FINAL REPORT

Disturbance of Soil Organic Matter and Nitrogen Dynamics: Implications For Soil and Water Quality

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DISTURBANCE OF SOIL ORGANIC MATTER AND NITROGEN DYNAMICS: IMPLICATIONS FOR SOIL AND WATER QUALITY (CS-1114D-00)

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14. ABSTRACT
Research was conducted on soil carbon and nitrogen dynamics at Fort Benning, GA, from October 1999 to June 2004. The objectives of the research were to (1) develop a better understanding of the effects of disturbance on key measures of soil quality at Fort Benning, and (2) determine if there are thresholds of soil quality that potentially affect ecosystem recovery or sustainability. The completed research was relevant to SERDP because it addressed several objectives in the Statement of Need No. CSSON-00-03 titled “Ecological Disturbance in the Context of Military Landscapes.” In particular, the research addressed the SON objective “to determine whether there are thresholds in spatial extent, intensity or frequency above and/or below which the natural system cannot sustain identified ecological and/or land use disturbances.”

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Introduction

Research was conducted on soil carbon and nitrogen dynamics at Fort Benning, GA, from October 1999 to June 2004. The objectives of the research were to (1) develop a better understanding of the effects of disturbance on key measures of soil quality at Fort Benning, and (2) determine if there are thresholds of soil quality that potentially affect ecosystem recovery or sustainability. The completed research was relevant to SERDP because it addressed several objectives in the Statement of Need No. CSSON-00-03 titled "Ecological Disturbance in the Context of Military Landscapes." In particular, the research addressed the SON objective "to determine whether there are thresholds in spatial extent, intensity or frequency above and/or below which the natural system cannot sustain identified ecological and/or land use disturbances."

There were five broadly based technical objectives associated with the research: (1) characterize effect of disturbances and land cover/land use on soil quality, (2) predict disturbance thresholds to ecosystem recovery, (3) model soil organic matter for different land cover types, (4) contribute to and conduct field experiments on ecosystem disturbance, and (5) analyze spatial patterns of soil carbon and nitrogen for the purpose of predicting potential non-point nitrogen sources on the landscape. Data from the research has been submitted to the SEMP Data Repository in multiple data sets and is available via the internet. A publication plan has been developed and is summarized in Table 1. The principal findings from each technical objective associated with the research are summarized in this final report. For additional details, the reader is referred to various ORNL technical reports (see Appendices).

Technical Objective 1:
Characterize Effect of Disturbances and Land Cover/Land Use on Soil Quality

The purpose of this task was to investigate the effects of soil disturbance on several key indicators of soil quality at Fort Benning, Georgia. Military activities at Fort Benning that result in soil disturbance include infantry, artillery, wheeled, and tracked vehicle training. Soil samples were collected along a disturbance gradient that included: (1) reference sites, (2) light military use, (3) moderate military use, (4) heavy military use, and (5) remediated sites (Appendix A). With the exception of surface soil bulk density, measured soil properties at reference and light use sites were similar. Relative to reference sites, greater surface soil bulk density, lower soil carbon concentrations, and less carbon and nitrogen in particulate organic matter (POM) were found at moderate use, heavy use, and remediated sites. Studies along a pine forest chronosequence indicated that carbon stocks in POM gradually increased with stand age. An analysis of soil C:N ratios, as well as soil carbon concentrations and stocks, indicated a recovery of soil quality at moderate military use and remediated sites relative to heavy military use sites. Measurements of soil carbon and nitrogen are ecological indicators that can be used by military land managers to identify changes in soil from training activities and to rank training areas on the basis of soil quality (Garten et al., 2003).

Table 1. ORNL Team 2 Publication Plan

Journal Publications

Published:
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.

In Preparation:

Garten, C.T., Jr., and T.L. Ashwood. Modeling Soil Quality Thresholds to Ecosystem Recovery at Fort Benning, Georgia, USA.
Journal: Ecological Engineering; Date of submission: 30 June May 26, 2004

Garten, C.T., Jr. Predicted Effects of Prescribed Burning and Timber Management on Forest Recovery and Sustainability at Fort Benning, Georgia, USA.
Journal: Journal of Environmental Management; Date of submission: 30 July 2004

Garten, C.T., Jr. Effects of Heavy, Tracked-Vehicle Disturbance on Forest Soil Properties at Fort Benning, Georgia, USA.
Journal: Science of the Total Environment; Date of submission: 30 August 2004

Published Technical Reports

Garten, C.T., Jr., and T.L. Ashwood. 2004. Land Cover Differences in Soil Carbon and Nitrogen at Fort Benning, Georgia (ORNL/TM-2004/14). Oak Ridge National Laboratory, Oak Ridge, Tennessee 37830
Garten, C.T., Jr., and T.L. Ashwood. 2004. Modeling Soil Quality Thresholds to Ecosystem Recovery at Fort Benning, Georgia, USA (ORNL/TM-2004/41). Oak Ridge National Laboratory, Oak Ridge, Tennessee 37830
Garten, C.T., Jr. 2004. Predicted Effects of Prescribed Burning and Timber Management on Forest Recovery and Sustainability at Fort Benning, Georgia (ORNL/TM-2004/77). Oak Ridge National Laboratory, Oak Ridge, Tennessee 37830
Garten, C.T., Jr., and T.L. Ashwood. 2004. Effects of Heavy, Tracked-Vehicle Disturbance on Forest Soil Properties at Fort Benning, Georgia (ORNL/TM-2004/76). Oak Ridge National Laboratory, Oak Ridge, Tennessee 37830.

Land cover characterization might also help land managers assess the impacts of management practices and land cover change on attributes linked to the maintenance and/or recovery of soil quality. However, connections between land cover and measures of soil quality are not well established. We examined differences in soil carbon and nitrogen among various land cover types at Fort Benning, Georgia. Forty-one sampling sites were classified into five major land cover types: deciduous forest, mixed forest, evergreen forest or plantation, transitional herbaceous vegetation, and barren land (Appendix B).

Key measures of soil quality (including mineral soil density, nitrogen availability, soil carbon and nitrogen stocks, as well as properties and chemistry of the O-horizon) were significantly different among the five land covers. In general, barren land had the poorest soil quality. Barren land , created through disturbance by tracked vehicles and/or erosion, had significantly greater soil density and a substantial loss of carbon and nitrogen relative to soils at less disturbed sites. It was estimated that recovery of soil carbon under barren land at Fort Benning to current day levels under transitional vegetation or forests would require about 60 years following reestablishment of vegetation. Maps of soil carbon and nitrogen were produced for Fort Benning based on a 1999 land cover map and field measurements of soil carbon and nitrogen stocks under different land cover categories (Garten and Ashwood, 2004a).

Technical Objective 2:
Determine Disturbance Thresholds to Ecosystem Recovery

The objective of this task was to use a simple model of soil carbon and nitrogen dynamics to predict nutrient thresholds to ecosystem recovery on degraded soils at Fort Benning, Georgia. The model calculates aboveground and belowground biomass, soil carbon inputs and dynamics, soil nitrogen stocks and availability, and plant nitrogen requirements. A threshold is crossed when predicted soil nitrogen supplies fall short of predicted nitrogen required to sustain biomass accrual at a specified recovery rate. Four factors were important to development of thresholds to recovery: (1) initial amounts of aboveground biomass, (2) initial soil carbon stocks (i.e., soil quality), (3) relative recovery rates of biomass, and (4) soil sand content (Appendix C). Thresholds to ecosystem recovery predicted by the model should not be interpreted independent of a specified recovery rate. Initial soil carbon stocks influenced the predicted patterns of recovery by both old field and forest ecosystems. Forests and old fields on soils with varying sand content had different predicted thresholds to recovery. Soil carbon stocks at barren sites on Fort Benning generally lie below predicted thresholds to 100% recovery of desired future ecosystem conditions defined on the basis of aboveground biomass (18000 versus 360 g m^{-2} for forests and old fields, respectively). Calculations with the model indicated that reestablishment of vegetation on barren sites to a level below the desired future condition is possible at recovery rates used in the model, but the time to 100% recovery of desired future conditions, without crossing a nutrient threshold, is prolonged by a reduced rate of forest growth. Predicted thresholds to ecosystem recovery were less on soils with more than 70% sand content. The lower thresholds for old field and forest recovery on more sandy soils are apparently due to higher relative rates of net soil nitrogen mineralization in more sandy soils. Calculations with the model indicate that a combination of desired future conditions, initial levels of soil quality (defined by soil carbon stocks), and the rate of biomass accumulation determines the predicted success of ecosystem recovery on disturbed soils (Garten and Ashwood, 2004b).

Technical Objective 3:
Model Soil Organic Matter for Different Land Cover Types

The objective of this task was to use a simple compartment model of soil carbon and nitrogen dynamics to predict forest recovery on degraded soils and forest sustainability, following recovery, under different regimes of prescribed fire and timber management. The task included a model-based analysis of the effect of prescribed burning and forest thinning or clearcutting on stand recovery and sustainability at Fort Benning, GA. I developed the model using Stella® Research Software (High Performance Systems, Inc., Hanover, NH) and parameterized the model using data from field studies at Fort Benning, literature sources, and parameter fitting (Appendix D). The model included (1) a tree biomass submodel that predicted aboveground and belowground tree biomass, (2) a litter production submodel that predicted the dynamics of herbaceous aboveground and belowground biomass, (3) a soil carbon and nitrogen submodel that predicted soil carbon and nitrogen stocks (to a 30 cm soil depth) and net soil nitrogen mineralization, and (4) an excess nitrogen submodel that calculated the difference between predicted plant nitrogen demands and soil nitrogen supplies. There was a modeled feedback from potential excess nitrogen (PEN) to tree growth such that forest growth was limited under conditions of nitrogen deficiency.

Two experiments were performed for the model-based analysis. In the first experiment, forest recovery from barren soils was predicted for 100 years with or without prescribed burning and with or without timber management by thinning or clearcutting. In the second experiment, simulations began with 100 years of predicted forest growth in the absence of fire or harvesting, and sustainability was predicted for a further 100 years either with or without prescribed burning and with or without forest management. Four performance variables (aboveground tree biomass, soil carbon stocks, soil nitrogen stocks, and PEN) were used to evaluate the predicted effects of timber harvesting and prescribed burning on forest recovery and sustainability.

Predictions of forest recovery and sustainability were directly affected by how prescribed fire affected PEN. Prescribed fire impacted soil nitrogen supplies by lowering predicted soil carbon and nitrogen stocks which reduced the soil nitrogen pool that contributed to the predicted annual flux of net soil nitrogen mineralization. On soils with inherently high nitrogen availability, increasing the fire frequency in combination with stand thinning or clearcutting had little effect on predictions of forest recovery and sustainability. However, experiments with the model indicated that combined effects of stand thinning (or clearcutting) and frequent prescribed burning could have adverse effects on forest recovery and sustainability when nitrogen availability was just at the point of limiting forest growth. Model predictions indicated that prescribed burning with a 3-year return interval would decrease soil carbon and nitrogen stocks but not adversely affect forest recovery from barren soils or sustainability following ecosystem recovery. On soils with inherently low nitrogen availability, prescribed burning with a 2-year return interval depressed predicted soil carbon and nitrogen stocks to the point where soil nitrogen deficiencies prevented forest recovery as well as forest sustainability following recovery (Garten, 2004).

Technical Objective 4:
Contribute to and Conduct Field Experiments on Ecosystem Disturbance

The purpose of this task was to examine the effects of heavy, tracked-vehicle disturbance on various measures of soil quality in training compartment K-11 at Fort Benning, Georgia. Pre-disturbance soil sampling in April and October of 2002 indicated statistically significant differences in soil properties between upland and riparian sites (Appendix E). Soil density was less at riparian sites, but riparian soils had significantly greater carbon and nitrogen concentrations and stocks than upland soils. Most of the carbon stock in riparian soils was associated with mineral-associated organic matter (i.e., the silt + clay fraction physically separated from whole mineral soil). Topographic differences in soil nitrogen availability were highly dependent on the time of sampling. Riparian soils had higher concentrations of extractable inorganic nitrogen than upland soils and also exhibited significantly greater soil nitrogen availability during the spring sampling.

The disturbance experiment was performed in May 2003 by driving a D7 bulldozer through the mixed pine/hardwood forest. Post-disturbance sampling was limited to upland sites because training with heavy, tracked vehicles at Fort Benning is generally confined to upland soils. Soil sampling approximately one month after the experiment indicated that effects of the bulldozer were limited primarily to the forest floor (O-horizon) and the surface (0-10 cm) mineral soil. O-horizon dry mass and carbon stocks were significantly reduced, relative to undisturbed sites, and there was an indication of reduced mineral soil carbon stocks in the disturbance zone. Differences in the surface (0-10 cm) mineral soil also indicated a significant increase in soil density as a result of disturbance by the bulldozer. Although there was some tendency for greater soil nitrogen availability in disturbed soils, the changes were not significantly different from undisturbed controls. It is expected that repeated soil disturbance over time, which will normally occur in a military training area, would simply intensify the changes in soil properties that were measured following a one-time soil disturbance at the K-11 training compartment.

The experiment was also useful for identifying soil measurements that are particularly sensitive to disturbance and therefore can be used successfully as indicators of a change in soil properties as a result of heavy, tracked-vehicle traffic at Fort Benning. Measurements related to total O-horizon mass and carbon concentrations or stocks exhibited changes that ranged from ≈25 to 75% following the one-time disturbance. Changes in surface (0-10 cm) mineral soil density or measures of surface soil carbon and nitrogen following the disturbance were less remarkable and ranged from ≈15 to 45% (relative to undisturbed controls). Soil nitrogen availability (measured as initial extractable soil nitrogen or nitrogen production in laboratory incubations) was the least sensitive and the least useful indicator for detecting a change in soil quality. Collectively, the results suggest that the best indicators of a change in soil quality will be found at the soil surface because there were no statistically significant effects of bulldozer disturbance at soil depths below 10 cm (Garten and Ashwood, 2004c).

Technical Objective 5:
Analyze Spatial Patterns of Soil Carbon and Nitrogen for the Purpose of Predicting Potential Non-Point Nitrogen Sources on the Landscape

The purpose of this task was to spatially assess the amount of potential excess nitrogen on Fort Benning through the use of a GIS-based model of nitrogen cycle processes. The analysis was performed in the following steps: (1) development of a conceptual model to quantify potential excess soil nitrogen (PEN), (2) acquisition and re-categorization of a land use/cover map of Fort Benning that was derived from Landsat Thematic Mapper data, (3) development of nitrogen flux maps for each of five nitrogen cycle processes by acquisition of field data and estimation of nitrogen fluxes under different land covers from a literature review, (4) calculation of seasonal and annual PEN using GIS-based spatial models, and (5) comparison of PEN between land use categories. The model predicted the spatial distribution of seasonal and annual nitrogen sources and sinks and estimated the amount of nitrogen flux using a mass balance model of three input processes (atmospheric nitrogen deposition, fertilization, net soil nitrogen mineralization) and two output processes (plant uptake and denitrification). Net soil nitrogen mineralization was the primary contributing process to annual and seasonal estimates of PEN. Potential excess nitrogen was positive (a potential source) when potential inputs exceeded potential outputs. Negative PEN indicated a potential sink. The results indicated that most of Fort Benning is a net sink for nitrogen only 6 % of the landscape was identified as a source of PEN. Positive PEN values were primarily associated with urban land uses, particularly roads and cantonment areas. Barren areas were also identified by the model as having positive PEN values. Information and experience obtained as a result of this technical objective will contribute to another SERDP Project (SERDP 1259) directed at developing a regional simulation model (RSim) to explore impacts of resource use and constraints in the five county region surrounding Fort Benning.

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APPENDIX A.

Effect of military training on indicators of soil quality at Fort Benning, Georgia

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Effect of military training on indicators of soil quality at Fort Benning, Georgia☆

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Abstract

The purpose of this research was to investigate the effects of soil disturbance on several key indicators of soil quality at Fort Benning, Georgia. Military activities at Fort Benning that result in soil disturbance include infantry, artillery, wheeled, and tracked vehicle training. Soil samples were collected along a disturbance gradient that included: (1) reference sites, (2) light military use, (3) moderate military use, (4) heavy military use, and (5) remediated sites. With the exception of surface soil bulk density, measured soil properties at reference and light use sites were similar. Relative to reference sites, greater surface soil bulk density, lower soil carbon concentrations, and less carbon and nitrogen in particulate organic matter (POM) were found at moderate use, heavy use, and remediated sites. Studies along a pine forest chronosequence indicated that carbon stocks in POM gradually increased with stand age. An analysis of soil C:N ratios, as well as soil carbon concentrations and stocks, indicated a recovery of soil quality at moderate military use and remediated sites relative to heavy military use sites. Measurements of soil carbon and nitrogen are ecological indicators that can be used by military land managers to identify changes in soil from training activities and to rank training areas on the basis of soil quality.

Keywords: Soil carbon; Soil nitrogen; Particulate organic matter; Mineral-associated organic matter; Soil disturbance gradient

1. Introduction

Military land managers are faced with the challenge of using a given amount of land for the purpose of military training and troop readiness. Ideally, this mission must be accomplished in an ecologically sound manner that meets military requirements and, at the same time, promotes the sustainability of ecosystems so that the military mission is not compromised by a degraded landscape. Military installations are, in some respects, representative of a larger set of issues faced by managers of government or public lands. Areas set aside for public benefit, like national parks and recreational areas, can suffer a slow and almost undetectable degradation if the land is over-utilized by long-term human activities. This occurs as an increasing number of people seek an outdoor experience on a finite amount of recreational land. Incipient degradation in a landscape is sometimes visible (e.g. trampling of vegetation, vehicle tracks through otherwise undisturbed areas, and erosion created by the overuse of trails), but changes

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in soil quality can be too subtle to be easily detected by visual observations.

Vegetation dynamics and the sustainability of terrestrial ecosystems depends, in part, on soil quality which can be defined as "the capacity of a soil to function within ecosystem boundaries to sustain biological productivity, maintain environmental quality and promote plant and animal health" (Doran and Parkin, 1994). Organic matter (indicated by soil carbon) and soil nitrogen are critical determinants of soil quality because of their relationship to soil structure and nutrient supply (Gregorich et al., 1994; Doran and Parkin, 1996). Soil organic matter also plays an essential part in soil aggregate formation, soil structure, and infiltration capacity (Boyle et al., 1989). Free organic matter, consisting mainly of fresh plant residues, is an important energy source for heterotrophic soil microorganisms and plant available soil nitrogen (Magdoff, 1996). Total soil carbon and nitrogen are potential ecological indicators that could help land managers determine when there is a change in the landscape that could signal either a disturbance or a recovery of soil quality.

Disturbance of soil structure followed by a change in the physical properties of the soil is a commonly reported effect associated with the use of heavy vehicles in military training (Iverson et al., 1981; Prose, 1985; Braunack, 1986; Thurow et al., 1993; Milchunas et al., 1999), forestry (Hatchell et al., 1970), and agriculture (Voorhees et al., 1986; Alakukku and Elonen, 1995). The purpose of this study was to investigate the effect of soil disturbance as a result of military activity on measures of soil carbon and nitrogen at Fort Benning which was established by the U.S. military in 1918 near Columbus, Georgia. Military activities at Fort Benning that can potentially result in soil disturbance include infantry, artillery, wheeled, and tracked vehicle training.

2. Site description

The land area at Fort Benning is approximately 73,500 ha and the annual number of troops on-site ranges between 18,000 and 23,000. Land use/land cover at the site is approximately 83% forest, 10% barren or developed land, 6% herbaceous grasslands, and 1% water (Jones and Davo, 1997). Based on the climatological normals (1971–2000) from the Columbus Municipal Airport, the mean annual temperature is 18.4 °C and mean annual precipitation is 123 cm. Most of the soils at this site are highly weathered Ultisols derived from coastal plain or alluvial deposits. Sands and loamy sands are common on upland sites while sandy loams and sandy clay loams are frequently found in the valleys and lands bordering streams and tributaries.

3. Methods

3.1. Characterization of disturbance categories

Study sites were classified into different disturbance categories with the help of personnel at Fort Benning who were familiar with current and past military land use. The five disturbance categories at Fort Benning included: (1) reference areas, (2) light military use, (3) moderate military use, (4) heavy military use, and (5) remediated sites. Information on the floristic composition of sites in each disturbance category can be found elsewhere (Dale et al., 2002). Stand age was determined by tree coring. The various disturbance categories are described briefly in the following paragraphs.

3.1.1. Reference sites

The definition of reference sites at Fort Benning is, at best, difficult because nearly all of the reservation has been used in the past for some type of military activity and prior to 1940 most of the area was extensively farmed. In this context, reference sites were defined as pine forests with only minimal current infantry training. The reference sites included a 28-year-old mixed pine stand, a 68-year-old longleaf pine (*Pinus palustris*) stand, and a 74-year-old longleaf pine stand. There was 75–100% ground cover by vegetation at the three reference sites. Soils at each reference site had a recognizable A-horizon that ranged from approximately 1 to 10 cm in depth (depending on location). Reference sites ranged from 117 to 156 m in elevation.

3.1.2. Light military use sites

This category included three sites used only for infantry foot training: a 65-year-old longleaf pine stand, a 77-year-old longleaf pine stand, and a 109-year-old mixed pine stand. There was 75–100% ground cover by vegetation at the three light use sites. Soils at these sites also had recognizable A-horizons that ranged from approximately 1 to 10 cm in depth (depending on location). Light use military sites ranged from 115 to 176 m in elevation.

3.1.3. Moderate military use sites

Moderate military use sites were selected adjacent to sites that had been used for tracked vehicle training. The moderate use category included sites with no forest overstory and about 25–50% bare ground. Generally, there was no recognizable A-horizon in soils at moderate use sites. These sites may be in various stages of recovery from past impacts of tracked vehicle traffic. Moderate use sites ranged from 172 to 277 m in elevation.

3.1.4. Heavy military use sites

This category included three sites recently used for tracked vehicle training. The sites included no overstory vegetation and typically had ≥95% bare ground. There was no recognizable A-horizon in soils at heavy military use sites. Elevation at the sites ranged from 162 to 223 m.

3.1.5. Remediated sites

Remediated sites included two recently planted longleaf pine plantations on highly disturbed soils. The first stand was 2–5 years old and the second stand was 8–10 years old. Bare ground ranged from 0 to 50% at the remediated sites, and the soils had not yet developed a recognizable A-horizon. Elevation at the two sites ranged from 173 to 185 m. No current military training was permitted in the remediated sites.

3.2. Soil sampling and sample preparation

Surface mineral soil samples were collected in September and October 1999, using a hand soil probe (2 cm diameter) to a mean (±S.D.) depth of 22 (±2) cm along a transect at each study site. Each sampling transect was 60 m long and soil samples were taken at five equally spaced (15 m) sampling points along the transect. Transects were placed at three study sites in each disturbance category, except remediated sites. There were only two sites in the remediated site category.

Soil samples were air-dried to a constant mass at room temperature (21 ± 1 °C) in a laboratory with a continuously operating dehumidifier. The dry soil samples were crushed using a rubber mallet and passed through a 2 mm sieve to remove gravel and coarse organic debris. The coarse fraction (>2 mm), when present, was weighed. Crushed and sieved soils were stored in airtight glass jars. A subsample of soil (<2 mm) from each jar was ground and homogenized using a mortar and pestle prior to elemental analysis.

3.3. Analysis of particulate organic matter

Organic matter in the surface soil samples was separated into particulate organic matter (POM) and mineral-associated organic matter (MOM) by wet sieving methods (Cambardella and Elliott, 1992). For this analysis, equal mass portions of soil from each sampling point along a transect were composited. Twenty grams of the composited soil were dispersed by shaking overnight in a 100 ml solution of sodium hexametaphosphate (5 g l^{-1}). The mixture was then sieved through a 0.053 mm sieve. The particulate organic matter (≥0.053 mm) was recovered by back-washing the sieve and filtration (Whatman filter paper #541). The MOM (<0.053 mm) was recovered by evaporation. Both fractions were weighed after oven drying (65 °C) and ground to a fine powder using a mortar and pestle prior to elemental analysis.

