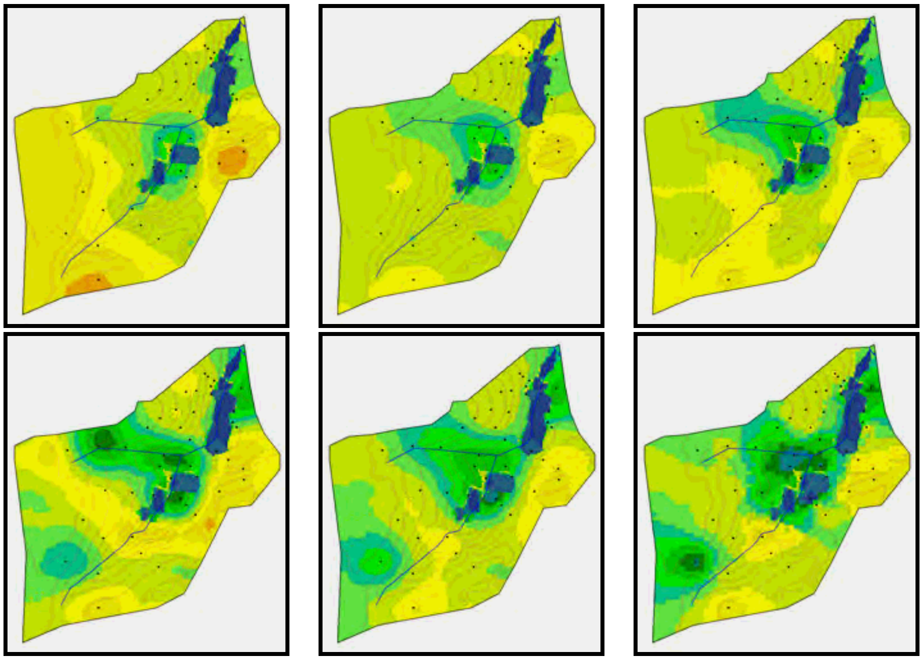


Figure 8. Depth-specific water content estimation maps of Bonham-1 watershed for March, 2002. Blue areas contain more water than yellow and brown areas.

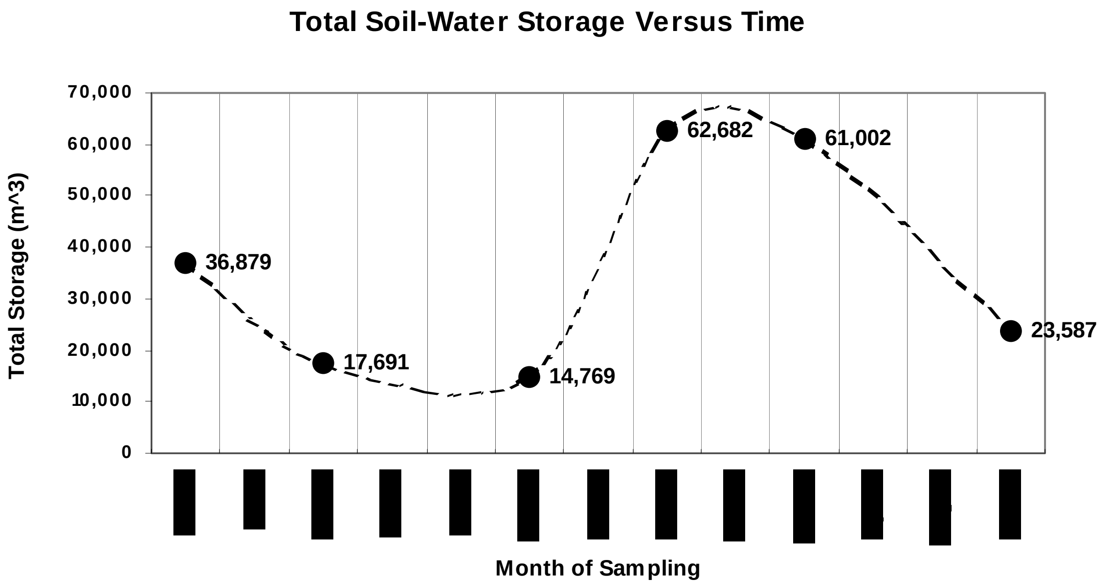


Figure 9. Observed temporal trend in total soil-water storage.