POM consisted of free organic debris in the soil, some larger fragments of organic matter (≥0.053 mm) released by dispersion of soil aggregates, and larger fragments of charcoal. MOM included organic matter bound to silt and clay size particles, some smaller fragments of organic matter (<0.053 mm) released by dispersion of soil aggregates, and charcoal.

3.4. Analysis of refractory soil carbon

Controlled ground fires are a common management practice used at Fort Benning to reduce understory vegetation, promote establishment of longleaf pine, and reduce the risk of wildfires by lowering the load of combustible materials on the forest floor. These fires produce charcoal, a highly refractory form of soil carbon. Refractory soil carbon in each

POM fraction was evaluated using acid–base digestions in an attempt to correct POM carbon, which is relatively labile, for the presence of refractory soil carbon. Two grams of POM were digested in 20 ml of boiling 1 M hydrochloric acid (HCl), followed by sequenced boiling in 2 M sodium hydroxide and 6 M HCl (each 1 h). Following each digestion, the supernatant acid or base was removed from the settled residue with a pipette. The final residue was recovered by filtration, thoroughly washed with distilled water and oven-dried (65 °C) prior to elemental analysis. A sand–charcoal "standard" was made for this analysis by mixing activated charcoal with dry, ignited beach sand. Replicate samples of the standard (0.82% C) were analyzed for refractory carbon using the same methods as applied to the POM fractions.

3.5. Elemental analysis

Whole soils and different soil fractions were analyzed for total carbon and nitrogen concentrations using a Perkin-Elmer 2400 Series II CHNS/O Analyzer (Perkin-Elmer Analytical Instruments, Norwalk, CT). Tyrosine ($0.597\,\mathrm{g\,C\,g^{-1}}$; $0.077\,\mathrm{g\,N\,g^{-1}}$) and tetraoctadecyl-ammonium bromide ($0.786\,\mathrm{g\,C\,g^{-1}}$; $0.012\,\mathrm{g\,N\,g^{-1}}$) were used as calibration standards. Coefficients of variation associated with repeated analysis of the standards ($n = 89$) were $<2\%$ for carbon and $<8\%$ for nitrogen.

3.6. Calculations and statistics

Stocks of carbon and nitrogen in whole soils, and different fractions of whole soil, were expressed on a centimeter square basis because that unit is most appropriate to the areal coverage of the soil sampling methods. The uncertainties associated with the extrapolation of the data over larger scales (e.g. $>1\,\mathrm{m^2}$) are unknown, particularly for forest soils. Correction factors for the soil volume occupied by large rocks, roots, and boulders have not been developed for the various study sites.

Carbon and nitrogen stocks ($\mathrm{g\,cm^{-2}}$) at each sampling location were calculated as the product of concentration ($\mathrm{g\,C\,g^{-1}}$ or $\mathrm{g\,N\,g^{-1}}$), surface soil bulk density ($\mathrm{g\,cm^{-3}}$), and surface soil increment depth (20 cm). Bulk density ($\mathrm{g\,cm^{-3}}$) was estimated from the air-dried soil mass (g) and the calculated volume ($\mathrm{cm^3}$) of the hand soil probe down to the sampling depth.

Soil carbon in particulate organic matter (g POM-C $\mathrm{g^{-1}}$ soil) or mineral-associated organic matter (g MOM $\mathrm{C\,g^{-1}}$ soil) was calculated by multiplying the dry mass of POM or MOM (g part $\mathrm{g^{-1}}$ soil) by the respective carbon concentration ($\mathrm{g\,C\,g^{-1}}$ part). The fraction of soil carbon in particulate organic matter (F_{pc}) was calculated as

$$(\mathrm{g\,POM\text{-}C\,g^{-1}\,soil}) \div (\mathrm{g\,POM\text{-}C\,g^{-1}\,soil} + \mathrm{g\,MOM\text{-}C\,g^{-1}\,soil})$$

The fraction of soil carbon in MOM (F_{mc}) was calculated as $1 - F_{pc}$. The carbon stock in the surface mineral soil that was associated with particulate organic matter was calculated as the product of F_{pc} and the soil carbon stock ($\mathrm{g\,C\,cm^{-2}}$).

Refractory soil carbon in each POM sample was calculated relative to that in the sand–charcoal "standard" which had an assumed refractory soil carbon fraction of 1. The refractory soil carbon fraction was calculated as: $(R_a/R_b)/R_s$, where R_a denotes the carbon concentration in POM residue after acid–base digestion, R_b represents the carbon concentration in the POM part prior to acid–base digestion, and R_s is the ratio of carbon concentrations in the sand–charcoal standard before and after acid–base digestion (0.66 ± 0.01). Thus, refractory soil carbon in each POM sample was normalized to the sand–charcoal standard, and the fraction of refractory soil carbon was used to correct POM-C as follows:

$$\begin{aligned}&\mathrm{corrected\ mg\,POM\text{-}C\,g^{-1}\,soil}\\&\quad = (\mathrm{mg\,POM\text{-}C\,g^{-1}\,soil})\\&\qquad - (\mathrm{mg\,refractory\,soil\,C\,g^{-1}\,soil})\end{aligned}$$

Differences among data grouped by disturbance category were first tested using analysis of variance (ANOVA). Differences between means were evaluated by calculating Fisher's least significant difference (LSD) (Kirk, 1968). Unless stated otherwise, statistical significance was indicated by $P \leq 0.05$. Variability about each mean was summarized using the coefficient of variation, CV = standard deviation/mean.

4. Results

4.1. Soil carbon and nitrogen concentrations and stocks

Disturbance effects on surface soil bulk density, carbon concentrations and stocks, nitrogen stocks, and soil C:N ratios are presented in Table 1. Disturbance category had a significant effect on surface soil bulk density ($F_{4,65} = 14.9$, $P = 0.001$). Soil density at heavy use, moderate use, and remediated sites was significantly greater than at reference and light use sites. Disturbance category also had a significant effect on surface soil carbon concentrations ($F_{4,65} = 16.8$, $P \leq 0.001$), carbon stocks ($F_{4,65} = 18.5$, $P \leq 0.001$), and nitrogen stocks ($F_{4,65} = 2.8$, $P \leq 0.05$). Carbon concentrations and stocks increased in the following order of disturbance categories: heavy use < moderate use = remediated < light use = reference sites. Patterns in soil nitrogen stocks were more complicated. Reference, light use, moderate use, and remediated sites had similar soil nitrogen stocks. Soil nitrogen stocks at heavy use sites were significantly less than those at light use and moderate use sites (but not different from those at reference and remediated sites). Soil C:N ratios at heavy use and moderate use sites were significantly less than those at reference, light use, and remediated sites (Table 1).

4.2. Carbon and nitrogen in particulate organic matter

Differences in POM and MOM carbon and nitrogen among disturbance categories are presented in Table 2. Disturbance category had no effect on the dry mass of POM or MOM in surface mineral soil. The mean (±S.E.) percentage of POM and MOM in whole soils was, respectively, 82 (±2) and 18 (±2)%. Thus, POM and MOM separations reflected the high sand and low silt-clay content of most soils at Fort Benning.

Table 1
Mean values for surface (0–20 cm) soil bulk density, carbon concentrations, carbon stocks, nitrogen stocks, and soil C:N ratios under various disturbance categories at Fort Benning, Georgia

Variable	Disturbance category				
	RF	LU	HU	MU	RM
Soil bulk density ($g\,cm^{-3}$)	1.25^{a} (0.09)	1.38^{b} (0.10)	1.53^{c} (0.08)	1.51^{c} (0.06)	1.51^{c} (0.06)
C concentration (%)	0.92^{a} (0.35)	0.98^{a} (0.50)	0.17^{b} (0.44)	0.56^{c} (0.46)	0.55^{c} (0.33)
C stock ($mg\,C\,cm^{-2}$)	226^{a} (0.30)	260^{a} (0.42)	53^{b} (0.46)	167^{c} (0.42)	164^{c} (0.30)
N stock ($mg\,N\,cm^{-2}$)	$6.11^{a,b}$ (0.24)	7.70^{a} (0.38)	5.03^{b} (0.39)	7.93^{a} (0.41)	$6.24^{a,b}$ (0.68)
Soil C:N ratio	38.8^{a} (0.38)	34.0^{a} (0.18)	12.8^{b} (0.68)	21.8^{c} (0.33)	31.7^{a} (0.39)

Coefficients of variation are in parenthesis. Mean values in the same row with different alphabetic superscripts are significantly different. Sample size is 15 for each mean (except those under RM where $n = 10$). RF: reference site; LU: light military use; HU: heavy military use; MU: moderate military use; RM: remediated site.

Table 2
Mean sand fraction, and fraction of carbon and nitrogen in particulate organic matter (POM) and mineral-associated organic matter (MOM) in surface (0–20 cm) soils from different disturbance categories at Fort Benning, Georgia

Variable	Disturbance category				
	RF	LU	HU	MU	RM
Sand fraction ($\geq 53\,\mu m$)	0.77^{a} (0.09)	0.83^{a} (0.05)	0.83^{a} (0.10)	0.85^{a} (0.07)	0.83^{a} (0.01)
Fraction POM-C	0.44^{a} (0.10)	$0.43^{a,b}$ (0.14)	0.19^{c} (0.47)	0.18^{c} (0.35)	$0.31^{b,c}$ (0.08)
Fraction MOM-C	0.56^{a} (0.08)	$0.57^{a,b}$ (0.10)	0.81^{c} (0.11)	0.82^{c} (0.08)	$0.69^{b,c}$ (0.03)
Fraction POM-N	0.28^{a} (0.30)	0.28^{a} (0.53)	0.00^{b} (0.00)	0.04^{b} (1.26)	0.05^{b} (0.70)
Fraction MOM-N	0.72^{a} (0.12)	0.72^{a} (0.20)	1.00^{b} (0.00)	0.96^{b} (0.06)	0.95^{b} (0.04)

Coefficients of variation are in parenthesis. Means in the same row with different alphabetic superscripts are significantly different. Sample size is 3 for each mean (except those under RM where $n = 2$). RF: reference site; LU: light military use; HU: heavy military use; MU: moderate military use; RM: remediated site.

Table 3
Soil carbon and nitrogen in particulate organic matter (POM) and mineral-associated organic matter (MOM), refractory soil carbon (determined by acid–base digestion), and corrected POM-C (adjusted for refractory soil carbon) in surface (0–20 cm) mineral soils from different disturbance categories at Fort Benning, Georgia

Variable	Disturbance category				
	RF	LU	HU	MU	RM
POM-N (mg POM-N g^{-1} soil)	0.083^{a} (0.32)	0.072^{a} (0.54)	0.0^{b}	0.016^{b} (1.32)	0.009^{b} (0.75)
MOM-N (mg MOM-N g^{-1} soil)	0.21^{a} (0.10)	$0.20^{a,b}$ (0.45)	0.08^{c} (0.28)	0.27^{a} (0.21)	$0.15^{b,c}$ (0.04)
Uncorrected POM-C (mg POM-C g^{-1} soil)	4.25^{a} (0.34)	4.11^{a} (0.24)	0.35^{b} (0.54)	1.15^{b} (0.44)	1.51^{b} (0.21)
Refractory C (fraction)	0.36^{a} (0.19)	0.37^{a} (0.21)	0.67^{a} (0.57)	0.37^{a} (0.60)	0.33^{a} (0.14)
Corrected POM-C (mg POM-C g^{-1} soil)	2.67^{a} (0.24)	2.62^{a} (0.28)	0.16^{b} (1.21)	0.76^{b} (0.63)	1.02^{b} (0.28)
MOM-C (mg MOM-C g^{-1} soil)	5.35^{a} (0.27)	5.70^{a} (0.39)	1.42^{b} (0.32)	4.96^{a} (0.02)	$3.34^{a,b}$ (0.10)
Refractory C (mg C g^{-1} soil)	1.58^{a} (0.53)	1.49^{a} (0.32)	0.19^{b} (0.48)	0.39^{b} (0.40)	0.49^{b} (0.07)

Coefficients of variation are in parenthesis. Means in the same row with different alphabetic superscripts are significantly different. Sample size is 3 for each mean (except those under RM where $n = 2$). RF: reference site; LU: light military use; HU: heavy military use; MU: moderate military use; RM: remediated site.

Disturbance category had a significant effect on the fraction of soil carbon in POM and MOM ($F_{4,9} = 11.4$, $P \leq 0.01$) and the fraction of soil nitrogen in POM and MOM ($F_{4,9} = 7.6$, $P \leq 0.01$). There was no difference between reference and light use sites for the fraction of POM-C, fraction of MOM-C, fraction of POM-N, and fraction of MOM-N (Table 2). The fraction of whole soil carbon in particulate organic matter was significantly less at heavy use, moderate use, and remediated sites compared to reference sites. The fraction of whole soil nitrogen in particulate organic matter exhibited a similar trend.

Measurements of POM-C, MOM-C, POM-N, and MOM-N in soils from different disturbance categories are summarized in Table 3. Levels of POM carbon ($F_{4,9} = 12.6$, $P \leq 0.001$) and nitrogen ($F_{4,9} = 7.2$, $P \leq 0.01$) were similar at reference and light use military sites but significantly less at heavy use, moderate use, and remediated sites. Trends for carbon and nitrogen in MOM were more complicated but, generally, levels of MOM-C ($F_{4,9} = 5.7$, $P \leq 0.05$) and MOM-N ($F_{4,9} = 5.4$, $P \leq 0.05$) at heavy use sites were significantly less than those in soils from reference, light use, and moderate use sites.

4.3. Refractory soil carbon

There was a tendency toward a greater fraction of refractory soil carbon at heavy use sites (Table 3), but the effect of disturbance category was not statistically significant and the mean (±S.E.) fraction of refractory soil carbon over all 14 transects was 0.43 (±0.22). Disturbance effects on amounts of refractory soil carbon (mg C g^{-1} soil) were statistically significant ($F_{4,9} = 5.9$, $P \leq 0.05$). More refractory soil carbon was found in soils from reference and light use sites than in soils from heavy use, moderate use, and remediated sites. Amounts of carbon in particulate organic matter were reduced when corrected for refractory soil carbon. The effect of disturbance category on corrected POM carbon ($F_{4,9} = 13.5$, $P \leq 0.001$) was statistically significant. Corrected POM carbon was significantly less at heavy use, moderate use, and remediated sites relative to reference and light use sites (Table 3). There was no detectable concentration of refractory soil nitrogen in any of the soils analyzed for refractory soil carbon.

4.4. Trends along a chronosequence

There was a weak positive correlation ($r = 0.47$, $P \leq 0.01$; $n = 40$) between whole soil carbon stocks and forest stand age. The forest stands included reference, light military use, and remediated sites. Relationships between stand age and measures of labile soil carbon (i.e. the fraction of forest soil POM carbon and POM carbon stocks corrected for refractory soil carbon) were much stronger than that for whole soil carbon (Fig. 1). The fraction of whole soil carbon in POM increased with stand age ($r^2 = 0.80$, $P \leq 0.01$) as did POM carbon stocks corrected for refractory soil carbon ($r^2 = 0.89$, $P \leq 0.01$).

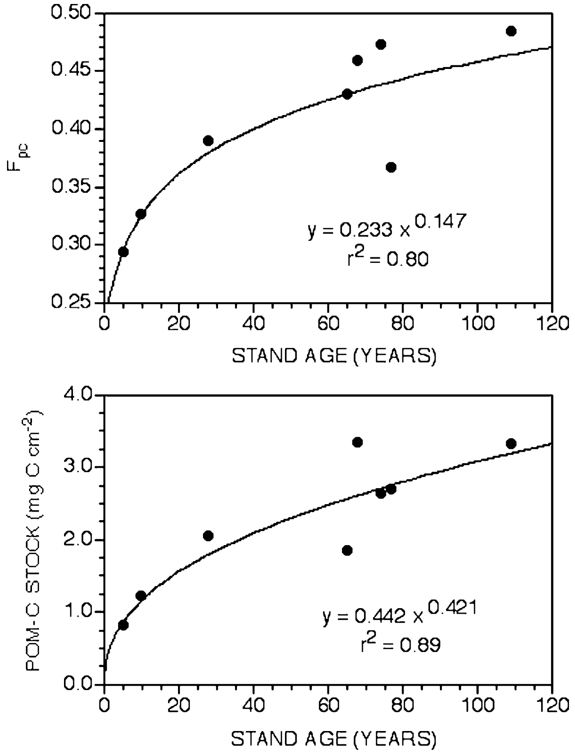

Fig. 1. Above: relationship between the fraction of POM carbon in surface (0–20 cm) mineral soil (F_{pc}) and forest stand age along a chronosequence of reference, light military use, and remediated sites at Fort Benning, Georgia. Below: relationship between the stock of POM carbon (corrected for refractory soil C) in surface mineral soil and forest stand age at the same sites.

5. Discussion

5.1. Comparisons of reference and light use sites

Reference and light use sites at Fort Benning were similar with respect to all measured properties, except for surface soil bulk density. Greater surface soil density was found at light military use sites that had a history of infantry training. Prior studies indicate that human trampling and encampments can result in increased surface soil bulk density as well as declines in forest litter and mineral soil carbon and nitrogen concentrations (Trumbull et al., 1994; Bhuju and Ohsawa, 1998). In the areas examined at Fort Benning, infantry training alone appeared to have a minimal effect on measures of soil quality aside from surface soil density (Table 1). This difference did not appear to adversely impact vegetation in light military use sites (Dale et al., 2002).

5.2. Sites with diminished soil quality

Heavy use, moderate use, and remediated sites at Fort Benning had similar surface soil bulk densities, which were significantly greater than those at reference and light use sites. Studies of military training on dry sandy soils indicate that surface soil compaction caused by heavy, tracked vehicles can persist for decades (Iverson et al., 1981). Soil compaction can change the properties of soil pores affecting infiltration capacity (Iverson et al., 1981), the accessibility of organic matter to microorganisms, the decomposition rate of soil organic matter, and soil nitrogen availability (Breland and Hansen, 1996). Soil disturbance by military traffic at Fort Benning reduces microbial biomass and alters soil microbial community composition (Peacock et al., 2001). Thus, soil compaction has a potentially overall adverse impact on soil quality.

The persistence of soil compaction depends on both clay content and soil moisture status at the time of disturbance. Wet soils are more prone to compaction by heavy vehicle traffic, but shrink/swell cycles in soils with significant clay content can reduce soil compaction over time (Thurow et al., 1993). Similarities in surface soil densities among upland heavy use, moderate use, and remediated sites at Fort Benning indicate that soil compaction is a potential long-term effect of disturbance by heavy vehicle traffic.

Heavy use sites at Fort Benning also had the lowest levels of labile soil carbon as indicated by corrected POM carbon. The highest fraction of refractory soil carbon and the lowest soil carbon concentrations and soil C:N ratios were also found in soils at heavy use sites. Soils in each of the disturbance categories surveyed at Fort Benning had a high (≥77%) sand fraction. Soils with an extremely high sand content are not favorable for the physical protection of organic matter through soil aggregate formation which would tend to preserve labile carbon following soil disturbance.

5.3. Burning and refractory soil carbon

One long-term effect of prescribed burning in forestry is the production of charcoal fragments that are resistant to microbial decomposition. Because of its refractory nature, charcoal-C has a much longer turnover time in surface mineral soils than carbon in POM and MOM. The turnover time of POM carbon is measured in years, and appears to be temperature dependent (Garten and Wullschleger, 2000), while that of MOM carbon is measured in decades or longer (Garten and Ashwood, 2002). Refractory soil carbon was highest at reference and light use sites (forest stands) where prescribed burning occurs on an approximate 3-year cycle at Fort Benning. Although we were unable to provide a precise chemical analysis of refractory soil carbon, it does appear to have the same chemical properties as charcoal-C in acid–base digestions.

5.4. Evidence for recovery of soil quality

Analysis of soil C:N ratios, as well as soil carbon concentrations and stocks at Fort Benning indicated some recovery of soil quality at moderate use and remediated sites relative to heavy use sites, despite a persistence of soil compaction at remediated sites. Soil carbon stocks, nitrogen stocks, MOM-C, MOM-N, and soil C:N ratios were significantly greater in soils from moderate use sites than in soils at heavy use sites. The land use history of the moderate use sites was unknown, but they were in an early stage of successional recovery from the bare ground characteristic of heavy use sites. Whole soil carbon concentrations and stocks, and soil C:N ratios, were also significantly greater at remediated sites relative to heavy use sites.

Soil disturbance and changes in management practice affect the amount of POM in surface mineral soils and it has been suggested that POM carbon and nitrogen may be valuable long-term indicators of changes in soil quality (Sikora et al., 1996). There was a trend for greater POM carbon and nitrogen at remediated sites relative to heavy use sites, but the differences were not statistically significant due to high variability and a small sample size within various disturbance categories. Although some recovery in POM carbon was indicated in a comparison of remediated and heavy use sites, trends along the chronosequence of forest stands (Fig. 1) indicated that recovery of POM carbon may be a slow process in the sandy soils common to Fort Benning. This is consistent with reports indicating a slow rate of mineral soil carbon accumulation in warm, coarse textured surface mineral soils characteristic of large areas in the southeastern United States (Richter et al., 1999).

Considering the indicated persistence of soil compaction, the potential slow accumulation of POM carbon in sandy soils, and reductions in soil microbial biomass (Peacock et al., 2001), heavy military use sites may be slow to recover soil quality without human intervention. The recovery of microbial biomass (which supplies glues and mucilaginous compounds critical for soil aggregate formation) in severely disturbed soils is apparently driven by inputs of fresh organic matter to the mineral soil horizons (Insam and Domsch, 1988). Most of the labile soil carbon at heavy use sites on Fort Benning has been significantly depleted by organic matter decomposition in the absence of soil carbon inputs from aboveground and belowground biomass. Changes in soil quality at sites subject to heavy military use might be reversed through conventional tillage and the establishment of fast growing perennial vegetation. Measurements of soil carbon and nitrogen are ecological indicators that can be used by military land managers to identify changes in soil from training activities and to rank training areas on the basis of soil quality.

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APPENDIX B.

Land Cover Differences in Soil Carbon and Nitrogen at Fort Benning, Georgia

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Land Cover Differences in Soil Carbon and Nitrogen at Fort Benning, Georgia

January 2004

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Environmental Sciences Division

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LAND COVER DIFFERENCES IN SOIL CARBON AND NITROGEN AT FORT BENNING, GEORGIA

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ABSTRACT

Land cover characterization might help land managers assess the impacts of management practices and land cover change on attributes linked to the maintenance and/or recovery of soil quality. However, connections between land cover and measures of soil quality are not well established. The objective of this limited investigation was to examine differences in soil carbon and nitrogen among various land cover types at Fort Benning, Georgia. Forty-one sampling sites were classified into five major land cover types: deciduous forest, mixed forest, evergreen forest or plantation, transitional herbaceous vegetation, and barren land. Key measures of soil quality (including mineral soil density, nitrogen availability, soil carbon and nitrogen stocks, as well as properties and chemistry of the O-horizon) were significantly different among the five land covers. In general, barren land had the poorest soil quality. Barren land , created through disturbance by tracked vehicles and/or erosion, had significantly greater soil density and a substantial loss of carbon and nitrogen relative to soils at less disturbed sites. We estimate that recovery of soil carbon under barren land at Fort Benning to current day levels under transitional vegetation or forests would require about 60 years following reestablishment of vegetation. Maps of soil carbon and nitrogen were produced for Fort Benning based on a 1999 land cover map and field measurements of soil carbon and nitrogen stocks under different land cover categories.

Key words: soil quality, soil nitrogen, soil carbon, particulate organic matter, ecosystem recovery, land management, soil management, military land

1. INTRODUCTION

Military land managers are faced with the challenge of using a fixed amount of land for the purpose of training and troop readiness. Ideally, this mission must be accomplished in a manner that promotes the sustainability of ecosystems and the maintenance of soil quality; otherwise, the military mission may be compromised by a degraded landscape and conflicts with regulatory agencies. Organic matter (or soil carbon) and nitrogen availability are critical components of soil quality (Doran and Parkin, 1996). Numerous studies (e.g., Compton et al., 1998; Compton and Boone, 2000; Garten and Ashwood, 2002) indicate that there are land cover differences in processes related to soil nitrogen availability. However, associations between land cover type and other measures of soil quality have been less well studied.

Land cover can be readily classified on the basis of remote sensing data that are increasingly available at multiple spatial scales. Land cover is also amenable to management for enhancing or restoring soil quality on degraded land. Associations between land cover and soil quality could be valuable for local and regional assessments of how land cover change potentially reflects changes in soil quality. However, if land cover characterization is to be used effectively in soil management, then a better understanding of soil quality under different land covers is needed. The objectives of this limited investigation was to determine how measures of soil quality differ among five land cover types at Fort Benning, Georgia, and to develop maps of soil carbon and nitrogen stocks across the installation based on field measurements.

2. METHODS

Fort Benning is located near Columbus, Georgia. Current land cover is 49% mixed forest, 25% deciduous forest, 10% barren or developed land, 7% evergreen forest, 6% herbaceous grasslands, 2% shrub land, and 1% water (Jones and Davo, 1997). Most soils at the site are highly weathered Ultisols derived from coastal plain or alluvial deposits. Sands and loamy sands are common on upland sites while sandy loams and sandy clay loams are found in valleys and riparian areas. Human activities that potentially produce soil disturbance include infantry, artillery, and wheeled and tracked vehicle training, as well as forest management practices. Further details on the biology, geology, physical setting, and history of Fort Benning are presented elsewhere (Jones and Davo, 1997).

Forty-one sampling sites were selected using on-the-ground surveillance and a Geographic Information System that included five major land cover types at Fort Benning: (1) deciduous forest, (2) mixed forest, (3) evergreen forest or plantation, (4) transitional land, and (5) barren land. Deciduous forests were comprised of various hardwoods (*Quercus, Carya, Acer, Liquidambar, Liriodendron, Betula, Fagus*) while evergreen forests included stands of longleaf (*Pinus palustris*), loblolly (*P. taeda*), shortleaf (*P. echinata*), slash (*P. elliotti*) and mixed pines. The mixed forest type included both pine/hardwood and hardwood/pine stands. Transitional lands were occupied by herbaceous annual and perennial vegetation and no overstory trees.

Barren land included training sites and erosional areas more than 95% devoid of surface vegetation.

Each site was treated as a single sampling unit and eight to nine sites were sampled within each land cover category in March, 2000. Three soil samples were collected at each site. The first sample (0-20 cm of mineral soil) was collected by hammering a PVC pipe (5.1 cm inner diameter) into the soil. Two remaining mineral soil samples (0-40 cm) were collected in butyrate plastic tubes using a soil recovery probe (2.54 cm inner diameter) with hammer attachment (AMS, American Falls, ID). Soil compression was minimal in the dominantly coarse textured soils at the study sites. When present, the O-horizon was sampled directly above each point sampled with the soil probe.

Mineral soil samples taken with the soil probe were cut into 10 cm depth increments and equivalent depth increments from each site were composited. Soil density (g cm^{-3}) was estimated from the dry mass of soil in each depth increment and the calculated increment volume. O-horizon samples were oven dried (65 °C) and mineral soil samples were air-dried to a constant mass. Mineral soil samples were crushed with a rubber mallet to pass a 2 mm sieve prior to elemental analysis. Carbon and nitrogen concentrations in dry, powdered samples were determined using a Perkin-Elmer 2400 Series II CHNS/O Analyzer (Perkin-Elmer Analytical Instruments, Norwalk, CT) and a LECO CN-2000 (LECO Corporation, St. Joseph, MI).

Soil samples collected with the soil probe were used to determine the depth profile and amounts of soil carbon and nitrogen under different land cover types. Carbon and nitrogen stocks (g m^{-2}) in the O-horizon were calculated as the product of concentration (g C g^{-1} or g N g^{-1}) and dry mass per unit area (g m^{-2}). Carbon and nitrogen stocks (g m^{-2}) in each mineral soil increment were calculated as a product of concentration (g C g^{-1} or g N g^{-1}), soil density (g cm^{-3}), and increment length (cm).

Soil samples collected using PVC pipes were used for determination of potential net soil nitrogen mineralization and nitrification in 12-week aerobic laboratory incubations using methods described elsewhere (Garten and Ashwood, 2002). Portions of these samples were also physically separated into particulate organic matter (POM) and mineral-associated organic matter (MOM) (Cambardella and Elliott, 1992) using methods described in other papers (Garten and Ashwood, 2002; Garten et al., 2003). The total soil carbon stock (to a depth of 20 cm) was subdivided among POM (corrected for refractory soil carbon), MOM, and refractory soil carbon. Refractory soil carbon in POM was evaluated using acid-base digestions (see Garten et al., 2003).

Differences among data grouped by land cover category were tested using analysis of variance (ANOVA). Differences between means were evaluated using Fisher's least significant difference (LSD). Unless stated otherwise, statistical significance was indicated by $P \leq 0.05$. The standard error (±SE) was used to summarize the variability about each mean.

A 1999 land cover map was obtained for Fort Benning from the University of Georgia's Natural

Resources Spatial Analysis Laboratory. The land cover categories, identified on the basis of remote sensing (LANDSAT), were water, bare ground, non-forest vegetation, pine forest, mixed forest, and deciduous forest. For the purpose of the spatial analysis, we assumed that non-forest vegetation corresponded to the land cover category designated “transitional vegetation”. Measured mean values for soil carbon and nitrogen stocks in the O-horizon and the mineral soil (0-20 cm) were assigned to the different land cover categories to produce maps illustrating an hypothesized spatial distribution of soil carbon and nitrogen across Fort Benning.

3. RESULTS AND DISCUSSION

3.1 SOIL DENSITY

At all increment depths, soil density was significantly greater under barren land than under deciduous or mixed forests on Fort Benning (Table 1). Studies of military training on dry sandy soils indicate that surface soil compaction caused by heavy, tracked vehicles can persist for decades (Iverson et al., 1981). Heavy machinery is also used to harvest and establish pine plantations. Land use that promotes soil compaction is concentrated on those land cover categories that have the highest soil densities. Barren soils have been created primarily through disturbance associated with heavy, tracked vehicles. Transitional areas include grassy fields that are maintained for vehicle maneuvers, parachute jump zones, and areas in early stages of secondary succession following forest clearing. Deciduous and mixed forests are subject to low-intensity impacts from infantry (foot or dismounted) training, and their large spatial coverage (74% of the total land area) probably "dilutes" human activities that could promote soil compaction throughout Fort Benning.

3.2 SOIL NITROGEN AVAILABILITY

Extractable soil ammonium, at the beginning of aerobic laboratory incubations, was significantly greater under mixed and deciduous forest stands than under barren land ($F_{4,36} = 2.8$; $P \leq 0.05$). In contrast, extractable soil nitrate was significantly greater under barren land than under forests ($F_{4,36} = 4.7$; $P \leq 0.01$). Concentrations of extractable soil ammonium and nitrate under transitional vegetation were intermediate between those for barren soils and soils under mixed and deciduous forest cover (Table 2).

Potential net nitrogen mineralization ($F_{4,36} = 4.8$; $P \leq 0.01$) and nitrification ($F_{4,36} = 3.5$; $P \leq 0.05$) in surface mineral soils differed among the various land cover types (Table 3). Variation in soil nitrogen transformations at Fort Benning was complex, but the results indicated greater soil nitrogen availability under deciduous forests, mixed forests, and transitional vegetation than under evergreen forests and barren land. Most of the potential net nitrogen mineralization in laboratory incubations terminated in nitrate production, however there was a pronounced lag in the onset of net nitrification under forest soils that was not observed in soils under barren land and transitional vegetation. High concentrations of extractable soil nitrate in barren soils were consistent with the absence of a time lag in the onset of net nitrification, and may indicate a pool of unused nitrogen that originates from atmospheric deposition.

Table 1. Mean (±SE) soil density (g cm^{-3}) under different land cover categories at Fort Benning, Georgia*

Soil depth (cm)	Land cover category					F-value†
	Barren	Transitional land	Evergreen forest	Mixed forest	Deciduous forest	
0-10	1.64^{a} ±0.02	1.37^{b} ±0.06	1.32^{b} ±0.06	1.16^{c} ±0.04	1.10^{c} ±0.04	18.5 ($P < 0.001$)
10-20	1.71^{a} ±0.03	1.60^{ab} ±0.05	1.43^{bc} ±0.10	1.34^{c} ±0.06	1.34^{c} ±0.05	6.1 ($P < 0.001$)
20-30	1.72^{a} ±0.03	1.61^{ab} ±0.08	1.52^{bc} ±0.09	1.36^{bc} ±0.08	1.41^{c} ±0.04	4.4 ($P < 0.01$)
30-40	1.68^{a} ±0.03	1.57^{ab} ±0.09	1.60^{ab} ±0.06	1.39^{b} ±0.08	1.47^{b} ±0.06	2.7 ($P < 0.05$)

* Means in the same row with different alphabetic superscripts are significantly different
† degrees of freedom (df) = 4,36 for each F-value, except 30-40 cm where df = 4,33

Table 2. Mean (±SE) concentrations (μg N g^{-1} soil) of extractable (2 M KCl) ammonium- and nitrate-N from surface (0-20 cm) mineral soil samples under different land cover categories at Fort Benning, Georgia*

Form of nitrogen	Land cover category					F-value
	Barren	Transitional land	Evergreen forest	Mixed forest	Deciduous forest	
NO_3-N	0.68^{a} ±0.23	0.38^{ab} ±0.07	0.13^{b} ±0.03	0.18^{b} ±0.04	0.13^{b} ±0.03	4.7 ($P < 0.01$)
NH_4-N	0.37^{a} ±0.16	1.48^{ab} ±0.36	1.32^{ab} ±0.22	2.41^{b} ±0.74	2.12^{b} ±0.60	2.8 ($P < 0.05$)
Inorganic N	1.05^{a} ±0.32	1.85^{a} ±0.35	1.45^{a} ±0.23	2.58^{a} ±0.77	2.26^{a} ±0.61	1.6

* Means in the same row with different alphabetic superscripts are significantly different

Table 3. Mean (±SE) potential net soil nitrogen mineralization (μg N g^{-1} soil) during a 12 week aerobic laboratory incubation and potential net nitrification (μg N g^{-1} soil) during the first six weeks (phase 1) and the second six weeks (phase 2) of aerobic laboratory incubations of surface (0-20 cm) mineral soil*

N production (μg N g^{-1} soil)	Land cover category					F-value
	Barren	Transitional land	Evergreen forest	Mixed forest	Deciduous forest	
Net soil N mineralization	1.79^{a} ±1.01	9.93^{bc} ±1.97	5.41^{ab} ±2.25	11.08^{bc} ±2.77	12.83^{c} ±1.80	4.8 ($P < 0.01$)
Net nitrification (phase 1)	1.01^{a} ±0.43	4.51^{b} ±1.45	1.05^{a} ±0.85	0.84^{a} ±0.53	1.43^{a} ±0.35	3.5 ($P < 0.05$)
Net nitrification (phase 2)	0.81^{a} ±0.70	5.56^{ab} ±1.10	3.42^{a} ±1.25	6.53^{b} ±2.42	9.60^{b} ±2.17	4.0 ($P < 0.01$)

* Means in the same row with different alphabetic superscripts are significantly different

3.3 O-HORIZONS

At Fort Benning, both the dry mass and chemistry of the O-horizon differed significantly among land cover types (Table 4). For nonbarren land cover categories, O-horizon dry mass and nitrogen stocks were greatest under deciduous forests and least under transitional vegetation. The O-horizon C:N ratio was significantly elevated under evergreen and mixed forest stands.

Net soil nitrogen mineralization is affected by litter quality (Scott and Binkley, 1997). Low O-horizon C:N ratios under deciduous forests and transitional land covers may promote greater net soil nitrogen mineralization (Table 3). High O-horizon C:N ratios under evergreen forests may reduce net soil nitrogen mineralization by promoting microbial immobilization of nitrogen.

3.4 MINERAL SOIL CARBON AND NITROGEN

Data from different forest types were combined for a depth profile analysis of mineral soil carbon and nitrogen because mineral soil carbon and nitrogen stocks at Fort Benning were not significantly different among the three forest categories. The depth profiles indicated that creation of barren land by heavy, tracked vehicles and/or erosion, results in a substantial loss of soil carbon and nitrogen (Table 5). For each depth increment examined, soil carbon and nitrogen stocks under barren land were significantly less than those under other land covers. On average, in the surface (0-20 cm) mineral soil, there was more than an 80% loss of C and more than a 60% loss of N under barren land.

Table 4. Mean (±SE) dry mass, carbon and nitrogen concentrations and stocks, and C:N ratios in the O-horizons under different land cover categories at Fort Benning, Georgia*

O-horizon property	Land cover category					F-value†
	Barren	Transitional land	Evergreen forest	Mixed forest	Deciduous forest	
Dry mass ($g\ m^{-2}$)	0.0^{a}	894^{b} ±474	1053^{b} ±123	1152^{bc} ±128	1821^{c} ±193	7.4 ($P < 0.001$)
Carbon (%)	--	18.3^{a} ±2.2	40.1^{b} ±1.5	37.5^{c} ±1.6	30.2^{c} ±1.2	33.9 ($P < 0.001$)
Nitrogen (%)	--	0.54^{a} ±0.05	0.54^{a} ±0.03	0.72^{b} ±0.02	0.79^{b} ±0.06	10.3 ($P < 0.001$)
Carbon stock ($g\ C\ m^{-2}$)	--	136^{a} ±59	413^{b} ±39	422^{bc} ±37	536^{c} ±31	15.4 ($P < 0.001$)
Nitrogen stock ($g\ N\ m^{-2}$)	--	5.3^{a} ±3.0	5.7^{a} ±0.8	8.3^{a} ±1.0	14.9^{b} ±2.4	5.1 ($P < 0.01$)
C:N ratio	--	34.5^{a} ±4.2	76.5^{b} ±4.3	52.1^{c} ±2.3	39.9^{a} ±3.9	25.1 ($P < 0.001$)

* Means in the same row with different alphabetic superscripts are significantly different
† df = 3,27 for each F-value, except for O-horizon dry mass where df = 4,36

Table 5. Mean (±SE) soil carbon and nitrogen stocks as a function of soil depth under different land cover categories at Fort Benning, Georgia

Soil depth (cm)	Soil carbon stock ($g\ C\ m^{-2}$)			Soil nitrogen stock ($g\ N\ m^{-2}$)		
	Barren	Transitional land	Forest	Barren	Transitional land	Forest
0-10	292 ±106	1616 ±188	1658 ±126	21.1 ±4.6	86.8 ±16.1	81.7 ±8.2
10-20	238 ±92	963 ±103	767 ±61	19.3 ±5.2	49.3 ±7.4	40.4 ±4.2
20-30	185 ±68	528 ±61	560 ±61	14.8 ±4.3	38.2 ±7.3	35.3 ±4.0
30-40	148 ±60	364 ±30	425 ±49	14.4 ±3.9	32.1 ±5.1	31.2 ±2.9

Partitioning of soil carbon stocks is important because various soil carbon pools may exhibit different sensitivities to a change in land cover. Soil carbon partitioning also reveals the potential for soil carbon and nitrogen change as a result of disturbance or land cover change. Carbon stocks in POM ($F_{2,37} = 9.9$; $P \leq 0.001$), MOM ($F_{2,38} = 14.2$; $P \leq 0.001$), and refractory ($F_{2,37} = 4.8$; $P \leq 0.05$) soil fractions differed significantly among various land cover types at Fort Benning (Table 6).

Table 6. Mean (±SE) carbon stocks (g C m^{-2}) in particulate organic matter (POM-C), mineral-associated organic matter (MOM-C), a refractory part of POM (REF-C), and surface mineral soil (0-20 cm) under different land covers at Fort Benning, Georgia*

Soil carbon fraction	Land cover category			F-value
	Barren	Transitional land	Forest	
POM-C	73^{a} ±26	462^{b} ±75	474^{b} ±53	9.9 (P < 0.001)
MOM-C	421^{a} ±157	1790^{b} ±236	1716^{b} ±132	14.2 (P < 0.001)
REF-C	34^{a} ±19	297^{b} ±54	247^{b} ±45	4.8 (P < 0.05)
Total	529^{a} ±197	2548^{b} ±257	2433^{b} ±169	19.9 (P < 0.001)

* Means in the same row with different alphabetic superscripts are significantly different

Based on other studies (e.g., see Garten and Ashwood, 2002), carbon in mineral-associated organic matter is expected to have a longer turnover time than carbon in particulate organic matter. Based on acid-base digestions, one-third of the carbon in particulate organic matter was refractory. We are unable to provide a precise chemical analysis of this refractory carbon, but it has chemical properties similar to charcoal (Garten et al., 2003) and probably originates from controlled ground fires that are regularly used for forest management at Fort Benning.

3.5 MAPS OF SOIL CARBON AND NITROGEN STOCKS

Figure 1 illustrates the hypothesized spatial distribution of soil carbon and nitrogen stocks at Fort Benning based on field measurements under different land cover categories and a 1999 land cover map. The highest O-horizon and surface mineral soil carbon stocks tend to occur in areas adjacent to stream drainages (e.g., Wolf Creek, Randal Creek, and Upatoi Creek in the northern part and Oswichee Creek in the southern part of the Fort Benning). Training areas in the northeastern corner and developed areas on the eastern edge of the installation are characterized by the lowest O-horizon carbon and nitrogen stocks. In the present maps (Figure 1), the hypothesized distribution of soil nitrogen is complex with no apparent higher stocks along streams and creeks, but high nitrogen stocks occur under herbaceous cover on transitional land. Nitrogen fixing plants, which are more prevalent in successional herbaceous communities than under forest cover, may contribute to greater soil nitrogen storage on transitional lands.

4. CONCLUSION

Although limited in scope, this small study indicates that military land managers at Fort Benning might infer differences in some measures of soil quality, like soil nitrogen availability and O-horizon properties, based on characterization of land cover. However, other measures (like soil carbon stocks) were similar under non-barren land cover categories indicating that ecosystem type was less useful for inferring some aspects of soil quality than the mere presence of perennial vegetation. Land cover change at barren sites on Fort Benning will probably require human intervention to accelerate recovery of soil quality for ecosystem rehabilitation. Following reestablishment of vegetation on barren sites and at an average rate of soil carbon accumulation beneath perennial vegetation, approximately 33 g C m^{-2} yr^{-1} (Post and Kwon, 2000), it would take about 60 years for soil carbon stocks at barren sites to reach current day levels under transitional vegetation or forest cover (Table 5). This predicted rate of recovery is consistent with an apparent slow accumulation of soil carbon stocks in particulate organic matter along a 100 year old pine chronosequence at Fort Benning, Georgia (Garten et al., 2003).

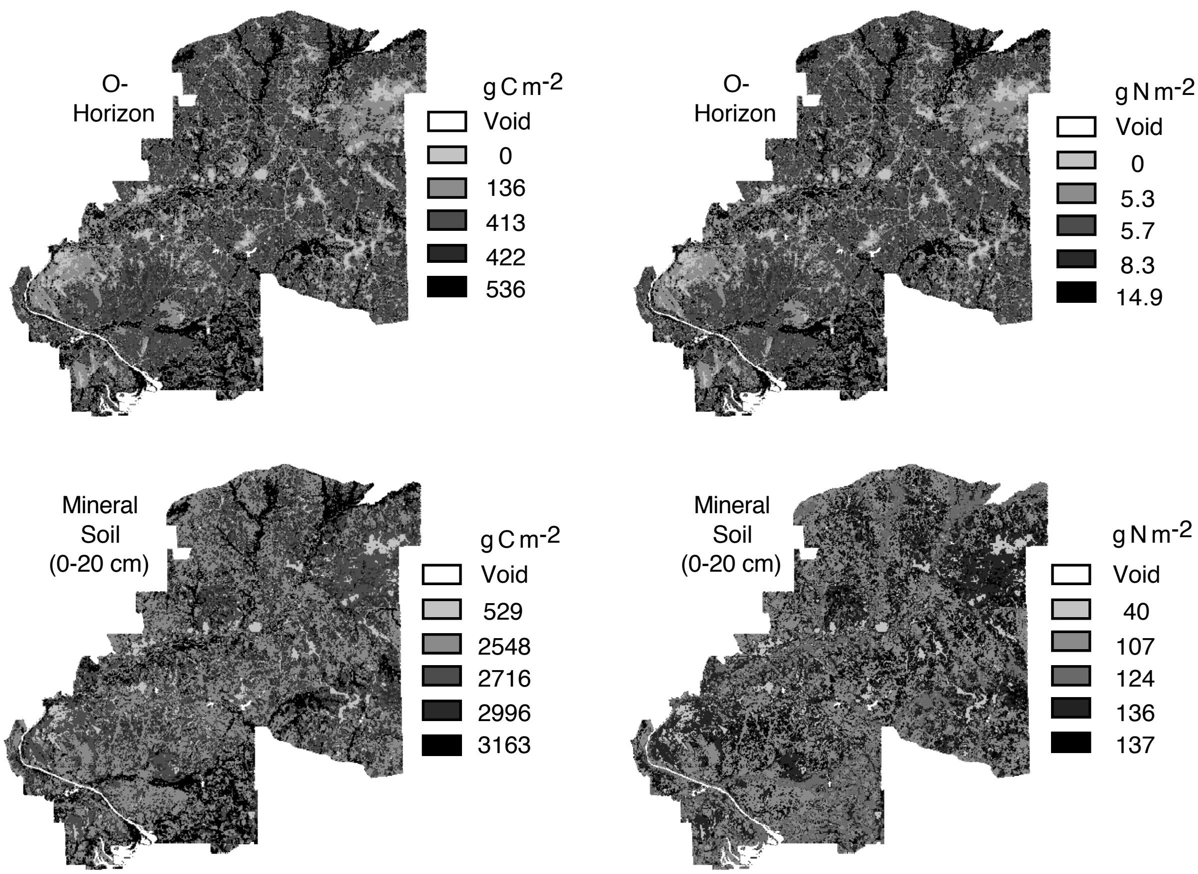

Fig. 1 Hypothesized spatial distribution of soil carbon and nitrogen stocks at Fort Benning based on the assignment of field measurements to an installation land cover map from 1999

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APPENDIX C.

Modeling Soil Quality Thresholds to Ecosystem Recovery at Fort Benning, Georgia, USA

ORNL/TM-2004/41

Modeling Soil Quality Thresholds to Ecosystem Recovery at Fort Benning, Georgia, USA

February 2004

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Environmental Sciences Division

ORNL/TM-2004/41

Environmental Sciences Division

MODELING SOIL QUALITY THRESHOLDS TO ECOSYSTEM RECOVERY AT FORT BENNING, GEORGIA, USA

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SUMMARY

1. The objective of this research was to use a simple model of soil C and N dynamics to predict nutrient thresholds to ecosystem recovery on degraded soils at Fort Benning, Georgia, in the southeastern USA. The model calculates aboveground and belowground biomass, soil C inputs and dynamics, soil N stocks and availability, and plant N requirements. A threshold is crossed when predicted soil N supplies fall short of predicted N required to sustain biomass accrual at a specified recovery rate.