4.0 PROJECT MILESTONES (FY2003)

CS-1114A-99 – Determination of Indicators of Ecological Change (Univ. of Florida)			
Presentation to Advisory Committee Meeting	Apr 04	Apr 04	Done
Submit project close-out plan as appropriate	Apr 04	August 04	See below
Hyperspectral Analysis of Soils	Apr 04	Apr 04	Publication in preparation
Generate water input data for Ft. Benning region	Apr 04	April 04	Final updates were completed on time. Several unsuccessful attempts were made to enter data in the respository. However, due to repository problems accepting access database files, the final submission was unable to be completed. A final attempt will be made in August, 2004. If repository uploads again fail, the data will be transferred via email with appropriate documentation.
Adapt riparian management model to Ft. Benning system	Apr 04	September 04	Will be conducted in August and September by Shirish Bhat in conjunction with USDA in Tifton, GA
Total of 14 journal submissions by 15 May	May 04	May 04	Eight publications submitted; five in preparation; three under revision
Litter decomposition and carbon dynamics	Jun 4 04	Jun 04	All litter bags and soil study samples recovered. Data analysis continuing.

5.0 CLOSE-OUT PLAN

- Submit photos of 276 study sites to SEMP Data Repository-September 2004
- Complete experimentation and testing of Riparian Management Model – September 2004

- Presented data on riparian assessment at Society of Wetland Scientists Annual Meeting – July 2004

- Complete preliminary analysis of hyperspectral data from Bonham riparian study – October 2004

- Final Report – September 2004

- Continue working on manuscripts

6.0 PUBLICATIONS, THESES, DISSERTATIONS, AND PRESENTATIONS

6.1 Publications

Archer, J and D.L. Miller. Understory Vegetation and Soil response to silvicultural activity in a southeastern mixed pine forest: A chronosequence study.
Date of submission: January 2004
Journal: Journal of Forest Ecology and Management
Status: **Submitted**

Bhat, S., J.M. Jacobs, K. Hatfield, J. Prenger. 2004. Ecological Indicators in Forested Watersheds: Relationships between Watershed Characteristics and Stream Water Quality in Fort Benning, GA.
Date of submission: February 2004
Journal: Journal of Hydrology

Bryant, M.L., Bhat, S., and J.M. Jacobs. 2004. Spatiotemporal throughfall characterization of heterogeneous forest communities in the southeastern U.S.
Date of submission: February 2003. Accepted for Publication
Journal: Journal of Hydrology
Status: **In Press**

Comerford, N., Prenger, J.P., B.L. Skulnick, and W.F. DeBusk. 2004. Organic C Storage and Cycling as Indicators of Soil Condition for Military and Forestry Land Management.
Date of submission: January 2004
Journal: Soil Science Society of America Journal.

Chen, Weiwei, A. Ogram, and W.F. DeBusk. 2004. Optimization of terminal restriction fragment length polymorphism (t-RFLP) and Characterization of AOB community structure in acidic soils determined by t-RFLP of *amoA*.
Date of submission: February 2004
Journal: Microbial Ecology.

Dabral, S., W. D. Graham, and J.P. Prenger. Development of a spectral reflectance technique for predicting soil quality and classifying ecological disturbance.