2. Four factors were important to development of thresholds to recovery: (1) initial amounts of aboveground biomass, (2) initial soil C stocks (i.e., soil quality), (3) relative recovery rates of biomass, and (4) soil sand content. Thresholds to ecosystem recovery predicted by the model should not be interpreted independent of a specified recovery rate. Initial soil C stocks influenced the predicted patterns of recovery by both old field and forest ecosystems.

3. Forests and old fields on soils with varying sand content had different predicted thresholds to recovery. Soil C stocks at barren sites on Fort Benning generally lie below predicted thresholds to 100% recovery of desired future ecosystem conditions defined on the basis of aboveground biomass (18000 versus 360 g m^{-2} for forests and old fields, respectively).

4. Calculations with the model indicated that reestablishment of vegetation on barren sites to a level below the desired future condition is possible at recovery rates used in the model, but the time to 100% recovery of desired future conditions, without crossing a nutrient threshold, is prolonged by a reduced rate of forest growth.

5. Predicted thresholds to ecosystem recovery were less on soils with more than 70% sand content. The lower thresholds for old field and forest recovery on more sandy soils are apparently due to higher relative rates of net soil N mineralization in more sandy soils. Calculations with the model indicate that a combination of desired future conditions, initial levels of soil quality (defined by soil C stocks), and the rate of biomass accumulation determines the predicted success of ecosystem recovery on disturbed soils.

Keywords: military land use, ecological thresholds, soil carbon, soil nitrogen, soil N availability, nutrient dynamics, old fields, forests

1. INTRODUCTION

The concept of thresholds has been applied extensively in science, economics, and regulatory law. Although "threshold" has been defined in various ways, it is generally regarded as the point that separates something true from something not true, or the point at which there is a discernible effect or change in behavior in response to a stimulus (Woolf, 1975). In ecology, thresholds have been defined as system discontinuities, which Muradian (2001) recently defined as "sudden change in any property of an ecological system as a consequence of smooth and continuous change in an independent variable." Other researchers have also defined thresholds as a deflection of system response (or an ecological discontinuity) as a consequence of stress -- indicating a breakdown in mechanisms regulating ecosystem function (Romme et al., 1998). Statistical problems associated with the precise quantification of thresholds, have caused some to question the legitimacy of the threshold concept (Slob, 1999), particularly as it might be applied to complex systems, like ecosystems (Van Straalen, 1997).

Many attributes related to the state of ecosystems or ecosystem processes can be described as continuous variables. Recently, there has been an interest in building connections between continuous measures of soil quality (such as soil density or organic matter content) and ecosystem sustainability in both agriculture (Hussain et al., 1999; Lewandowski et al., 1999; Arshad and Martin, 2002) and forestry (Page-Dumroese et al., 2000; Schoenholtz et al., 2000). Studies that have utilized continuous variables to define ecological thresholds have met both success and failure. For example, in pinon-juniper ecosystems, thresholds to soil erosion are related to the extent of ground cover, and erosion and may change dramatically once a threshold in ground cover is crossed (Davenport et al., 1998). On the other hand, Hunter and White (1997), who examined a variety of continuous (and discontinuous) variables, failed to find thresholds that would distinguish when a forest officially becomes "old growth". They were also unable to discern thresholds to forest disturbance. Although the terms can mean different things to different groups of people, "threshold" and "sustainability" seem to be irreversibly linked (Zinck and Farshad, 1995).

Difficulties in identifying and quantifying thresholds using either continuous or discontinuous variables can be attributed to the complexities of natural systems, our limited predictive capabilities in ecology, and the large uncertainties that sometimes surround the quantification of ecosystem properties and processes (Muradian, 2001). There are many unresolved issues surrounding the use of thresholds to predict when a system is sustainable and when it is not sustainable. Page-Dumroese et al. (2000) concluded, based on a study of several indicators of soil quality (i.e., soil C, N, erosion, and cation exchange capacity) in northwestern forests, that generalized thresholds cannot be successfully applied over disparate soil types and that site-specific information is critical to a valid application of thresholds in forest sustainability. In practical terms, the threshold concept requires an ability to ascertain whether the state of a system lies above or below a threshold, or within some acceptable limits that permit sustainability. For example, thresholds for natural resource management can be defined as an upper limit to harvesting individuals and harvests that exceed the limit endanger the sustainability of the population (Lande et al., 1997).

The current research addresses thresholds to ecosystem recovery, which may involve either ecosystem restoration or rehabilitation. Restoration returns an ecosystem to a state that is as similar as possible to its native condition. Ultimately, complete restoration depends on thresholds for recruitment, growth, and mortality of different species as well as the roles that environmental factors play as constraints or as boundaries within which restoration can occur. Complex ecosystems, with multiple interacting species, may have a variety of thresholds. In simple ecosystems, with few interacting species, thresholds to restoration may be more similar to those that define the success or failure of ecosystem rehabilitation (i.e., management toward a desired state not necessarily consistent with, and usually more simple than, the historical native condition). However, ecosystem recovery on severely degraded soils is ultimately related to soil quality and, in particular, nutrient availability. In such systems, seed banks or surrounding vegetation that would serve as a source for colonizing species would have little influence on recovery if critical limiting factors associated with soil quality precluded or inhibited plant growth.

Disturbance of soil physical properties and/or soil structure are commonly reported effects associated with the use of heavy machinery in agriculture (Voorhees et al., 1986; Alakukku and Elonen, 1995), forestry (Hatchell et al., 1970), and military training (Iverson et al., 1981; Prose, 1985; Braunack, 1986; Thurow et al., 1993; Milchunas et al., 1999). At Fort Benning, Georgia, field training with tracked vehicles has resulted in an overall loss of soil quality at some training sites (Garten et al., 2003). Barren, heavily disturbed soils at Fort Benning have negligible O-horizons, lower soil N availability, and lower soil C and N stocks than soils subject to minimal military use (Garten and Ashwood, 2004). In some environments, it has been shown that the effects of soil disturbance by military vehicles can persist for decades (e.g., Iverson et al., 1981). This leads to questions about what factors are at work that might prevent or slow ecosystem recovery following soil disturbance and whether thresholds exist between barren land and the reestablishment of perennial vegetation. The revegetation of barren soils represents an extreme case of ecosystem recovery.

The objective of this research was to use simple models of soil C and N dynamics to predict thresholds to ecosystem recovery from degraded soils at Fort Benning, Georgia, in the southeastern USA. Although ecosystem rehabilitation can be less complex than restoration, especially if monocultures are used, there are likely to be thresholds associated with soil properties, vegetation characteristics, and land management. Of these thresholds, various aspects of soil quality may be the root cause that determines the success of ecosystem rehabilitation. In particular, soil organic matter and soil N availability can be of major importance. Many favorable properties associated with organic matter, such as improved soil structure and greater soil nutrient reserves, argue strongly for the adoption of soil organic matter content (or soil organic C) as one critical metric in defining thresholds to soil quality (Sikora et al., 1996; Seybold et al., 1997). Net primary productivity and standing crop biomass, which partly depend on soil N availability, are associated measures of success in ecosystem rehabilitation. However, the rate of ecosystem recovery to a desired future condition and its degree of success is ultimately constrained by aspects of soil quality.

2. METHODS

2.1 SITE DESCRIPTION

Fort Benning was established by the U.S. military, near Columbus, Georgia, in 1918, and additional land area was added in 1941. The land area at Fort Benning is ≈73,600 ha, and the number of troops onsite ranges between 18,000 and 23,000 annually. Land use prior to acquisition by the U.S. Government was primarily a mixture of agriculture and forestry. Current land cover at the site is ≈49% mixed forest, 25% deciduous forest, 10% barren or developed land, 7% evergreen forest, 6% herbaceous grasslands, 2% shrub land, and 1% water (Jones and Davo, 1997). Mean annual temperature at Fort Benning is 18.3 °C and mean annual precipitation is 130 cm.

Soils at the site are highly weathered Ultisols, mostly of Coastal Plain origin but with some minor inclusion of alluviums derived from the Piedmont ecological unit to the north. Two dominant Coastal Plain ecological units that cover most of the installation are Sand Hills and Upper Loam Hills. The major soil series associated with the former units are Ailey loamy coarse sand, Cowarts loamy sand, Nankin sandy clay loam, Pelion loamy sand, Troup, Troup loamy fine sand, Vaucluse, and Vaucluse sandy loam. Sands and loamy sands are common on upland sites while sandy loams and sandy clay loams are frequently found in valleys and riparian areas. Further details on the biology, geology, physical setting, and history of Fort Benning are available elsewhere (Jones and Davo, 1997).

2.2 CONCEPTUAL MODEL

Wail et al. (1999) have proposed that biogeochemical cycles of C and N connect all the abiotic and biotic components of ecosystems to one another in a holistic way. The concept of the nutrient threshold model (Fig. 1) attempts to summarize these connections in as simple a manner a possible. There are several components to the model that couple soil C and N dynamics with ecosystem biomass dynamics: (1) calculation of aboveground and belowground biomass and dynamics, (2) calculation of soil C inputs and soil C dynamics, (3) calculation of soil N stocks and availability, and (4) calculation of plant N requirements. The nutrient threshold test is represented by a single question, "Are soil N supplies sufficient to meet the N demands of growing biomass on track to a desired future ecosystem condition?" A threshold is crossed when soil N supplies are not sufficient to meet the demands of growing biomass and calculations indicate that the desired future condition, measured in terms of biomass, is not attainable at the specified recovery rate due to resource limitation (i.e., soil N deficiency).

A central concept in the model (Fig. 1) is "desired future condition" because it represents the state against which the success of ecosystem recovery is measured. There are countless attributes that can be used as "metrics" to describe a desired future ecosystem condition. In particular, different target values for aboveground biomass can be associated with different desired future ecosystem conditions. Qualitatively, an observer can see that a forest has more aboveground biomass than an herbaceous field. Quantitatively, we can derive statistics on standing crop biomass for different types of ecosystems and use the mean, the median, or the maximum values as targets for ecosystem recovery. Natural variation in the target value for standing crop biomass and/or net primary production (as indicated by confidence limits about the measure of central tendency) can also be evaluated to determine if an ecosystem is within the expected boundaries for a desired future condition.

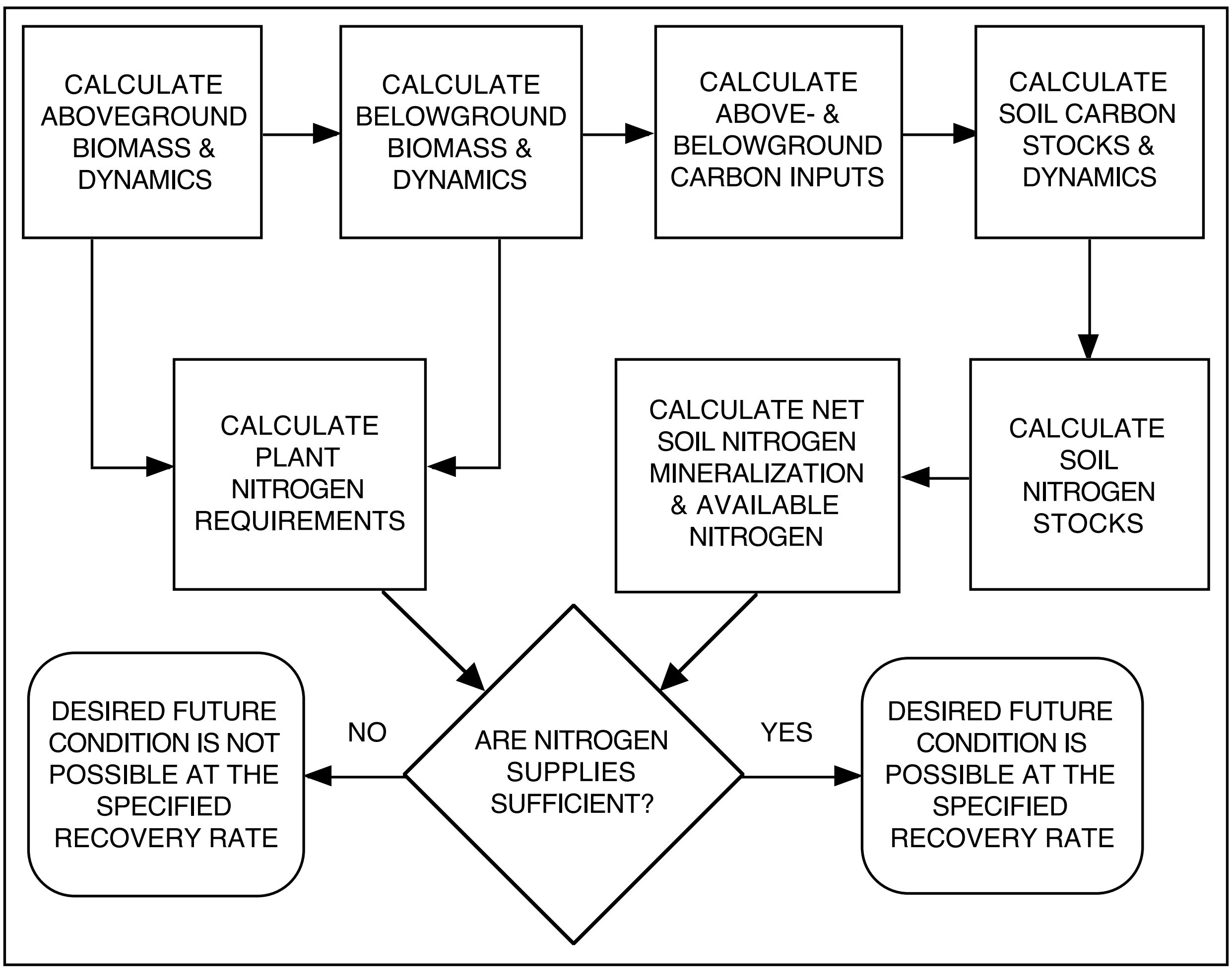

Fig. 1. Conceptual model and steps leading up to the nutrient threshold test in the spreadsheet model. The desired future condition is a target for aboveground standing crop biomass.

The type of resource limiting model described here has been used before, but for different purposes. The model concept is a simplified version of NuCSS (Nutrient Cycling Spreadsheet) that simulates, among other things, forest biomass production and soil N availability (Verburg and Johnson, 2001). NuCSS was developed as a tool for forest nutrient management. Like NuCSS, the model equations were written in a spreadsheet format and they do not include negative feedbacks between soil nutrient supplies and growth of biomass. Incorporation of such feedbacks within the spreadsheet model produces unsolvable circularities in logic. Unlike NuCSS, the model used here does not simulate other element fluxes (including leaching), and has far fewer required model parameters. Although the current model (Fig. 1) is an oversimplification of C and N biogeochemistry, the model predictions are potentially useful for guiding military land management decisions.

2.3 MODEL EQUATIONS

2.3.1 Calculations of Biomass

Relative biomass (%) over time (t) is calculated from the following equation which yields a logistic growth curve:

$$B_t = B_{t-1} + [(B_{t-1}) * (B_r)]*[R - (B_{t-1})/R] \qquad [1]$$

where B_t is relative biomass at time t (%), B_r is the fractional growth rate of biomass (per year), and R is the percent of biomass recovery to the maximum (maximum relative biomass is 100%). Adjustment of R allows for the recovery of biomass to some value less than or more than the target (or desired future condition).

Aboveground biomass (B_a, g m^{-2}) is predicted from:

$$B_a = (B_t/100) * (B_{max}) \qquad [2]$$

where B_{max} is the maximum or target aboveground biomass associated with a future desired ecosystem condition (in this case, the maximum biomass and the target biomass are equivalent). Photosynthetically active biomass (B_f, g m^{-2}) is predicted from: $B_f = B_a * f_L$, where f_L is the fraction of photosynthetically active aboveground biomass (i.e., leaves and green stems).

Belowground biomass (B_b, g m^{-2}) is predicted from:

$$B_b = B_a * R_w \qquad [3]$$

where R_w is the root:shoot ratio for the ecosystem under consideration. Total biomass (B_g, g m^{-2}) is predicted from:

$$B_g = B_a + B_b. \qquad [4].$$

For old fields, the aboveground biomass growth increment (ΔB_a) and the belowground biomass growth increment (ΔB_b) in each year equal, respectively, the aboveground (B_a) and belowground (B_b) biomass. This assumes that herbaceous old field biomass is replaced every year by new growth following tissue senescence prior to the dormant season. For forests, the aboveground biomass growth increment (ΔB_a) for each year was calculated from:

$$\Delta B_a = B_{a(t+1)} - B_{a(t)} \qquad [5].$$

The belowground forest biomass growth increment (ΔB_b) for each year was calculated from:

$$\Delta B_b = B_{b(t+1)} - B_{b(t)} \qquad [6].$$

2.3.2 Soil Carbon Dynamics

Inputs to soil C in the model are derived from both aboveground and belowground biomass. Annual belowground root mortality (R_m, g m^{-2}) is calculated from:

$$R_m = B_b * (1/T_b) \qquad [7]$$

where T_b is the turnover time of roots (years).

Annual soil C inputs (I, g m^{-2}) are calculated from:

$$I = (B_f * C_b) + (R_m * C_b) \qquad [8]$$

where C_b is the C concentration (g C g^{-1}) in biomass. The latter equation assumes that photosynthetically active biomass in old fields and forests is returned to the soil each year in seasonal litterfall. Results from other studies (Bray and Gorham, 1964; Sharpe et al., 1980) indicate that evergreen and deciduous forests have comparable annual amounts of aboveground leaf litterfall, therefore no distinction is made here between different forest types.

2.3.3 Soil Carbon and Nitrogen Stocks

The predicted soil C stock depends on the initial conditions for soil C (S_0), the decomposition rate, and the calculated C inputs to soil. The initial soil C stock is specified at the beginning of the model calculations.

The model tracks both new (i.e., fresh C inputs) and old soil C. Soil C stocks at time t (C_t, g C m^{-2}) are calculated from:

$$C_t = S_{t-1} + I_t - [(I_{t-1} * D_n) + (S_{t-1} * D_o)] \qquad [9]$$

where, S_{t-1} is the soil C stock at time t-1, I_t is the calculated soil C input at time t, D_n is the decomposition rate for fresh organic matter inputs (yr^{-1}), and D_o is the decomposition rate (yr^{-1})

for mineral soil C.

The soil N stock at time t (N_t, g N m^{-2}) is calculated on the basis of the predicted mineral soil C stock (C_m, g C m^{-2}) and a soil C:N ratio (R_s): $N_t = C_m/R_s$. Fresh soil C inputs are subtracted from C_t to estimate C_m. It is assumed that the fresh soil C inputs make no contribution to net soil N mineralization due to a high C:N ratio. Other studies indicate that mineral soil is the primary contributor to soil N availability whereas new soil C inputs result primarily in N immobilization (e.g., Whalen et al., 2000). Annual net soil N mineralization (N_m, g N m^{-2}) is calculated from:

$$N_m = N_t * M_r, \qquad [10]$$

where M_r is the potential annual rate of net soil N mineralization or, in other words, the fraction of bulk soil N that is made available for uptake by plant roots through decomposition of soil organic matter.

Total soil N supplies (N_s, g N m^{-2}) for plant nutrition and growth are predicted each year from the following equation:

$$N_s = N_m + N_f + N_d \qquad [11]$$

where N_f is annual N fertilizer additions to soil (g N m^{-2}), and N_d is annual atmospheric N deposition (g N m^{-2}).

2.3.4 Biomass Nitrogen Requirement

The annual net N requirement of biomass (B_n, g N m^{-2}) is calculated as:

$$B_n = (\Delta B_a * W_n) + (\Delta B_b * R_n) + [(B_f * L_n) * ((100 - T_f)/100)] \qquad [12]$$

where W_n is the N concentration (g N g^{-1}) in woody tissues, R_n is the N concentration (g N g^{-1}) in roots, L_n is the N concentration in photosynthetically active tissue (g N g^{-1}), and T_f is a translocation factor (fraction) that adjusts the N requirement based on N reserves that reside within the plant.

2.3.5 Threshold Test

Based on predicted biomass and soil C and N dynamics, the model calculates the annual N supply (N_s, g N m^{-2}) and subtracts the annual plant N requirement (B_n, g N m^{-2}) to arrive at an estimate of annual potential excess N (PEN, g N m^{-2}). If potential excess N is negative in any year, then a threshold has been crossed because available soil nutrient resources can not theoretically meet the N demands of the vegetation on track to a desired future ecosystem condition. If potential excess N is always positive, then nutrient resources are sufficient to achieve the desired future condition as defined by a target aboveground biomass and a specified rate of ecosystem recovery.

2.3.6 Model Parameter Summary

The model was parameterized to predict thresholds to recovery for old field and forest ecosystems at Fort Benning, Georgia. Model parameters (Table 1) were derived on the basis of (1) field studies, (2) literature values, and (3) approximation or parameter fitting. Parameters in the latter category included: the recovery rate for aboveground biomass (B_r), the decomposition rate of fresh litter inputs (D_n), and wood and root tissue N concentrations (W_n and R_n, respectively). Data from the literature were used to set parameter values for root:shoot ratios (R_w), the fraction of photosynthetically active biomass (f_L), root turnover times (T_b), C concentrations in biomass (C_b), the turnover time of mineral soil C (D_o), leaf N concentrations (L_n), and N translocation factors (T_r). Field studies at Fort Benning, complemented by data from the literature, were used to establish the following parameter values: targets for aboveground biomass (B_{max}), initial soil C stocks (S_0), soil C:N ratios (R_s), and potential net soil N mineralization rates (M_r). Even though it is included in the model equations, none of the ecosystems that are modeled here receive N fertilizer.

2.4 FIELD MEASUREMENTS

From 1999 to 2002, a variety of field studies were conducted to establish mean values for some soil attributes under different land cover categories at Fort Benning, Georgia. The data set included 14, 18, and 90 sets of measurements from barren sites, old fields, and forest sites, respectively. Sampling sites were widely distributed over the 73,000 ha installation. Details on the sampling methods are published elsewhere (Garten and Ashwood, 2004) but are briefly summarized here for the reader's convenience.

2.4.1 Aboveground Biomass

In April, 2002, measurements of diameter at breast height (DBH) were made along 40 m transects in four relatively undisturbed longleaf pine (*Pinus palustris*) stands that ranged from 56 to 82 years old. It is possible that the four sites may have been exposed to light military use (i.e., at most, light infantry foot training), but prior studies indicate the effects of such training on measures of soil C and N dynamics are not statistically significant (Garten et al., 2003). The basal area was calculated for each stand and converted to estimates of foliar biomass, woody biomass, and total aboveground biomass density (g m^{-2}) using regression equations, specific to longleaf pine, from Mitchell et al. (1999). Along with other estimates of maximum aboveground biomass in forests on the Piedmont (Johnson and Lindberg, 1992) and southeastern Coastal Plain (Switzer et al., 1968), the field data from mature longleaf pine stands were used to parameterize the desired future condition (as defined by aboveground biomass) for forest ecosystems at Fort Benning. Future site management plans include converting approximately half of the installation to longleaf pine forest.

Table 1. Parameter set for spreadsheet models of old field and forest soil C and N dynamics		
Parameter (symbol)	Units	Description (data source)
Aboveground biomass (B_{max})	$g\ m^{-2}$	Desired future condition (based on field data or literature values)
Recovery (R)	%	Recovery to maximum biomass
Recovery rate (B_r)	yr^{-1}	Estimated rate of aboveground biomass accumulation (see text)
Root:shoot (R_w)	ratio	Published sources (see text)
Photosynthetically active biomass (f_L)	fraction	Becomes annual leaf litterfall (see text)
Root turnover time (T_b)	years	From Gill and Jackson (2000)
Biomass C (C_b)	$g\ C\ g^{-1}$	Various sources (see text)
Decomposition rate of fresh litter inputs (D_n)	yr^{-1}	Fitted parameter to yield steady state soil C stock (see text)
Turnover time for mineral soil C (D_o)	years	Derived value (see text)
Initial soil C stock (S_0)	$g\ C\ m^{-2}$	Mean value to 30 cm soil depth (from field data)
Soil C:N (R_s)	ratio	Mineral soil (from field data)
Net soil N mineralization rate (M_r)	yr^{-1}	Potential rate (estimated from laboratory incubations)
Leaf N (L_n)	$g\ N\ g^{-1}$	Various sources (see text)
Wood N (W_n)	$g\ N\ g^{-1}$	Various sources (see text)
Root N (R_n)	$g\ N\ g^{-1}$	Various sources (see text)
Translocation factor (T_f)	fraction	Estimated internal N cycling (see text)
Annual N deposition (N_d)	$g\ N\ m^{-2}$	Estimated (from NADP/NTN data)
Annual N fertilizer (N_f)	$g\ N\ m^{-2}$	None

2.4.2 Soil Carbon and Nitrogen Stocks

Over a period of three years, soil C and N concentrations (g element g^{-1} soil) and stocks (g element m^{-2}) were measured at barren sites, old fields, and forests on Fort Benning to a 30 cm soil depth (Garten and Ashwood, 2004). Replicate soil samples were collected at each site using a stainless steel soil recovery probe (2.54 cm inner diameter) with hammer attachment (AMS, American Falls, ID). When present, the O-horizon was removed from a 214 cm^2 area directly above each soil sampling point. O-horizon dry mass was determined by drying at 65 °C, and soil samples were air-dried (22 °C) to a constant weight. Air dry soil samples were crushed using a rubber mallet and passed through a 2 mm sieve. A 20 gram portion of the sieved soil was dispersed by shaking overnight in 100 mL of sodium hexametaphosphate (5 g L^{-1}) and the mixture was wet sieved through a 0.053 mm sieve to estimate sand (g sand g^{-1} soil) and silt+clay content.

Soil density (g m^{-3}) was estimated from the dry soil mass and the calculated volume of each soil core. O-horizon and mineral soil samples were ground and homogenized and analyzed for C and N concentrations using a Perkin-Elmer 2400 Series II CHNS/O Analyzer (Perkin Elmer Analytical Instruments, Norwalk, CT) or a LECO CN-2000 (LECO Corporation, St. Joseph, MI). Carbon and N stocks in the O-horizon were calculated as the product of concentration (g element g^{-1} dry mass) and dry mass per unit area (g m^{-2}). Soil stocks were calculated as the product of concentration (g element g^{-1} soil), soil density (g soil cm^{-3}), and sampling depth (cm). The field data were used to parameterize soil C stocks (S_0) and soil C:N ratios (R_s) in the model.

2.4.3 Soil Nitrogen Availability

Potential net soil N mineralization was measured in mineral soil (0-20 cm deep) samples using aerobic laboratory incubations (Hart et al., 1994). The fresh mineral soil was passed through a 6.3 mm sieve to exclude rocks and coarse debris. Using methods described elsewhere (Garten et al., 2003; Garten and Ashwood, 2004), part of the sieved soil was used for the determination of C stocks in particulate organic matter (POM). A separate portion of sieved soil was extracted by shaking for two hours with 2 molar potassium chloride (1 part soil:10 parts solution) to determine initial extractable soil NH_4-N and NO_3-N. The sieved soils were incubated in closed plastic jars, in the dark, at room temperature (21 °C). The lids were briefly removed from the jars each week to aerate the soil samples. Extractions of incubating soils were repeated after 12 weeks to determine the net production rate of NH_4-N- and NO_3-N. Soil extracts were analyzed by digital colorimetry using a Bran+Luebbe AutoAnalyzer 3. Potential net soil N mineralization was calculated as the difference between extractable inorganic N (NH_4-N + NO_3-N) at 12 weeks and the initial extractable inorganic soil N. The units were μg N produced g^{-1} air dry soil based on the moisture content of the initial soil sample. For each sample, net soil N mineralization (μg N produced g^{-1} soil) over the entire 12-week incubation was normalized for soil N concentration (g N g^{-1} soil) and extrapolated to a potential annual rate (i.e., the fraction of soil N mineralized each year).

3. RESULTS

3.1 MODEL PARAMETERIZATION FROM FIELD DATA

3.1.1 Aboveground Biomass (B_{max})

Estimates of aboveground biomass in longleaf pine stands at Fort Benning were similar to stand biomass in mature (45 year old) forests on the southern Piedmont (≈18000 g m^{-2} based on data in Johnson and Lindberg, 1992) and stand biomass in loblolly pine (*Pinus taeda*) after 50 to

60 years of stand development (≈21000 g m^{-2} based on data in Switzer et al. 1968). Calculated mean (±SE) total aboveground biomass in the four longleaf pine stands (ranging from 56 to 82 years old) was 17995 ·±2415 g m^{-2}. Estimates of foliar and woody biomass in these same stands were 843 ± 111 g m^{-2} and 17163 ± 2306 g m^{-2}, respectively.

Although old-growth forests in the eastern U.S. may have somewhat greater aboveground biomass densities (22000 to 26000 g m^{-2} based on Brown et al., 1997), the desired future condition for modeling aboveground forest biomass at Fort Benning was set at 18000 g m^{-2}. The latter value is in the range of aboveground biomass densities of saw timber stands and forest stands in advanced stages of recovery (after forest clearing) in the eastern U.S. (Brown et al., 1997). Aboveground biomass targets for old field vegetation, 360 g m^{-2}, were established on the basis of other studies (Odum, 1960). The selected desired future conditions are merely examples of average recovery targets for the purpose of developing a model to predict thresholds to ecosystem recovery. They do not reflect Fort Benning land management goals that must consider a variety of ecological issues before establishing desired future ecosystem conditions, and which may or may not include maximizing standing biomass.

3.1.2 Soil Carbon and Nitrogen Stocks

Sand content in 129 soil samples collected at Fort Benning ranged from 12 to 95%. The mean sand content was 70% and two-thirds of the samples collected had a sand content that exceeded the mean. For the purpose of further analysis, each soil sample was binned into one of two categories (i.e., "less sandy" or "more sandy") based on whether the sand content was less than or more than 70%.

Barren Sites -- Soils from barren sites, with the exception of one sample, had a sand content greater than 70% (the exception was 69% sand). Mineral soil C and N stocks at barren sites (Table 2) were significantly less than those measured under old fields and forest cover (Garten and Ashwood, 2004). Because of a lack of plant cover, barren sites were generally devoid of any O-horizon material. Soil C:N ratios at barren sites were highly skewed with an inflated the mean due to a few samples with low soil N concentrations. The median soil C:N ratio at barren sites was 16.2 and the geometric mean was 22.2.

Old Fields -- Table 3 summarizes measured C and N stocks and C:N ratios in the O-horizon and mineral soil under old fields with less than or more than 70% sand content. Old fields on less sandy soils had significantly greater soil C and N stocks than those on more sandy soils. The more sandy soils also tended to have higher soil C:N ratios, but the difference was not statistically significant.

Forests -- Differences in forest soil C stocks under less sandy and more sandy soils were not statistically significant (Table 3). Forest O-horizon N stocks were significantly greater on more sandy soils but the mineral soil N stocks were significantly lower than those on less sandy sites. Similar to old field sites, forest mineral soil C:N ratios were elevated at sites with more than 70% sand content.

Table 2. Mineral soil C and N stocks, soil C:N ratios, extractable inorganic soil N, net N and NO_3-N production in 12-week aerobic laboratory incubations, and estimated potential net N mineralization rate for barren soils at Fort Benning, GA

Variable	Units	n	Mean	SE [a]	C.V. [b]
Soil C stock	g C m^{-2}	14	629	146	0.87
Soil N stock	g N m^{-2}	14	39.4	9.5	0.90
Soil C:N ratio	none	12	37.7	16.4	1.51
Extractable inorganic N	µg N g^{-1}	14	0.97	0.20	0.76
Net N production	µg N g^{-1}	14	1.13	0.62	2.05
Net NO_3-N production	µg N g^{-1}	14	0.84	0.02	0.09
Net N mineralization rate	% yr^{-1}	14	4.6	3.2	2.60

[a] Standard error
[b] Coefficient of variation

3.1.3 Soil Nitrogen Availability

Barren sites -- Consistent with lower soil N stocks, there was less soil N availability under barren sites compared to sites occupied by perennial vegetation (Garten and Ashwood, 2004). Absolute amounts of potential net soil N mineralization and net nitrification in aerobic laboratory incubations were reduced at barren sites (Table 2) . However, the potential annual net N mineralization rate for barren soils was comparable to old field and forest soils.

Old fields -- Differences in N availability between old fields on less sandy or more sandy soils are presented in Table 4. Compared to more sandy soils, less sandy soils tended to have higher levels of extractable inorganic soil N (there was a 10% probability that this difference occurred by chance). Differences between less sandy and more sandy soils in net N and NO_3-N production during the 12 week aerobic laboratory incubations were not statistically significant. The mean potential net N mineralization rate, expressed on an annual basis, tended to be greater in old field soils with more than 70% sand content ($P < 0.10$). If not all, most of the soil N mineralization terminated by production of NO_3-N, a highly available form of soil N.

Forests -- Measures of soil N availability in forest soils were similar to those for old field soils (Table 4). Based on results from the laboratory incubations, there were no statistically significant differences in N availability between less sandy and more sandy forest sites. However, consistent with trends observed for old field soils, the potential N mineralization rate was greater in forest soils with more than 70% sand content. As in old field soils, most of the net soil N mineralization terminated in the production of NO_3-N.

Table 3. Carbon and N stocks (g element m^{-2}) and C:N ratios in old field and forest soils with less than ("less sandy") or more than ("more sandy") 70% sand content at Fort Benning, GA								
Variable	Soil part	Less sandy			More sandy			F-value
		n	Mean	SE	n	Mean	SE	
		- Old field soils -						
C stock	O-horizon	6	151	82	9	90	25	0.7
	Mineral soil[a]	7	3457	192	11	2440	230	9.6**
	Total	6	3702	252	10	2514	261	9.2**
N stock	O-horizon	6	5.6	4.0	9	2.2	0.7	1.1
	Mineral soil[a]	7	189	33	11	99	15	7.7*
	Total	6	197	43	10	103	16	6.0*
C:N	O-horizon	3	38.9	12.8	7	44.6	4.3	0.3
	Mineral soil[a]	7	21.6	3.8	11	34.6	9.0	1.2
		- Forest soils -						
C stock	O-horizon	33	378	24	56	450	27	3.4
	Mineral soil[a]	34	3342	186	56	3397	211	0.0
	Total	34	3709	188	56	3847	221	0.2
N stock	O-horizon	33	6.4	0.6	56	8.7	0.7	5.0*
	Mineral soil[a]	34	167	11	56	109	6	24***
	Total	34	173	12	56	118	6	21***
C:N	O-horizon	33	66.7	4.3	56	60.9	3.6	1.0
	Mineral soil[a]	34	21.4	1.1	56	35.6	3.9	7.9**

[a]Depth of the mineral soil is 30 cm
* $P \leq 0.05$; ** $P \leq 0.01$; *** $P \leq 0.001$

3.2 MODEL PARAMETERIZATION FROM OTHER SOURCES

Many parameter values (Table 5) in the model were established on the basis of sources other than field data because site-specific data were not available. The rationale for setting parameter values based on other sources is described in the following paragraphs.

3.2.1 Recovery Rate (B_r)

The recovery (or growth) rate for forest stands was set to 0.15 yr^{-1}. At this rate, the stand achieves 95% of its target desired future condition (18000 g m^{-2}) in ≈50 years when starting from an initial condition of 360 g m^{-2}. The annual recovery rate for herbaceous old fields was set at 1.0 yr^{-1} on the basis that herbaceous communities tend to rapidly achieve a steady state standing biomass from existing soil seed banks and recolonization by opportunistic species. Recovery rates can be adjusted at the discretion of the user for different types of plant communities.

Table 4. Extractable inorganic soil N (µg N g^{-1} soil), net N and NO_3-N production in 12-week laboratory incubations (µg N g^{-1} soil), and estimated potential net N mineralization rate (% yr^{-1}) in old field and forest soils with less than ("less sandy") or more than ("more sandy") 70% sand content at Fort Benning, GA

Variable	Less sandy			More sandy			F-value
	n	Mean	SE	n	Mean	SE	
	- Old field soils -						
Extractable inorganic N	7	4.3	1.9	11	1.5	0.2	3.5[a]
Net N production	7	4.7	3.2	11	5.4	1.3	0.0
Net NO_3-N production	7	4.5	2.3	11	6.2	1.3	0.4
Net N mineralization rate	7	2.5	1.9	11	7.1	1.4	4.1[a]
	- Forest soils -						
Extractable inorganic N	34	2.6	0.3	56	2.2	0.2	0.8
Net N production	34	4.6	1.1	56	5.4	1.0	2.7
Net NO_3-N production	34	3.5	1.0	56	4.3	0.9	0.4
Net N mineralization rate	34	2.6	0.5	56	6.4	1.1	6.9**

[a] $P \leq 0.10$
** $P \leq 0.01$

3.2.2 Root:shoot Ratios (R_w)

The root:shoot ratio in forest stands was set at 0.23 based on information presented by Jackson et al. (1996) and Cairns et al. (1997) who summarized global data on root biomass from terrestrial biomes and upland forests, respectively. The root:shoot ratio under old fields was set to 1.0 based on studies by Kelly (1975) who measured root:shoot ratios of 0.78 and 1.4 in two east Tennessee old field communities.

3.2.3 Fraction of Photosynthetically Active Biomass (f_L)

This parameter determines the portion of aboveground biomass that contributes to annual soil C inputs. Based on data from forests in the southeastern U.S. (Johnson and Van Hook, 1989; Johnson and Lindberg, 1992), leaf biomass is typically 2 to 5% of total aboveground biomass. Unlike trees, both stems and leaves are photosynthetically active in many herbaceous plants. For the purposes of modeling thresholds to recovery, the fraction of photosynthetically active biomass in forests and herbaceous old fields was set at 3.5% and 100%, respectively.

Table 5. Parameter sets for modeling the nutrient threshold to recovery of old field and forest communities on less sandy (less than 70% sand) and more sandy (more than 70% sand) soils at Fort Benning, GA

Parameter	Units	Less sandy		More sandy	
		Old field	Forest	Old field	Forest
B_{max}	g m^{-2}	360	18000	360	18000
R	%	100	100	100	100
B_r	yr^{-1}	1.0	0.15	1.0	0.15
R_w	ratio	1.0	0.23	1.0	0.23
f_L	fraction	1.0	0.035	1.0	0.035
T_b	years	2.0	10	2.0	10
C_b	g C g^{-1}	0.45	0.45	0.45	0.45
D_n	yr^{-1}	0.8982	0.9474	0.9310	0.9454
D_o	years	150	150	150	150
S_0	g C m^{-2}	3702	3709	2514	3846
R_s	ratio	21.6	21.4	34.6	35.6
M_r	yr^{-1}	0.0249	0.0259	0.0711	0.0637
L_n	g N g^{-1}	0.01	0.01	0.01	0.01
W_n	g N g^{-1}	0	0.001	0	0.001
R_n	g N g^{-1}	0.01	0.01	0.01	0.01
T_f	fraction	0.5	0.5	0.5	0.5
N_d	g N m^{-2}	0.7	0.7	0.7	0.7
N_f	g N m^{-2}	0.0	0.0	0.0	0.0

3.2.4 Root Turnover Times (T_b)

Root turnover times for plant communities at Fort Benning were estimated on the basis of globally averaged root turnover rates in grasslands (50% per year) and forests (10% per year) (Gill and Jackson, 2000).

3.2.5 Decomposition Rate of Fresh Litter Inputs (D_n)

The mean residence time of soil C associated with above- and belowground litter inputs was derived by parameter fitting below an estimated upper limit. Based on regional estimates of forest litterfall (Sharpe et al., 1980) and a concentration of 0.5 g C g^{-1} litter, the estimated annual input of C to forest O-horizons at Fort Benning is 204 g C m^{-2}. Measured mean (±SE) O-horizon C stocks at 89 forest sites on Fort Benning were 423 ±19 g C m^{-2}. Assuming the O-horizon C stocks are at steady state, an upper limit to the mean residence time of fresh litter inputs at this site is ≈2 years. The fitted mean residence times for fresh litter inputs to old field and forest soils (Table 5) were approximately half the estimated upper limit because the litter C inputs are underestimated by not considering belowground inputs from roots. The final fitted values for D_n yielded steady state values for both potential excess N and soil C stocks in the model.

3.2.6 Turnover Time of Mineral Soil Carbon (D_o)

Soil C under transitional herbaceous vegetation and forests at Fort Benning includes ≈10% refractory C that is chemically similar to charcoal (Garten and Ashwood, 2004) and probably has a turnover time on the order of 1000 years. This refractory C originates from frequent use of controlled burning in land management. Most of the remaining C is found in mineral-associated soil organic matter (Garten and Ashwood, 2004) and is assumed to have a turnover time of 56 years based on data from multiple studies (Garten and Ashwood, 2002). The turnover time of mineral soil C under both forests and old fields was estimated as a weighted mean of the two pools (i.e., 150 years).

3.2.7 Tissue Carbon and Nitrogen Concentrations

It was assumed that biomass had a C concentration of 0.45 g C g^{-1} dry mass. Leaf N concentrations in the model were set at 1% based on data from multiple sources (Birk and Vitousek, 1986; Yin, 1993). Nitrogen concentrations in roots were assumed to equal those in foliage based on studies of loblolly pine on upper Coastal Plain sites (Birk and Vitousek, 1986). Concentrations of N in tree wood were set at 0.1% which approximates those measured in loblolly pine in the southeastern U.S. (Switzer et al., 1968; Birk and Vitousek, 1986).

3.2.8 Translocation Factor (T_f)

Seasonal translocation of N from foliar to woody tissues in trees (Luxmoore et al., 1981; Ostman and Weaver, 1982) and from aboveground tissues to roots in herbaceous plants (Li et al., 1992) is a well known process. The translocated N is available for production of new tissues at the beginning of the next growing season. In the model, the N requirements of photosynthetically active tissues were reduced each year based on the estimated N recycling within the plant. Studies of loblolly pine on sandy soils indicate that about 50% of the foliar N is translocated to wood prior to leaf senescence (Birk and Vitousek, 1986). Under conditions of low soil N availability, ≈50% of the N required for production of new biomass in herbaceous vegetation may be derived from internal translocation (e.g., Li et al., 1992). Therefore, in the absence of site-specific information, the translocation factor was set at 50% in both forests and old fields on Fort Benning.

3.2.9 Annual Nitrogen Deposition (N_d)

Based on data from the National Atmospheric Deposition Program, annual wet only N deposition in the Fort Benning area is ≈0.35 g N m^{-2}. A scaling factor (2.0) to convert wet deposition to total N deposition (wet + dry) was derived from data previously collected at four sites in the southeastern U.S. (Lovett and Lindberg, 1993). Total annual N deposition in the model was set at 0.7 g N m^{-2}.

3.3 PREDICTED THRESHOLDS TO RECOVERY

Table 5 presents a summary of the parameter values used to model thresholds to recovery for both old field and forest vegetation. Different parameter sets were used for old fields or forests depending on soil sand content. Many of the parameters (e.g., aboveground biomass targets, root:shoot ratio, and root turnover times) exhibited strong differences between the two ecosystems. However, some parameter values derived from field and laboratory studies (e.g., soil C:N ratios and annual potential rates of net soil N mineralization) were similar for different ecosystems within the same soil category.

Four factors were particularly important to development of a threshold event (i.e., a negative value for potential excess N) during modeled ecosystem recovery : (1) initial amounts of aboveground biomass, (2) initial soil C stocks, (3) relative recovery rate of aboveground biomass, and (4) soil sand content. In this study, stocks of aboveground forest biomass were initialized by assuming 2% of the desired future condition (i.e., 360 g m^{-2}) was present at the start of ecosystem recovery. Simulations of old fields were initialized by assuming aboveground biomass was 25% of the desired future condition (i.e., 90 g m^{-2}) at the start of ecosystem recovery.

Initial soil C stocks in the model determined predicted patterns of recovery by both old field and forest ecosystems. Figure 2 illustrates the predicted recovery of (a) aboveground biomass, (b) potential excess N, and (c) soil C stocks for two different soils (1000 and 2000 g soil C m^{-2}) in old field ecosystems on “more sandy” soils. Starting from an initial stock of 1000 g soil C m^{-2}, a nutrient threshold to recovery was crossed in the fourth year. Predicted potential excess N remained negative for the duration of the simulation indicating that the desired future condition (360 g m^{-2} aboveground biomass) could not be achieved. Starting from an initial stock of 2000 g soil C m^{-2}, predicted potential excess N was positive (1 g N m^{-2}) indicating the desired future condition was achievable and sustainable for old fields on “more sandy” soils. In the latter case, predicted soil C stocks increased by about 12% over 50 years.

In the model, a slower recovery rate (B_r) could prevent forests from crossing a nutrient threshold during ecosystem recovery. Figure 3 illustrates the change in (a) aboveground biomass, (b) potential excess N, and (c) soil C stocks over 120 years at two different rates of forest growth. Starting from an initial condition of 1700 g soil C m^{-2} (90% of the barren sites examined at Fort Benning had soil C stocks less than this value), and at a default recovery rate of 0.15 yr^{-1},

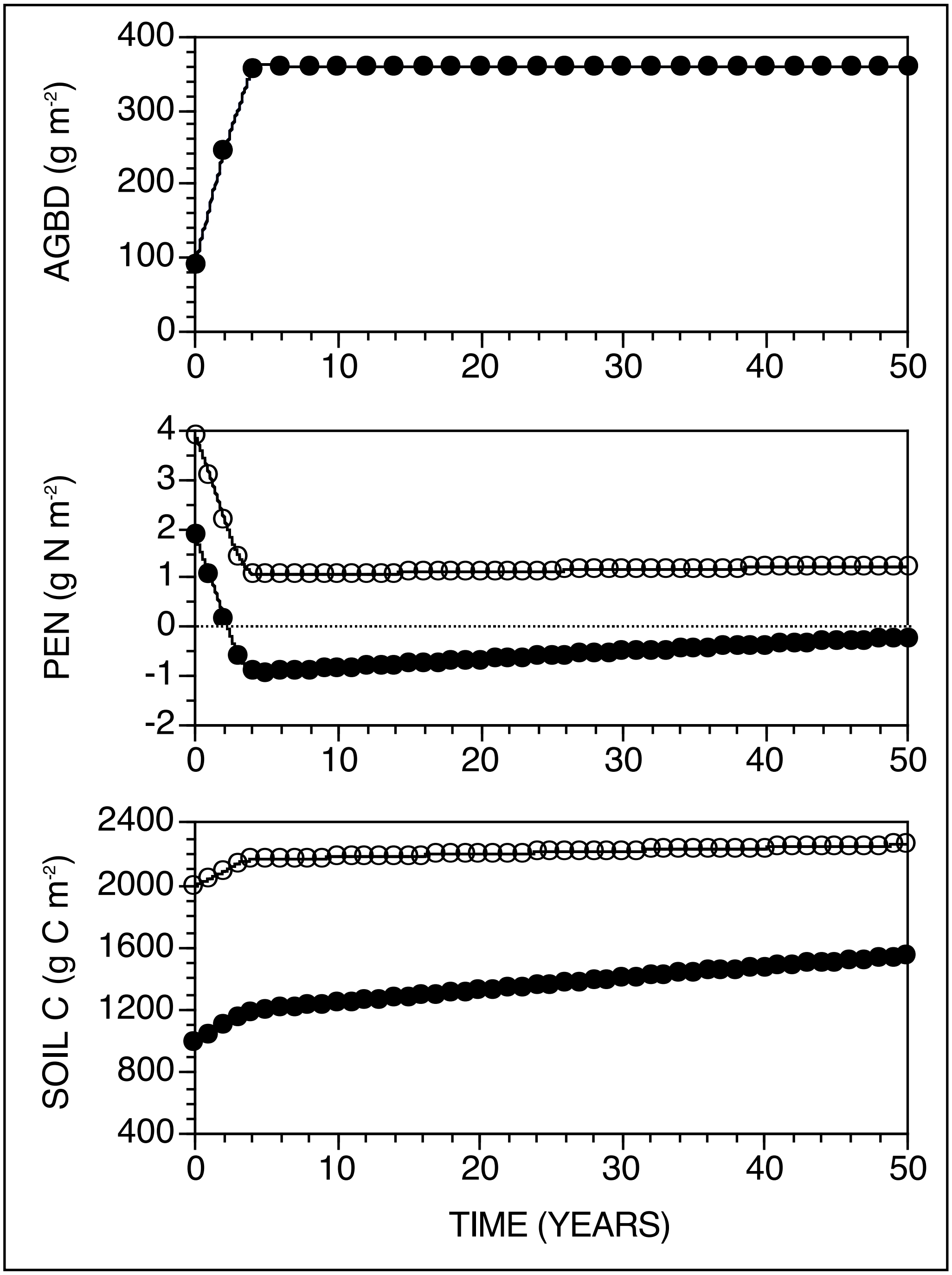

Fig. 2. Predicted recovery of aboveground biomass density (AGBD) (upper panel), potential excess N (PEN) (middle panel), and soil C stocks (lower panel) for old field development on soils with two different levels of initial soil C stocks at Fort Benning, GA. Starting from 1000 g C m^{-2} soil (closed circles), predicted PEN values quickly become negative (middle panel) indicating that a nutrient threshold precludes ecosystem recovery at the specified recovery rate (see text). Calculations with the model indicate that old field recovery, as illustrated in the upper and bottom panels, is possible starting from 2000 g C m^{-2} soil (open circles).