Date of submission: February 2004
Journal: Forest Ecology and Management

Cohen, M.J., S. Dabral, W. D. Graham, and J.P. Prenger. Quantitative analysis of soil nutrient concentrations with near infrared spectroscopy and partial least squares regression.
Date of submission: October 2004
Journal: Journal of Environmental Mangement

Perkins, D., N. Haws, B. S. Das, and P.S. C. Rao. 2004. Soil Hydraulic Properties as Indicators of Land Quality for Upland Soils in Forested Watersheds with Military Training Impacts.
Date of submission: December 2003
Journal: Soil Science Society of America Journal.
Status: **Submitted**

Perkins, D., N. Haws, S. Rao, J. Jawitz. 2004. Hydraulic Conductivity of Upland Soils in Forested Watersheds at Ft Benning, GA: Assessment of Mechanized Military Training.
Date of submission: March 2004
Journal: Soil Science Society of America Journal.

Prenger, J.P., Bhat, S., J.M. Jacobs, and K. R. Reddy. Microbial Nutrient Cycling in the Riparian Zone and its Influence on Stream Chemistry.
Date of submission: May 2004
Journal: Journal of Environmental Quality.

Prenger, J. P., W. F. DeBusk, and K. R. Reddy. 2004. Influence of military land management on extracellular soil enzymes.
Date of submission: September 2004
Journal: Soil and Water Conservation

Reddy, K. R, and all PIs. Hydrologic, soil, and vegetation indicators of change in forested ecosystem: Synthesis of an interdisciplinary project.
Date of submission: October, 2004
Journal: Ecological Indicators

DeBusk, W. F., B. L. Skulnick, J. P. Prenger, and K. R. Reddy. Response of Soil Organic Carbon Dynamics to Disturbance by Military Training in the Southeastern U.S.
Date of submission: September, 2004
Journal: Soil and Water Conservation

Tanner, G.W. and D.L. Miller. Understory vegetation composition along a disturbance gradient within a mixed pine-hardwood forest, Ft. Benning Army Reservation, Georgia.
Date of submission: February 2004
Journal: Ecological Restoration

6.2 Theses And Dissertations

Archer, J. K. Understory vegetation and soil response to silvicultural activity in a southeastern mixed pine forest: A chronosequence study. M.S. Thesis. University of Florida.

Bryant, M.L. 2002 Spatiotemporal Throughfall Characterization of Heterogeneous Forest Communities in the Southeastern U.S. M.S. Thesis. University of Florida

Chen, Weiwei. 2001. Optimization of terminal restriction fragment length polymorphism and evaluation of microbial community structure as indicator of ecosystem integrity. M.S. Thesis, University of Florida.

Perkins, D. 2003. Soil Hydrologic Characterization and Soil-water Storage Dynamics in a Forested Watershed. M. S. Thesis. Purdue University.

Skulnick, B. L. 2002. Soil carbon biogeochemistry: Indicators of ecological disturbance. M. S. Thesis. University of Florida

Tkaczyk, M 2002. Rainfall Runoff and Subsurface Flow Analysis to Investigate the Flow Paths in Forested Watersheds Utilizing TOPMODEL . M. S. Thesis. Civil and Materials Engineering Department, University of Illinois at Chicago

6.3 Presentations

2000

Prenger, J.P., B.L. Skulnick, and W.F. DeBusk. 2000. Enzyme activity assays as indicators of environmental impact. Annual Meeting of Soil Science Society of America, Minneapolis, MN, November 5-9, 2000.

2001

Bhat, S., J.M. Jacobs, W. Graham, P.S. Rao, N. Haws, W.F. DeBusk, J.W. Jawitz. 2001. Identification of Eco-Hydrologic Indicators of Ecological Impact: Phase I Results from Fort Benning, Georgia Watersheds, Eos Trans. AGU, 82 (20), Spring Meet. Suppl., Abstract H42C-02, 2001.

Dabral, S., W. D. Graham, J. Prenger, and W. F. DeBusk. 2001. Determination of Soil, Hydrologic, and Vegetation Indicators for Military Land Management Ft. Benning Georgia. Graduate Research Forum, University of Florida, 2001.

DeBusk, W. F., and J. P. Prenger. 2001. Wetland soil biogeochemical indicators of ecological condition for military land management. Poster presented at Annual Meeting of Society of Wetland Scientists, May 28 – June 1, 2001, Chicago, IL

Jacobs, J. J., S. Bhat, W. D. Graham, P. S. C. Rao, N. Haws, W. F. DeBusk, and J. W. Jawitz. 2001. Identification of eco-hydrologic indicators of ecological impact: Phase I results from Fort Benning, Georgia watersheds. Poster presented at Spring 2001 Meeting of the American Geophysical Union, May 29 – June 1, 2001, Boston, MA.

2002

Archer, Jessica K. and D. Miller. 2002. Vegetation and soil response to timber thinning operations: a chronosequence study. Abst. Annual Meeting Society of Ecological Society of America, Tuscon AZ. August 2-9, 2002.

Chen, Weiwei, W. DeBusk, and A. Ogram. 2002. Evaluation of T-RFLP and characterization of Type II methanotroph assemblage composition in degraded and undegraded forest soils. Abstracts of the Annual Meeting of the American Society for Microbiology, Salt Lake City, UT.

DeBusk, W. F., and J. P. Prenger. 2002. Soil Biogeochemical Indicators for Wetland and Watershed Assessment. Oral presentation at Annual Meeting of Society of Wetland Scientists, June 2-7, 2002, Lake Placid, NY.

DeBusk, W. F. 2002. Determination of indicators of ecological change. Presented at the TAC meeting during April 15-17, 2002, Arlington, VA.

Prenger, J.P., B.L. Skulnick, and W.F. DeBusk. 2002. Organic C Storage and Cycling As Indicators of Ecological Condition For Military Land Management. Poster to be presented at Annual Meeting of Soil Science Society of America, November 11-14, 2002, Indianapolis, IN.

Reddy, K. R. 2002. Determination of indicators of ecological change. Presented at the TAC meeting during Oct. 28-29, 2002, Columbus, GA.

Tkaczyk, M, J.W. Jawitz, J.M. Jacobs, S. Bhat, P.S. Rao, N. Haws. 2002. Rainfall/Runoff Analysis to Investigate the Effects of Soil Heterogeneity on Watershed Response Utilizing Topmodel, Eos Trans. AGU, 83 (19), Spring Meet. Suppl., Abstract H42C-02, 2002.

Tkaczyk, M, J.W. Jawitz, J.M. Jacobs, S. Bhat, P.S. Rao, N. Haws, Rainfall/Runoff Analysis to Investigate the Effects of Soil Heterogeneity on Watershed Response Utilizing Topmodel, Eos Trans. AGU, 83 (19), Spring Meet. Suppl., Abstract H42C-02, 2002.

2003

Bhat, S., S.R. Satti, J.M. Jacobs, K. Hatfield, Ecological Indicators in Diversified Forested Watersheds: Relationships between Watershed Characteristics and Stream Water Quality in Fort Benning, GA, Eos Trans. AGU, 84 (20), Fall Meet. Suppl., 2003.

Prenger, J.P., and W.F. DeBusk. 2003. Changes In Soil Microbial Activity Related To Military Training And Forestry Activities. Poster presented at Annual Meeting Society of Ecological Society of America, Savannah GA. August 3-8, 2003.

Prenger, J.P., and W.F. DeBusk. 2003. Changes In Soil Microbial Activity In Riparian Wetlands Related To Military Training And Forestry Activities. Poster presented at Annual Meeting of Society of Wetland Scientists, June 8-13, 2003, New Orleans, LA.

Prenger, J.P., K. R. Reddy, S. Bhat, and J. Jacobs. 2003. Microbial nutrient cycling in the riparian zone of a coastal plain stream. Poster presented at the 8th International Symposium on Biogeochemistry of Wetlands, Sept. 14-17, 2003. Gent, Belgium.

2004

Prenger, J.P., and M.J. Cohen. 2004. The Use of Near Infrared Reflectance Spectroscopy on Riparian Soils to Characterize Watershed Disturbance. Oral presentation at Annual Meeting of Society of Wetland Scientists, July 18-23, 2004, Seattle, WA.