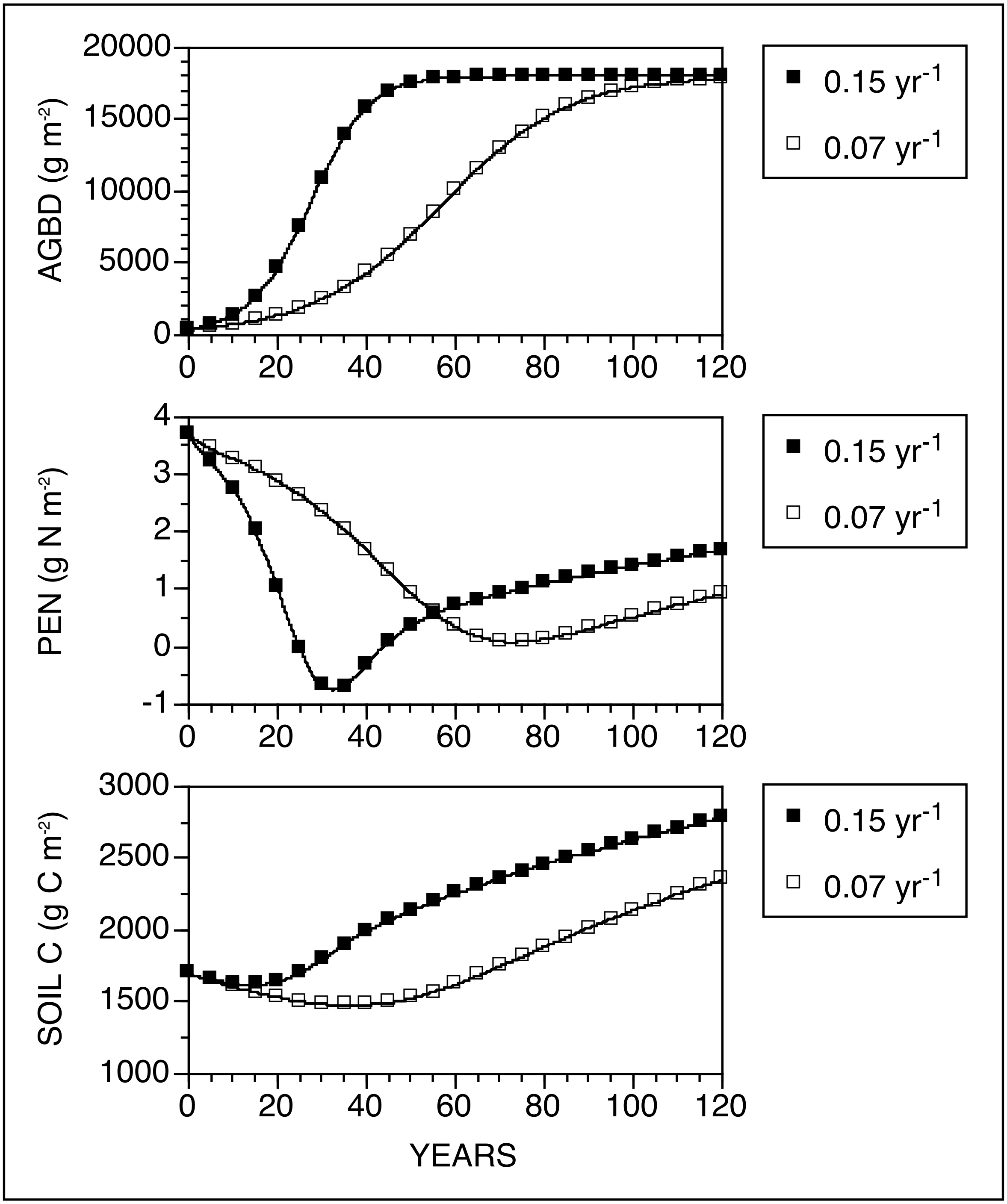

Fig. 3. Predicted recovery of aboveground biomass density (AGBD) (upper panel), potential excess N (PEN) (middle panel), and soil C stocks (lower panel) for forest ecosystems with different recovery rates at Fort Benning, GA. When the recovery rate is 0.15 yr^{-1}, then predicted PEN values are negative indicating that a nutrient threshold precludes ecosystem recovery (see text). Calculations with the model indicate that forest recovery, as shown in the upper and bottom panels, is possible when the recovery rate is reduced to 0.07 yr^{-1}.

the desired future condition for aboveground forest biomass was theoretically attained in ≈50 years. However, at the default recovery rate, potential excess N becomes negative from 25 to 45 years into the simulation indicating N deficiency could prevent recovery to the desired future condition with the specified parameter set. Even though the accumulation of soil organic matter was slower when the growth rate was lowered to 0.07 yr^{-1}, a nutrient threshold to forest recovery was not crossed and the desired future condition for forest biomass was achieved following 110 years of forest growth.

Predicted thresholds to recovery for forests and old fields at Fort Benning are illustrated in Figure 4. The various lines in the graph define initial soil C stocks that allow recovery to a desired future condition with the model (using the parameter sets presented in Table 5). Above

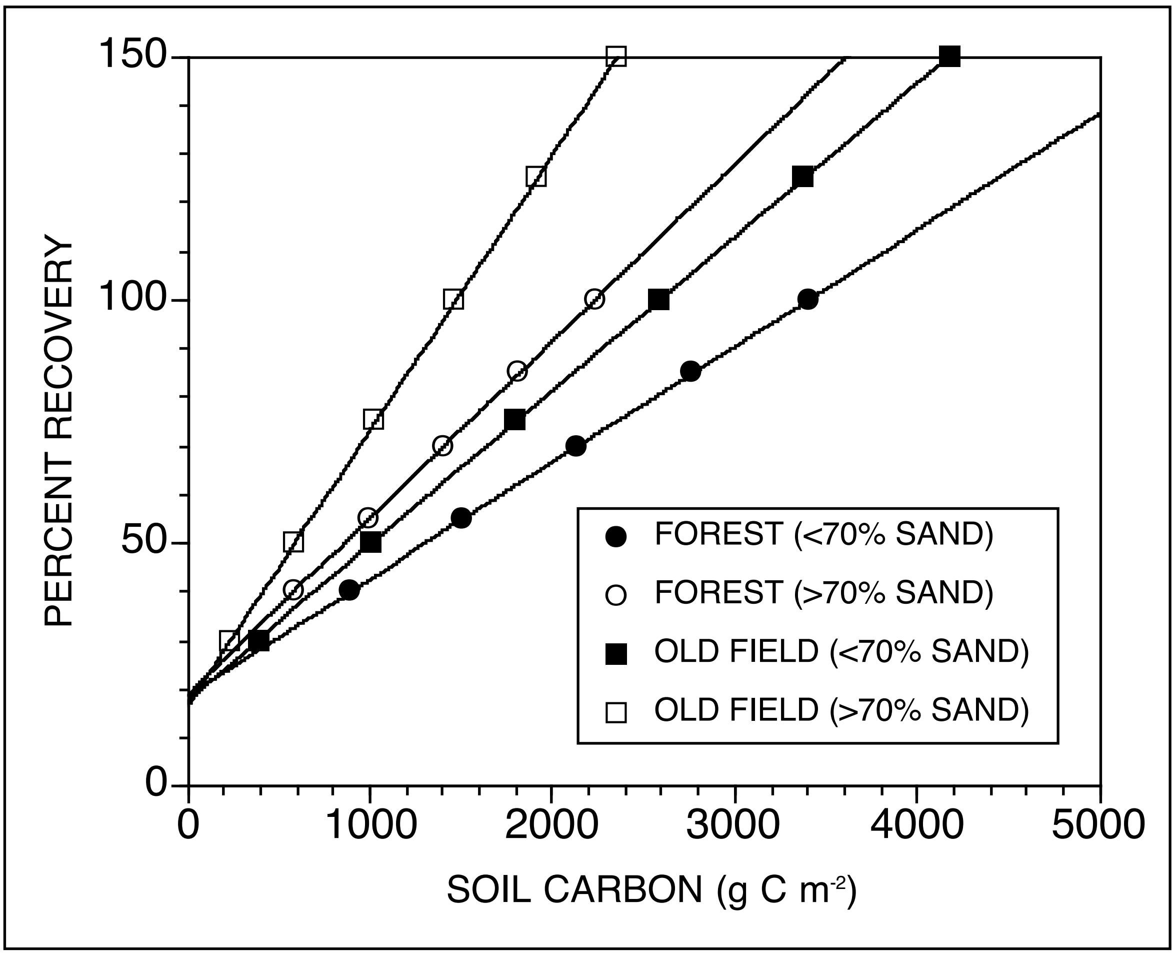

Fig. 4. Predicted thresholds to recovery for old field and forest ecosystems on more sandy (>70% sand content) and less sandy (<70% sand content) soils at Fort Benning, GA. Nutrient thresholds to recovery exist in the region above each line. In the region below each line, ecosystem recovery proceeds to a desired future condition without crossing a threshold (i.e., negative potential excess N). The predicted thresholds depend on the specified recovery rate (see text).

each line, the predicted potential excess N becomes negative at some time during the simulation of ecosystem recovery while below each line ecosystem recovery proceeds to the desired future condition with continuously positive potential excess N. For example, starting from an initial condition of 2000 g soil C m^{-2} on soils with less than 70% sand content, the model predicts recovery to 80% of the desired future condition in old fields without crossing a threshold to recovery (i.e., a negative potential excess N). Higher percent recoveries cannot be achieved because the nutrient threshold is crossed. Stated in another way, the model predicts recovery to 80% of the desired future condition for old fields on "less sandy" soils at Fort Benning can be achieved from an initial starting soil quality of 2000 g C m^{-2} or greater.

Forests and old fields on soils with differing sand content had different predicted thresholds to recovery (Figure 4). Within each ecosystem type, predicted soil N stocks were greater on less sandy soils due to their lower soil C:N ratios. However, rates of annual potential net soil N mineralization, derived from laboratory incubations, were higher on more sandy soils than less sandy soils. Consequently, predicted thresholds to recovery of old fields and forests were lower on soils with more than 70% sand content. For example, the model predicted 100% forest recovery at ≈2200 g C m^{-2} on more sandy soils but only 70% recovery on less sandy soils with the same soil C stock (Fig. 4). More sandy soils under perennial vegetation had a significantly ($F_{1,107} = 17.5$; $P < 0.001$) greater fraction of soil C in POM and significantly ($F_{1,107} = 4.2$, $P < 0.05$) greater stocks of surface mineral soil POM carbon than less sandy soils (Fig. 5). Particulate organic matter is a highly labile C pool that may be important to N retention and availability in some soils (e.g., Hook and Burke, 2000; Willson et al., 2001). Greater amounts of labile soil organic matter may be one factor contributing to higher potential net soil N mineralization rates in more sandy soils at Fort Benning.

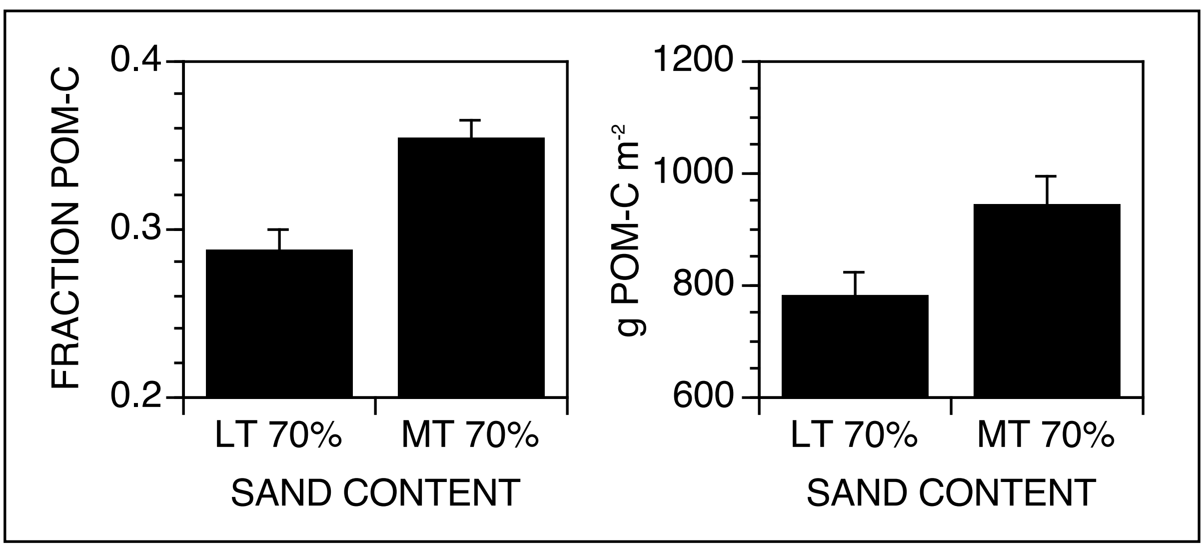

Fig. 5. Fraction of surface (0-20 cm) soil C associated with particulate organic matter (left panel) and stocks of POM carbon (right panel) in soils with less than (LT) or more than (MT) 70% sand content under perennial vegetation at Fort Benning, GA.

Using the parameter sets presented in Table 5, barren sites at Fort Benning generally lie below predicted thresholds to 100% recovery of desired future conditions. The 95% confidence interval about mean soil C stocks on barren land at Fort Benning was 313 to 944 g C m^{-2}. On more sandy soils (typical of barren sites), predicted forest recoveries exceeding 55% of the desired future condition cross the nutrient threshold to recovery at ≈950 g C m^{-2}. However, at the default recovery rates (Table 5), the model predicts up to 70% recovery of old field biomass can be achieved from an initial soil C stock of 950 g C m^{-2} and up to 50% recovery can be achieved from an initial soil C stock of 600 g C m^{-2}. Figure 6 further illustrates the effect of varying recovery rate and percent recovery on calculated thresholds to forest recovery on less sandy and more sandy soils at Fort Benning. Predicted recovery to 100% of the desired future condition, without crossing a threshold in N availability, is indicated even at relatively low initial soil C stocks on more sandy soils, but only at low recovery rates.

4. DISCUSSION

The concept of thresholds has not been widely applied in ecosystem management (Brown et al., 1999). Ecologists have been successful in identifying factors associated with thresholds (e.g., nutrient loading leading to eutrophication, overgrazing leading to the loss of range land, habitat fragmentation leading to loss of biodiversity), but threshold quantification has been more problematic. In some cases, a single well-defined threshold may not exist or the threshold may depend on site-specific factors that make predictions beyond the local conditions difficult or impossible. In other cases, thresholds may be influenced by one or more factors that are indirectly related to the stimulus or stress that causes a response. Given the large likelihood that no two ecosystems are totally identical in time or in space, we can expect natural variation in thresholds from one time to another and from one location to another.

Relatively intensive monitoring of ecosystem structure and function may be required for empirical detection and quantification of ecological thresholds (Muradian, 2001). Reestablishment of vegetation and recovery of soil quality on degraded soils can be a long-term process requiring decades to centuries, thus (with the possible exception of chronosequence studies) empirical investigations directed at discerning nutrient thresholds to ecosystem recovery are impractical. Periodic measurements of indicators of ecosystem "health" may be sufficient to ascertain general trends, but considering the extent of spatial and temporal variation in natural systems, the level of monitoring required to detect thresholds through field studies is, practically speaking, prohibitively expensive. Furthermore, some thresholds may only be recognized "after the fact" from analysis of long-term monitoring data (in which case it may be too late for land managers to initiate corrective actions). There is a much higher probability of detecting an ecological threshold when a system "jumps" to an entirely new state or undergoes total collapse as a result of a recognizable disturbance (e.g., hurricane, disease, crown fire, or overgrazing), particularly when such a change is manifested as a sudden and dramatic difference in vegetation structure (like the creation of barren land through soil disturbance).

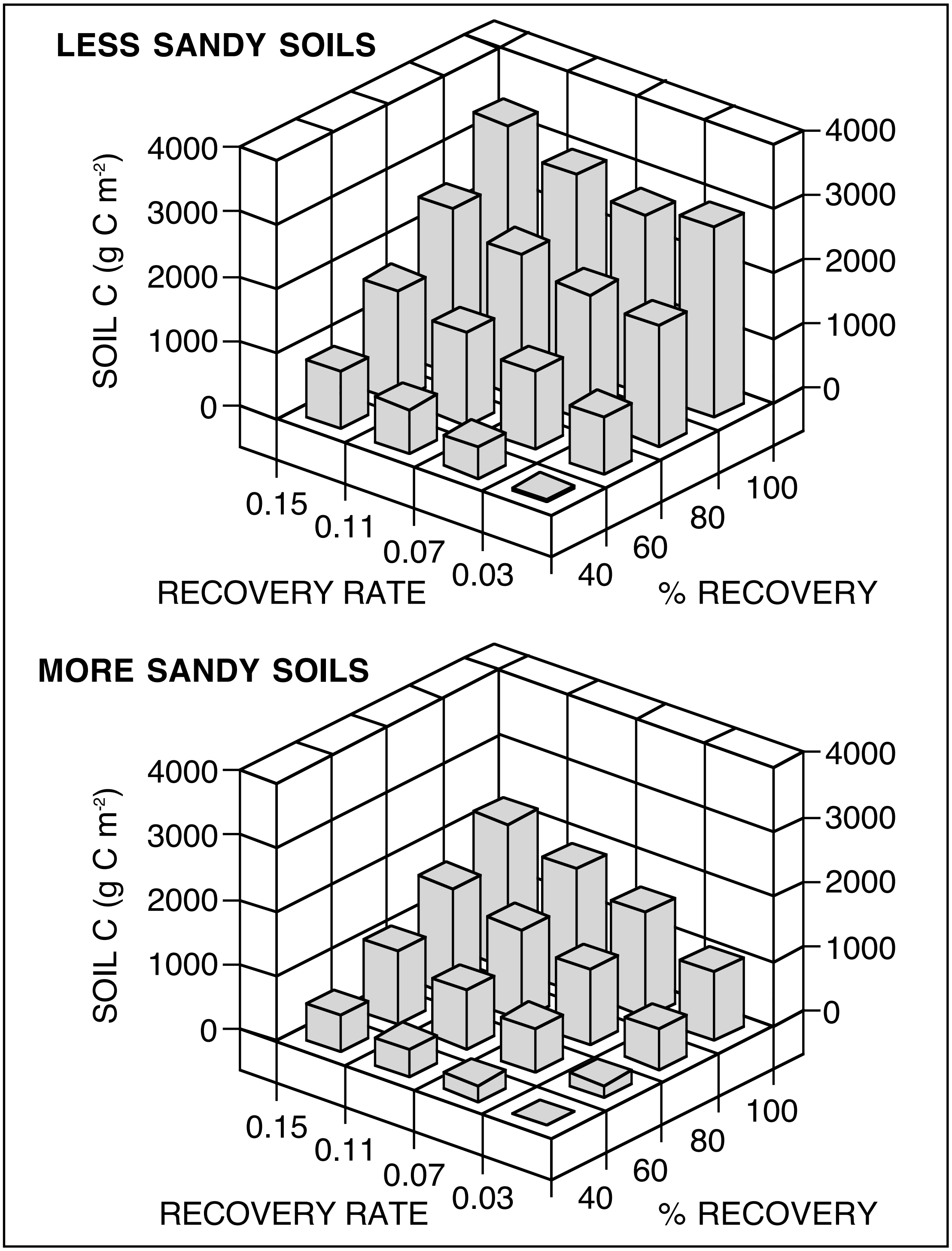

Fig. 6. Predicted thresholds to forest recovery (expressed as initial soil C stocks) on more sandy (> 70% sand content) and less sandy (< 70% sand content) soils as a function of recovery rate (yr^{-1}) and percent recovery to a desired future condition (18000 g aboveground biomass m^{-2}). When soil C stocks are less than those shown, recovery to the desired future condition at the indicated recovery rate is theoretically precluded by a deficiency in soil N availability at time point during forest development.

In this research, simple mathematical models of C and N (as indices of soil quality) are used to predict a nutrient resource threshold to ecosystem recovery. Biomass production in the southeastern U.S. is often limited by available soil N (Fisher and Garbett, 1980; Birk and Vitousek, 1986; Vose and Allen, 1991), thus the current model is developed around N limitation to the achievement of a desired future condition, measured in terms of aboveground standing crop biomass and a specified recovery rate. Modeling is usually an oversimplification, and it is certainly no less so in the present study, but the use of models for estimation of thresholds to ecosystem recovery can meet two important needs: (1) the use of site-specific information for critical parameter values, and (2) the removal of constraints associated with evaluating thresholds from a limited set of *apriori* conditions (as is frequently the case in empirical studies). Furthermore, in mathematical models, thresholds can be posed as simple "true or false" questions; or framed such that a threshold test indicates if a situation will or will not occur.

Mathematical models have their own unique set of problems when applied to the estimation of ecological thresholds (Moir and Mowrer, 1995; Hansen and Jones, 1996). Errors in model structure and parameterization can lead to erroneous predictions of thresholds or, at least, contribute to large, unknown uncertainties in the accuracy of estimated thresholds. In addition, there is a pervasive skepticism about the usefulness of models (e.g., Passioura, 1996), whether they be simple or complex. Complex models may give greater representation to system properties and processes, but they are more difficult to understand, parameterize, and verify (Schoenholtz et al., 2000). Simple models are easier to use and understand but may neglect important system properties or processes.

Calculations with the model indicate that thresholds to ecosystem recovery from degraded land, expressed as initial soil C stocks, are lower on more sandy soils than on less sandy soils at Fort Benning. The lower thresholds for old field and forest recovery on more sandy soils are largely due to higher estimated relative rates of net soil N mineralization in more sandy soils. Correct parameterization of net soil N mineralization is particularly important to the accurate estimation of thresholds to ecosystem recovery. If rates of potential net soil N mineralization are overestimated in the model, then predicted thresholds to recovery (expressed as initial soil C stocks) are underestimated. At near steady-state soil C and N stocks, the calculated soil N mineralization flux in the model ranged from about 4 to 6 g N m^{-2} yr^{-1} and approximated the annual *in situ* mineralization flux (about 5 g N m^{-2}) measured in uncut loblolly pine plantations growing on the North Carolina Piedmont (Vitousek and Matson, 1985). The calculated rates from Fort Benning are, however, substantially greater than previously measured *in situ* net soil N mineralization rates (0.5 to 1.2 g N m^{-2}) in longleaf pine ecosystems on the coastal plain in southwest Georgia, which the investigators (Wilson et al., 1999) acknowledge as “among the lowest rates recorded for North American forests”. Such differences between studies in soil N availability reinforce the need for site-specific information when models are used to estimate nutrient thresholds to ecosystem recovery.

Much is already known about the development of old fields and their importance in secondary ecological succession, N accrual, and ecosystem recovery on degraded or abandoned agricultural land (Keever,1950; Odum, 1960; Wiegert and Evans, 1964; Robles and Burke, 1997; Knops and Tilman, 2000), but more site specific information is needed before N fixation at Fort Benning can be accurately represented in the current model. Frequent controlled burning may promote the establishment and persistence of legumes in southern pine ecosystems, but the annual contribution of these legume populations to overall ecosystem N balance is relatively small (<1 g N m^{-2}) and may merely balance N losses incurred through prescribed burning (Hendricks and Boring, 1999). Nitrogen fixation on the order of 2 g N m^{-2} yr^{-1} was reported by Jorgensen and Wells (1971) in annually burned loblolly pine stands on the lower coastal plain of South Carolina, but the process exhibited a high degree of spatial variability. Other studies indicate a decline in biological N fixation well before a decline in N accrual during ecological succession (Rastetter et al., 2001) and that atmospheric deposition may contribute more to N accrual during forest regrowth on abandoned agricultural land in the southeastern U.S. than N fixation (Richter et al., 2000) . Thus, the omission of N fixation in the current model may not seriously bias calculated thresholds to ecosystem recovery.

Finally, calculations with the model indicate that a combination of desired future conditions, initial levels of soil quality (defined by soil C stocks), and the rate of biomass accumulation determines the predicted success of ecosystem recovery on disturbed soils. Thresholds to ecosystem recovery predicted by the model should not be interpreted independent of the specified recovery rate to a desired future condition. This is best illustrated by graphing thresholds to forest recovery (expressed as initial soil C stocks) as a function of percent recovery to a desired future condition and recovery rate on less sandy and more sandy soils at Fort Benning (Fig. 6). Thresholds of soil C stocks do not indicate that ecosystem recovery is strictly precluded, only that it is precluded at a specified rate of aboveground biomass accumulation. A lack of feedback between soil N availability and the rate of biomass accumulation in the model causes the ecosystem to grow into a state of N deficiency (i.e., negative potential excess N). In a similar fashion, forests developing on nutrient poor soils in the southeastern U.S. tend to grow into a state of acute N deficiency that eventually limits biomass production (Richter et al., 2000; Gholz et al., 1985). Within the constraints imposed by soil organic matter, ecosystem recovery is also related to the frequency of disturbance and land management actions (e.g., fertilization) that affect soil quality and productivity of vegetation. In this respect, the threshold model can be used to predict how much and how long N fertilizer would need to be applied to enable ecosystem recovery from a specified initial level of soil organic matter at Fort Benning, Georgia.

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APPENDIX D.

Predicted Effects of Prescribed Burning and Timber Management on Forest Recovery and Sustainability at Fort Benning, Georgia

ORNL/TM-2004/77

Predicted Effects of Prescribed Burning and Timber Management on Forest Recovery and Sustainability at Fort Benning, Georgia

April 2004

C.T. Garten, Jr.

Environmental Sciences Division

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PREDICTED EFFECTS OF PRESCRIBED BURNING AND TIMBER MANAGEMENT ON FOREST RECOVERY AND SUSTAINABILITY AT FORT BENNING, GEORGIA

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LIST OF ABBREVIATED TERMS

AGIN	annual aboveground woody growth increment (g m^{-2} yr^{-1})
AGROWTH	input to aboveground woody biomass (g m^{-2} yr^{-1})
AGRT:ST	ratio of belowground tree biomass to aboveground woody biomass
AGWB	aboveground woody biomass (g m^{-2})
AGWB%	aboveground wood biomass expressed as a percentage of maximum value
ATMN	atmospheric N deposition (g N m^{-2} yr^{-1})
BGROWTH	input to belowground tree root biomass (g m^{-2} yr^{-1})
BGWB	belowground tree root biomass (g m^{-2})
CONCN	plant tissue N concentration (g N g^{-1})
CUT	operator in the model that turns tree harvesting “on” or “off”
CUTFREQ	return interval of forest thinning or harvesting (years)
FIRE	operator in the model that turns prescribed burning “on” or “off”
FIREFRQ	return interval of prescribed burning
FIRELOSS	amount of soil C lost as a result of prescribed burning (g C m^{-2} yr^{-1})
fLEAF	fraction of aboveground biomass that is foliage
FRACOH	fraction of soil C stock in the O-horizon that is lost via prescribed burning
GRWTHMOD	factor modifying input to aboveground woody biomass (N feedback)
HARVST	removal of aboveground woody biomass by forest thinning or harvesting
HBAG	herbaceous aboveground biomass (g m^{-2})
HBAGDIFF	difference between minimum and maximum herbaceous aboveground biomass
HBAGMIN	minimum expected herbaceous aboveground biomass (g m^{-2})
HBBG	herbaceous belowground biomass (g m^{-2})
HBNREQ	annual N demand by herbaceous plants (g N m^{-2} yr^{-1})
HBRT:ST	herbaceous root biomass:shoot biomass ratio
HBRTMORT	annual mortality of herbaceous root biomass (g m^{-2} yr^{-1})
HBRTT	turnover time of herbaceous roots (years)
HBTF	fraction of N demand met by internal translocation in herbaceous plants
INITSOC	initial soil C stock (g C m^{-2})
LEAFTT	turnover time of tree foliage (years)
LFBMSS	tree leaf biomass (g m^{-2})
LFLIT	annual leaf litterfall (g m^{-2})
LITIN	total annual aboveground and belowground litter production (g m^{-2} yr^{-1})
NMINFLUX	annual flux of net soil N mineralization (g N m^{-2} yr^{-1})
NMINRATE	annual rate of net soil N mineralization (yr^{-1})
PEN	potential excess N (g N m^{-2})
PLNTNREQ	total annual plant N demand (g N m^{-2} yr^{-1})
REMOVAL	percent of aboveground woody biomass removed by thinning or harvest
RTMORT	annual removal of belowground biomass by tree root mortality (g m^{-2} yr^{-1})
RTTT	turnover time of tree root biomass (years)
SOC	soil C stock (g C m^{-2})
SOCIN	annual soil C inputs (g C m^{-2} yr^{-1})

SOCOUT	annual soil C losses (g C m^{-2} yr^{-1})
SOCTT	turnover time of soil C stock (years)
SOILC:N	soil C:N ratio
SOILN	soil N stock (g N m^{-2})
TARGET	maximum aboveground woody biomass (g m^{-2})
TRNREQ	annual N demand of forest trees (g N m^{-2} yr^{-1})
WDYTF	fraction of annual N demand met by internal translocation in trees
SERDP	Strategic Environmental Research and Development Program
SEMP	SERDP Ecosystem Management Program

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SUMMARY

The objective of this work was to use a simple compartment model of soil carbon (C) and nitrogen (N) dynamics to predict forest recovery on degraded soils and forest sustainability, following recovery, under different regimes of prescribed fire and timber management. This report describes the model and a model-based analysis of the effect of prescribed burning and forest thinning or clearcutting on stand recovery and sustainability at Fort Benning, GA. I developed the model using Stella® Research Software (High Performance Systems, Inc., Hanover, NH) and parameterized the model using data from field studies at Fort Benning, literature sources, and parameter fitting. The model included (1) a tree biomass submodel that predicted aboveground and belowground tree biomass, (2) a litter production submodel that predicted the dynamics of herbaceous aboveground and belowground biomass, (3) a soil C and N submodel that predicted soil C and N stocks (to a 30 cm soil depth) and net soil N mineralization, and (4) an excess N submodel that calculated the difference between predicted plant N demands and soil N supplies. There was a modeled feedback from potential excess N (PEN) to tree growth such that forest growth was limited under conditions of N deficiency.

Two experiments were performed for the model-based analysis. In the first experiment, forest recovery from barren soils was predicted for 100 years with or without prescribed burning and with or without timber management by thinning or clearcutting. In the second experiment, simulations began with 100 years of predicted forest growth in the absence of fire or harvesting, and sustainability was predicted for a further 100 years either with or without prescribed burning and with or without forest management. Four performance variables (aboveground tree biomass, soil C stocks, soil N stocks, and PEN) were used to evaluate the predicted effects of timber harvesting and prescribed burning on forest recovery and sustainability.

Predictions of forest recovery and sustainability were directly affected by how prescribed fire affected PEN. Prescribed fire impacted soil N supplies by lowering predicted soil C and N stocks which reduced the soil N pool that contributed to the predicted annual flux of net soil N mineralization. On soils with inherently high N availability, increasing the fire frequency in combination with stand thinning or clearcutting had little effect on predictions of forest recovery and sustainability. However, experiments with the model indicated that combined effects of stand thinning (or clearcutting) and frequent prescribed burning could have adverse effects on forest recovery and sustainability when N availability was just at the point of limiting forest growth. Model predictions indicated that prescribed burning with a 3-year return interval would decrease soil C and N stocks but not adversely affect forest recovery from barren soils or sustainability following ecosystem recovery. On soils with inherently low N availability, prescribed burning with a 2-year return interval depressed predicted soil C and N stocks to the point where soil N deficiencies prevented forest recovery as well as forest sustainability following recovery.

Keywords: soil carbon and nitrogen, forest recovery, forest sustainability, ecosystem modeling, fire, thinning, clearcutting, degraded soils, military land management

1. INTRODUCTION

Military land managers are faced with the challenge of using a given amount of land for the purpose of training and troop readiness. This mission must be accomplished in an ecologically sound manner that meets military requirements and, at the same time, promotes ecosystem sustainability so that military activities are not compromised by a degraded landscape. One aspect of ecosystem sustainability is preserving natural resources on a landscape that may be intensively used for military training. A second aspect involves restoring terrestrial ecosystems on soils that have been degraded by continuous military use. Military activities that can potentially result in degraded lands include the use of heavy weapons, and off-road wheeled, and tracked vehicle training.

Disturbance of soil physical properties and soil structure are commonly reported effects associated with use of heavy machinery in forestry (Hatchell et al., 1970) and military training (Iverson et al., 1981; Prose, 1985; Braunack, 1986; Thurow et al., 1993). At Fort Benning, GA, field training with tracked vehicles has resulted in an overall loss of soil quality at some training sites (Garten et al., 2003). Barren, heavily disturbed soils have negligible O-horizons, lower soil N availability, and lower C and N stocks than soils subject to minimal military use (Garten and Ashwood, 2004a). In some environments, the effects of soil disturbance by military vehicles can persist for decades (e.g., Iverson et al., 1981). This leads to questions about how land management practices affect ecosystem recovery following soil disturbance.

Land management at Fort Benning includes the use of prescribed fire and tree thinning or clearcutting to promote healthy forests. For example, prescribed fire is a common land management practice to clear herbaceous and woody shrubs from beneath forest stands because it improves access for military training and timber management and reduces the fuel load that might otherwise contribute to wildfires. Burning also helps to restore and maintain fire-dependent plant communities (e.g., longleaf pine) that are important habitat for threatened and endangered species at Fort Benning. One of the installation's forest management goals is to restore fire-dependent longleaf pine communities, and to meet this goal ≈10,000 ha are subject to prescribed burning each year. Each training compartment at Fort Benning is burned, on average, once every 3 years (the range is once a year to once every 5 years). The red-cockaded woodpecker recovery plan requires controlled burns approximately every 3 years in habitat used by that endangered species. In addition to restoration of longleaf pine, timber management at Fort Benning generally involves thinning pine and pine/hardwood forests (≈2,800 ha yr^{-1}) and clearcutting of diseased or insect-damaged stands. Current forest management guidelines include maintenance of a 100 year harvest rotation for healthy loblolly and shortleaf pine if threatened or endangered wildlife species are not adversely impacted by forest removal (Swiderek et al., 2002).

The challenge of military land use in the southeastern US is further complicated by the potential effects of prescribed fire and timber management on highly weathered, coarse-textured Ultisols. The complexity of land management on such nutrient poor soils raises questions about how possible interactions between soil N availability, prescribed burning, and forest harvesting may limit ecosystem recovery on degraded land or prevent ecosystem sustainability following forest recovery. These questions are difficult to answer with field experiments because: (1) the study of ecosystem recovery requires a prolonged period of measurements, and (2) replication of such long-term experiments can be problematic. The objective of this research was to use a simple compartment model of soil carbon (C) and nitrogen (N) dynamics to predict forest recovery on degraded soils and forest sustainability, following recovery, under different regimes of prescribed fire and timber management. This report describes the model and a model-based analysis of the effect of prescribed burning and forest thinning or clearcutting on stand recovery and sustainability for two different soil types at Fort Benning, GA. The model was parameterized for a generalized forest cover and it is potentially useful for predicting both the recovery of forest biomass and soil quality on degraded land.

2. METHODS

2.1 STUDY SITE

Fort Benning was established by the US military near Columbus, GA, in 1918 and considerable additional land was added to the installation in 1941. The number of troops onsite ranges between 18,000 and 23,000 annually. The land area at Fort Benning is ≈73,600 ha, and land use prior to acquisition by the US Government was primarily a mixture of agriculture and forestry. Current land cover at Fort Benning is ≈49% mixed forest, 25% deciduous forest, 10% barren or developed land, 7% evergreen forest, 6% herbaceous grassland, 2% shrub land, and 1% water (Jones and Davo, 1997). Mean annual temperature in the Columbus area is 18.3 °C and mean annual precipitation is 130 cm.

Soils at the site are highly weathered Ultisols, mostly of Coastal Plain origin but with some minor inclusion of alluviums derived from the Piedmont ecological unit to the north. The two dominant Coastal Plain ecological units that cover most of the installation are Sand Hills and Upper Loam Hills. The major soil series associated with these soil units are Ailey loamy coarse sand, Cowarts loamy sand, Nankin sandy clay loam, Pelion loamy sand, Troup, Troup loamy fine sand, Vaucluse, and Vaucluse sandy loam. Sands and loamy sands are common on upland sites while sandy loams and sandy clay loams are commonly found in valleys and riparian areas. Further details on the biology, geology, physical setting, and history of Fort Benning are available elsewhere (Jones and Davo, 1997).

2.2 MODEL STRUCTURE

2.2.1 Software Platform

I developed the model using Stella® Research Software (High Performance Systems, Inc., Hanover, NH) Version 7.0.2 for Power Macintosh computers. The first-order differential equations, of the general form dx/dt = fluxes into a compartment - fluxes from a compartment, were solved on an annual time step with Euler's integration method. Although Euler's method is less precise than Runge-Kutta methods, its use was mandated by certain "if-then" type statements in the model. In this report, model equations are presented in Stella® language format. This will facilitate reproduction of the model by other investigators using Stella® software. Throughout this report, variable names are identified by abbreviations with all capital letters.

2.2.2 Tree Biomass Submodel

The tree biomass submodel (Fig. 1) had two state variables: (1) aboveground woody biomass (AGWB, g m^{-2}), and (2) belowground root biomass (BGWB, g m^{-2}). The change in AGWB was calculated as:

$$AGWB(t) = AGWB(t-dt) + (AGROWTH - HARVST) \bullet dt \quad [1]$$

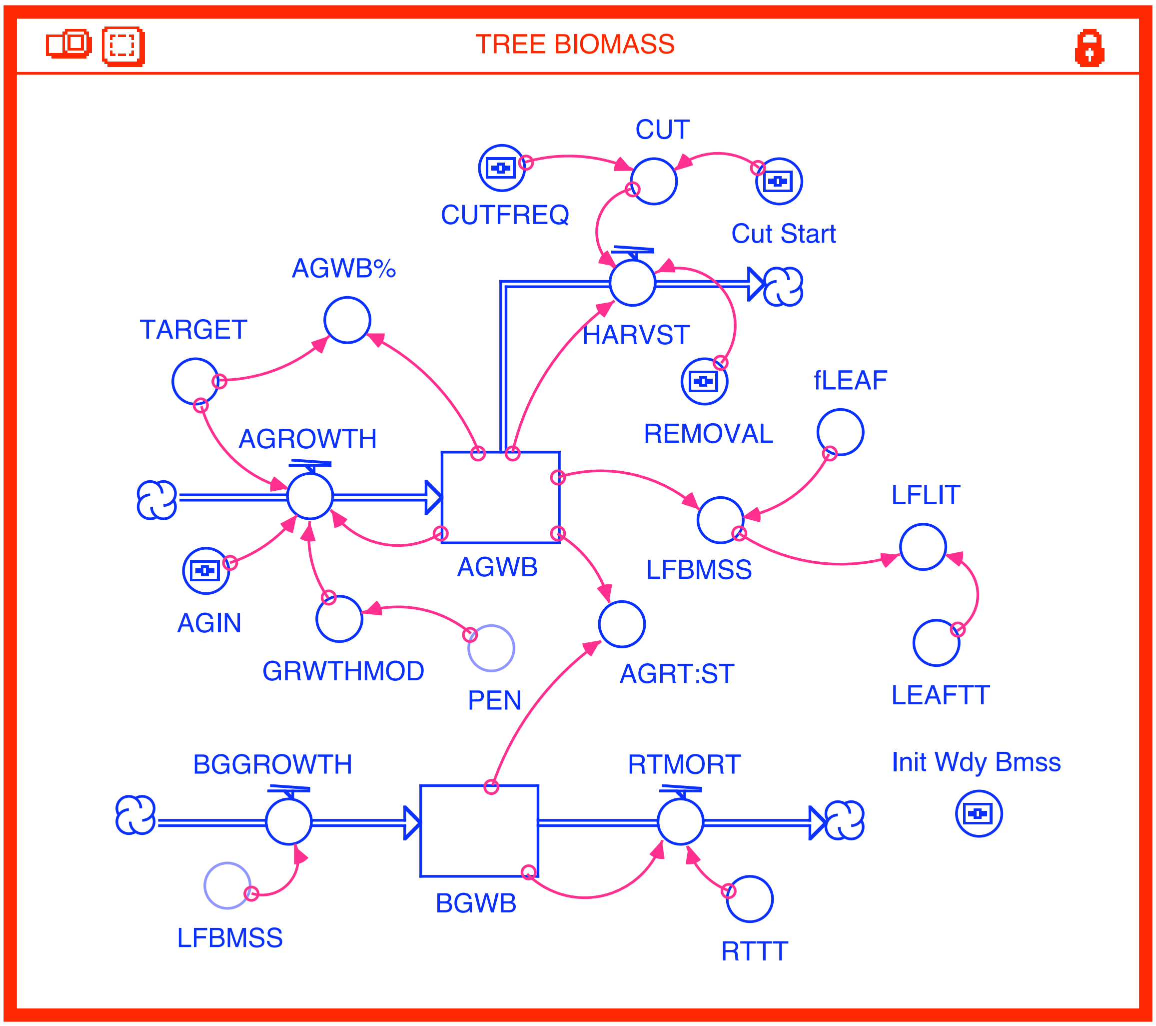

Fig. 1. Tree biomass submodel in Stella® model format.

where AGROWTH (g m^{-2} yr^{-1}) is the input to AGWB and HARVST is removal of AGWB by forest thinning or harvesting. AGROWTH was calculated as:

$$AGROWTH = (AGIN \bullet ((TARGET - AGWB)/(TARGET))) \bullet GRWTHMOD \quad [2]$$

where AGIN (g m^{-2} yr^{-1}) is the annual aboveground woody growth increment, TARGET (g m^{-2}) is the maximum aboveground woody biomass, and GRWTHMOD is a modifier that allows for the feedback of N availability on AGROWTH. GRWTHMOD was represented as:

$$GRWTHMOD = \text{if } PEN < 0.0 \text{ then } 0 \text{ else } 1 \quad [3]$$

where PEN (g N m^{-2}) is potential excess N. The modifier set AGROWTH to zero if PEN was less than zero. Otherwise, AGROWTH assumed its full calculated value.

The thinning or harvest of AGWB was calculated as:

$$HARVST = \text{if } CUT = 1 \text{ then } (AGWB \bullet (REMOVAL/100)) \text{ else } 0 \quad [4]$$

where REMOVAL was the percent of AGWB removed. HARVST was triggered when CUT = 1 and the latter variable was represented by pulse function with a recurring harvest frequency (CUTFREQ) in the model. When CUTFREQ assumed any value other than 1, there was no loss of AGWB through harvesting.

Tree leaf biomass (LFBMSS, g m^{-2}) was calculated as the product of AGWB and the fraction of aboveground biomass that was foliage (fLEAF). Annual leaf litterfall (LFLIT, g m^{-2} yr^{-1}) was calculated as:

$$LFLIT = LFBMSS \bullet (1/LEAFTT) \quad [5]$$

where LEAFTT was the turnover time (years) of tree foliage.

The change in BGWB (g m^{-2}) was calculated as:

$$BGWB(t) = BGWB(t-dt) + (BGGROWTH - RTMORT) \bullet dt \quad [6]$$

where BGROWTH (g m^{-2} yr^{-1}) is the input to BGWB and RTMORT (g m^{-2} yr^{-1}) is the removal of BGWB by root mortality. BGROWTH was assumed to be equivalent to LFBMSS. RTMORT was calculated as:

$$RTMORT = BGWB \bullet (1/RTTT) \quad [7]$$

where RTTT was the turnover time (years) of root biomass.

For convenience, two other variables were calculated by the model: (1) AGRT:ST or the ratio of BGWB to AGWB, and (2) AGWB% or the amount of AGWB expressed as a percentage of the TARGET aboveground woody biomass. At steady state, AGRT:ST was ≈0.25.

2.2.3 Litter Production Submodel

The litter production submodel (Fig. 2) represented the dynamics of herbaceous aboveground and belowground biomass and calculated litter inputs to the soil C submodel. Herbaceous aboveground biomass (HBAG, g m^{-2}) was calculated as:

$$HBAG = ((1-(AGWB/TARGET)) \cdot HBAGDIFF) + HBAGMIN \qquad [8]$$

where HBAGDIFF (g m^{-2}) is the difference between minimum and maximum expected herbaceous aboveground biomass and HBAGMIN (g m^{-2}) is the expected minimum herbaceous aboveground biomass. The equation makes herbaceous aboveground biomass decline from a maximum to a minimum value with the development of AGWB. Herbaceous belowground biomass (HBBG, g m-2) was calculated as the product of HBAG and a root:shoot ratio (HBRT:ST). The mortality of herbaceous root biomass (HBRTMORT, g m^{-2} yr^{-1}) was calculated as:

$$HBRTMORT = HBBG \cdot (1/HBRTT) \qquad [9]$$

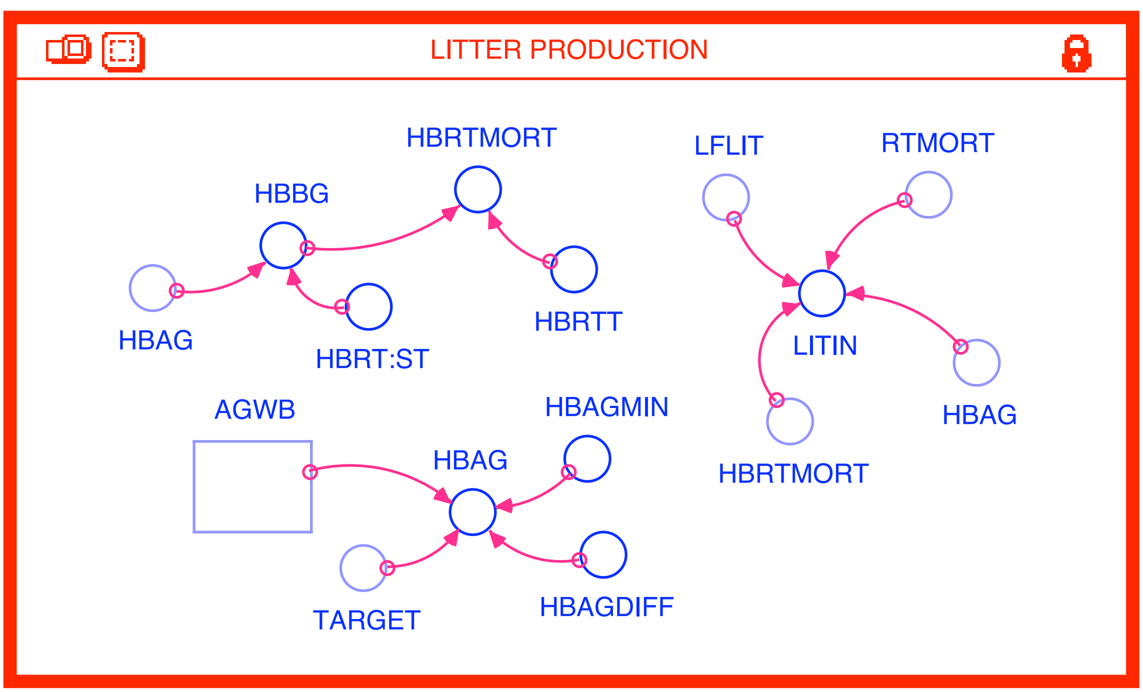

Fig. 2. Litter production submodel in Stella® model format.

where HBRTT (years) is the turnover time of herbaceous roots. Finally, total annual aboveground and belowground litter production (LITIN, g m^{-2} yr^{-1}) was calculated as:

$$LITIN = LFLIT + HBAG + HBRTMORT + RTMORT \qquad [10]$$

The latter equation assumes that each year all herbaceous aboveground biomass dies and is returned to the surface soil.

2.2.4 Soil Carbon and Nitrogen Submodel

Soil C (SOC, g C m^{-2}) was represented by a single compartment that included both O-horizons and the surface 30 cm of mineral soil (Fig. 3). The change in SOC was calculated as:

$$SOC(t) = SOC(t-dt) + (SOCIN - SOCOUT - FIRELOSS) \bullet dt \qquad [11]$$

where SOCIN (g C m^{-2} yr^{-1}) denotes soil C inputs, SOCOUT (g C m^{-2} yr^{-1}) denotes soil C losses through organic matter decomposition, and FIRELOSS (g C m^{-2} yr^{-1}) is the amount of soil C lost as a result of prescribed burning. SOCIN was calculated as the product of plant tissue C concentration (0.5 g C g^{-1}) and LITIN. SOCOUT was calculated as:

$$SOCOUT = SOC \bullet (1/SOCTT) \qquad [12]$$

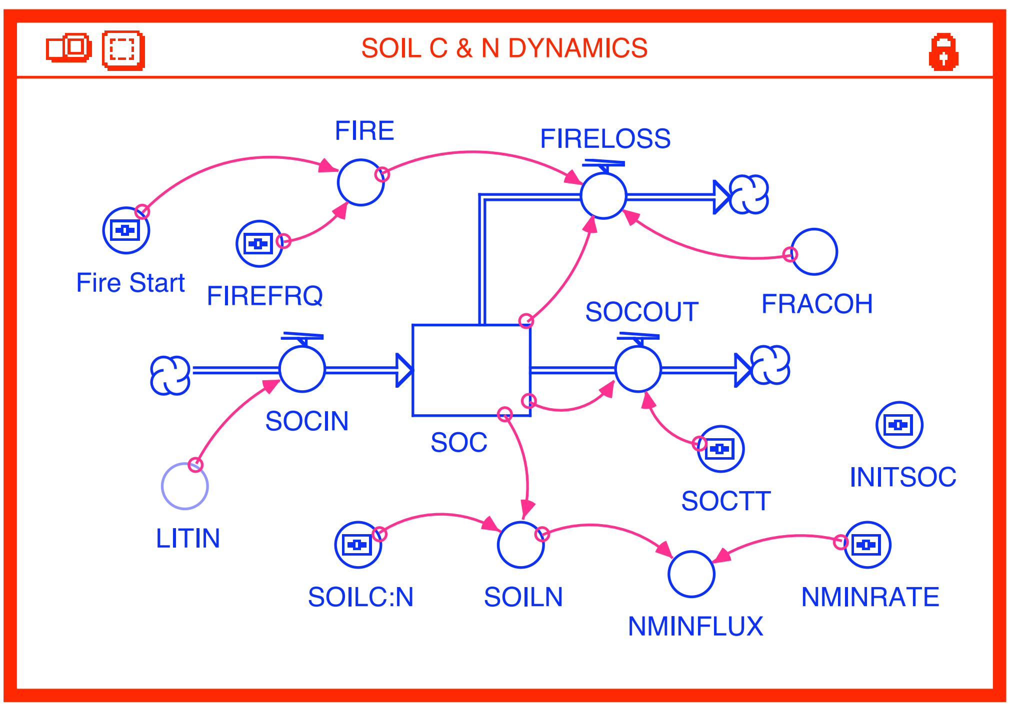

Fig. 3. Soil C and N submodel in Stella® model format.

where SOCTT (years) is the turnover time of soil C. Initial soil C stocks (INITSOC, g C m^{-2}) were set at the beginning of a model run accordant with the starting soil quality.

FIRELOSS in the soil C and N submodel was calculated as:

$$FIRELOSS = SOC \cdot FIRE \cdot FRACOH \qquad [13]$$

where FRACOH is the fraction of soil C in the O-horizon that is lost during a prescribed burn. FIRE was represented by a pulse function that set FIRE equal to unity whenever prescribed burning occurred (otherwise FIRE = 0). A separate variable, FIREFRQ (years), was used to establish the return interval of prescribed burning.

Soil N (SOILN, g N m^{-2}) was calculated by dividing SOC by the soil C:N ratio (SOILC:N). The net flux of net soil N mineralization (NMINFLUX, g N m^{-2} yr^{-1}) or the amount of soil organic N that is annually transformed to NH_4-N and NO_3-N was calculated as the product of SOILN and the net soil N mineralization rate (NMINRATE, yr^{-1}).

2.2.5 Excess Nitrogen Submodel

Potential excess N (PEN, g N m^{-2}) was calculated as the difference between N inputs and outputs to a pool of plant-available soil N (Fig. 4). PEN was calculated as:

$$PEN = ATMN + NMINFLUX - PLNTNREQ \qquad [14]$$

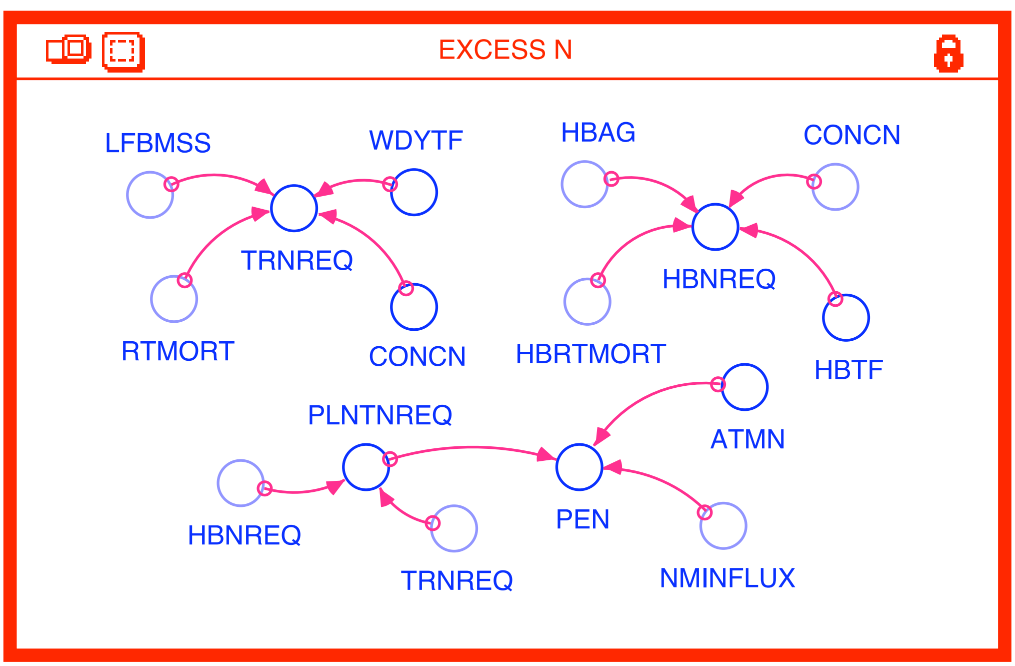

Fig. 4. Potential excess N submodel in Stella® model format.

where ATMN was total atmospheric N deposition (g N m^{-2} yr^{-1}), NMINFLUX is the net flux of soil N mineralization, and PLNTNREQ (g N m^{-2} yr^{-1}) is the annual plant N requirement.

The N demand of herbaceous plants (HBNREQ, g N m^{-2} yr^{-1}) was calculated as:

$$HBNREQ = ((HBAG + HBRTMORT) \bullet (CONCN/100) \bullet (1 - HBTF) \qquad [15]$$

where HBTF is the fraction of the N demand met by internal N translocation within herbaceous plants, and CONCN (g N g^{-1}) is the plant tissue N concentration. The equation assumes herbaceous aboveground biomass regrows annually. The N demand of forest trees (TRNREQ, g N m^{-2} yr^{-1}) was calculated as:

$$TRNREQ = (LFBMSS + RTMORT) \bullet (CONCN/100) \bullet (1 - WDYTF) \qquad [16]$$

where WDYTF is the fraction of N demand met by internal tree N translocation.

Total plant N demand (PLNTNREQ, g N m^{-2} yr^{-1}) was calculated as the sum of HBNREQ and TRNREQ. Equations for predicting HBNREQ and TRNREQ both assume: (1) belowground tree biomass production is in approximate balance with tree root mortality, and (2) new biomass production above and belowground has roughly the same tissue N concentration.

2.3 MODEL PARAMETERIZATION

The model was parameterized using information from field studies at Fort Benning (Garten and Ashwood, 2004a; 2004b), literature sources, and parameter fitting. Prior research (Garten and Ashwood, 2004b) indicated that soils with varying sand content had different predicted thresholds to ecosystem recovery at Fort Benning. Predicted thresholds to recovery were less on soils with more than 70% sand content, apparently due to higher relative rates of net soil N mineralization in more sandy soils. Consequently, the model was parameterized for soils with > 70% sand (identified as "more sandy" or "high soil N availability") and < 70% sand (identified as "less sandy" or "less soil N availability").

2.3.1 Tree Biomass Submodel

Parameters associated with the tree biomass submodel are presented in Table 1. Aboveground woody growth increment (AGIN) was set to 1000 g m^{-2} yr^{-1}. At this growth rate, predicted aboveground forest biomass (AGWB) reached 94% of steady state in ≈50 years. Maximum aboveground tree biomass (TARGET) was set to 18000 g m^{-2} which is in agreement with estimates of aboveground biomass in longleaf pine stands at Fort Benning (Garten and Ashwood, 2004b), in agreement with stand biomass in mature (45 year old) forests on the southern Piedmont (≈18000 g m^{-2} based on data in Johnson and Lindberg, 1992), and similar to aboveground biomass in loblolly pine after 50 to 60 years of stand development (≈21000 g m^{-2};

Table 1. Parameter values for variables in the tree biomass submodel			
Variable	Abbreviation	Units	Value
Aboveground woody growth increment	AGIN	g m^{-2} yr^{-1}	1000
Fraction of aboveground tree biomass in foliage	fLEAF	fraction	0.023
Harvest frequency	CUTFREQ	years	0 to 100[a]
Tree leaf turnover	LEAFTT	years	1
Percent of tree biomass removed at harvest	REMOVAL	%	0 to 99[a]
Turnover time of tree root biomass	RTTT	years	10
Maximum aboveground tree biomass	TARGET	g m^{-2}	18000
[a]Depending on the simulation, the value varied within the indicated range.			

Switzer et al., 1968). The maximum aboveground tree biomass selected for Fort Benning is also in the range of aboveground biomass densities of saw timber stands and forest stands in advanced stages of recovery (following forest clearing) in the eastern US (Brown et al., 1997). Depending on the simulation, the interval between stand harvests (CUTFREQ) varied from 0 to 100 years and the percent of tree biomass removed at harvest (REMOVAL) varied from 0 to 99%. It was assumed that stand thinning operations removed 50% of the aboveground tree biomass.

The fraction of aboveground tree biomass represented by foliage (fLEAF) was set to 0.023 based on data from southeastern forests (Johnson and Van Hook, 1989; Johnson and Lindberg, 1992) indicating that leaf biomass is typically 2 to 5% of total aboveground biomass. Using this fraction, the predicted steady state tree leaf biomass (LFBMSS) was 414 g m^{-2} which is in good agreement with estimates of annual leaf litterfall in the southeastern US (Sharpe et al., 1980). Tree leaf turnover time (LEAFTT) was assumed to be one year and the turnover time of tree root biomass (RTTT) was set to 10 years (Gill and Jackson, 2000).

2.3.2 Litter Production Submodel

Parameters associated with the litter production submodel are presented in Table 2. The minimum herbaceous aboveground biomass (HBAGMIN) was set to 150 g m^{-2} based on

Table 2. Parameter values for variables in the litter production submodel			
Variable	Abbreviation	Units	Value
Difference between minimum and maximum herbaceous aboveground biomass	HBAGDIFF	g m^{-2}	200
Minimum herbaceous aboveground biomass	HBAGMIN	g m^{-2}	150
Herbaceous root:shoot ratio	HBRT:ST	fraction	1
Herbaceous root turnover time	HBRTT	years	2

measured standing biomass of ground cover in longleaf pine stands on xeric sites (Kirkman et al., 2001). In the model, herbaceous aboveground biomass increased with decreasing aboveground tree biomass, and HBAGMIN was the predicted amount of herbaceous aboveground biomass in mature forests. The difference between minimum and maximum herbaceous aboveground biomass was set to 200 g m^{-2}. Other studies (Odum, 1960) indicate that ≈360 g m^{-2} is a reasonable value for aboveground oldfield biomass in Georgia. The model predicted 350 g m^{-2} herbaceous aboveground biomass when aboveground tree biomass was at its minimum. The root:shoot ratio for herbaceous biomass (HBRT:ST) was set to 1.0 based on studies by Kelly (1975) who measured root:shoot ratios of 0.78 and 1.4 in two east Tennessee old field communities. The turnover time of herbaceous plant roots (HBRTT) was set to 2 years on the basis of an average root turnover time in grasslands (Gill and Jackson, 2000).

2.3.3 Soil Carbon and Nitrogen Submodel

Parameters associated with the soil C and N submodel are presented in Table 3. Prior field work (Garten and Ashwood, 2004b) indicated that Fort Benning soils have a high sand content (on average, 70% sand; two-thirds of 129 soil samples collected onsite had a sand content that exceeded 70%). The turnover time of soil C (SOCTT) in these coarse textured soils, that offer little physical protection from decomposition of soil organic matter, was set to 10 years. Both net soil N mineralization rates (NMINRATE) and soil C:N ratios (SOILC:N) varied depending on soil type (Garten and Ashwood, 2004b). Based on field measurements, the rate of net soil N mineralization was 0.026 and 0.064 yr^{-1}, respectively, on less sandy (< 70% sand content) and more sandy (> 70% sand content) soils. The mean soil C:N ratio was 21 on less sandy soils and 36 on more sandy soils (Garten and Ashwood, 2004b).

Table 3. Parameter values for variables in the soil C and N submodel

Variable	Abbreviation	Units	Value	
			Less sandy soils	More sandy soils
Frequency of prescribed fire	FIREFREQ	years	0 to 3[a]	0 to 3[a]
Fraction of soil C lost in fire	FRACOH	fraction	0.12	0.12
Initial soil C on barren land	INITSOC	g C m^{-2}	630	630
Net soil N mineralization	NMINRATE	$year^{-1}$	0.026	0.064
Soil C turnover time	SOCTT	years	10	10
Soil C:N ratio	SOILC:N	ratio	21	36

[a]Depending on the simulation,the value was varied within the indicated range

The frequency of prescribed fire was varied depending on the model scenario. The fraction of soil C in O-horizons at Fort Benning is ≈12% (Garten and Ashwood, 2004b) and the impact of ground fires is limited primarily to O-horizons. For the purpose of simulating fire effects, it was assumed that the fraction of soil C lost during prescribed burning (FRACOH) was equivalent to the fraction of soil C residing in the O-horizon. Each simulation discussed in this report started from barren land, and initial soil C stocks (INITSOC) were set to 630 g C m^{-2} based on data from 14 barren sites at Fort Benning (Garten and Ashwood, 2004b).

2.3.4 Excess Nitrogen Submodel

The parameters associated with the excess N submodel are presented in Table 4. Data on atmospheric N deposition were obtained from the National Atmospheric Deposition Program[1] for monitoring stations in the vicinity of Fort Benning. Annual wet only N deposition (0.35 g N m^{-2} yr^{-1}) was converted to total N deposition using a factor of 2.0 that was derived from data collected in the southeastern US (Lovett and Lindberg, 1993). It was assumed that plant tissue C concentrations were 0.5 g C g^{-1} and plant tissue N concentrations were set to 1% based on data from different sources (Birk and Vitousek, 1986; Yin, 1993). Nitrogen concentrations in roots were assumed to be the same as those in foliage based on studies of loblolly pine on the upper Coastal Plain (Birk and Vitousek, 1986).

Table 4. Parameter values for variables in the excess N submodel

Variable	Abbreviation	Units	Value
Atmospheric N deposition	ATMN	g N m^{-2} yr^{-1}	0.7
Plant tissue N concentration	CONCN	%	1.0
Fraction of herbaceous plant N demand met through internal N translocation	HBTF	fraction	0.5
Fraction of tree N demand met through internal N translocation	WDYTF	fraction	0.5

Seasonal translocation of N in trees (Luxmoore et al., 1981; Ostman and Weaver, 1982) and herbaceous plants (Li et al., 1992) is a well known process. Its overall importance to plant nutrition is that under circumstances where soil N supplies limit plant growth, N demands for new tissue production are met through a redistribution of internal plant N. Studies of loblolly pine on sandy soils indicate that about 50% of the foliar N is translocated to wood prior to leaf senescence (Birk and Vitousek, 1986). Under conditions of low soil N availability, ≈50% of the N required for production of new biomass in herbaceous vegetation may be derived from internal N translocation (e.g., Li et al., 1992). Therefore, in the absence of site-specific information, the translocation factor was set at 50% for forest and herbaceous plant communities at Fort Benning.

[1] http://nadp.sws.uiuc.edu/

2.4 SENSITIVITY ANALYSIS

A sensitivity analysis was used to identify the parameters that had the greatest influence on model predictions. Measurement accuracy is most important for those parameters that have the greatest effect on model outputs. In addition, if variation in a parameter value does not alter model behavior, then steps to alter the process it represents might be of little use for promoting forest recovery and sustainability. Variance estimates are not well established for most parameters in the model. Therefore, the sensitivity analysis was performed by systematically varying each parameter in the model by ±20% of its base value, while holding all other parameters constant, and running the model to 100 years. The base value for NMINRATE and SOILC:N was set at 0.025 yr^{-1} and 20, respectively.

2.5 MODEL SCENARIOS

Both forest recovery and forest sustainability on less sandy and more sandy soils under different fire and timber management regimes was predicted with the model. Four performance variables (AGWB, SOC, SOILN, and PEN) were used to evaluate the predicted effects of harvesting and prescribed burning on forest recovery and sustainability. Two experiments were performed with the model. The first experiment tested the effect of prescribed burning and timber management on forest recovery from barren soils. The second experiment tested the effect of prescribed burning and timber management on forest sustainability following stand recovery from barren soils.

In the first experiment, forest recovery from barren soils was predicted for 100 years with or without prescribed burning and with or without forest management by thinning (50% REMOVAL) or clearcutting (99% REMOVAL). The time interval between harvests was fixed at 50 years. Each recovery scenario started with 300 g m^{-2} aboveground tree biomass, 350 g m^{-2} herbaceous aboveground biomass, and 630 g soil C m^{-2}. Data for the performance variables were recorded at the end of 100 years and model predictions were compared among the various scenarios. In experiment 1, less sandy soils (i.e., < 70% sand) and more sandy soils (i.e., > 70% sand) represented soils with “low N availability” and “high N availability”, respectively.

In the second experiment, forest sustainability was predicted for a second 100 year cycle which followed 100 years of recovery in the absence of fire and timber management. In other words, after 100 years of forest growth, sustainability was predicted for a further 100 years either with or without prescribed burning and with or without forest management by thinning (50% REMOVAL) or clearcutting (100% REMOVAL). The time interval between harvests was fixed at 50 years. The initial conditions for aboveground tree biomass, herbaceous aboveground biomass, and soil C stocks were the same as in experiment 1. Data for the performance variables were recorded at the end of 200 years and the model predictions were compared among the various scenarios. As in experiment 1, less sandy soils (i.e., those with < 70% sand) and more sandy soils (i.e., those with > 70% sand) represented soils with “low N availability” and “high N availability", respectively. It is noted that these parameters are not necessarily those which Fort Benning forest managers are following or plan to follow in detail, but are believed to be representative of a realistic range of regionally observed practices.

3. RESULTS

3.1 SENSITIVITY ANALYSIS

The sensitivity analysis was used to examine the relative change in the four performance variables (AGWB, SOC, SOILN, and PEN) when each parameter value in the model was varied by ±20%. Predictions of aboveground tree biomass (AGWB) were most affected by a single parameter, maximum aboveground tree biomass (TARGET). A ±20% change in TARGET produced a proportional change in AGWB.

Predictions of soil C (SOC) were affected most by changes in SOCTT, fLEAF, TARGET, and LEAFTT. A ±20% change in each of the foregoing parameters produced a 5 to 20% change in predicted soil C stocks. Predictions of soil N (SOILN) were affected by the same set of parameters as predictions of SOC. However, the most important model parameter to predictions of soil N stocks was the soil C:N ratio (SOILC:N).

In order of relative importance, the most important parameters for prediction of PEN were SOILC:N, NMINRATE, SOCTT, CONCN, WDYTF, and LEAFTT. A ±20% change in each of the foregoing parameters produced more than a 20% change in predicted potential excess N (PEN). A 20% change in fLEAF, HBTF, TARGET, and ATMN produced a 5 to 20% change in predicted PEN. Potential excess N was a critical feedback on the recovery rate of aboveground tree biomass (AGWB), hence any change in parameters affecting PEN can be translated into changes in AGWB, soil C, and soil N.

3.2 FOREST RECOVERY FROM BARREN LAND

Predicted forest recovery from barren land on less sandy soils (with low N availability) with or without prescribed burning is illustrated in Figure 5. In the absence of prescribed fires, AGWB, SOC, SOILN, and PEN increased to steady state values in ≈100 years or less. At a fire return interval of 3 years, the recovery of AGWB was slowed because available soil N began to limit tree growth (i.e., PEN was intermittently > 0 and < 0). Predicted soil C and N stocks after 100 years were also substantially reduced with a fire return interval of 3 years. Increasing the fire return interval to 2 years, dramatically reduced predicted AGWB, SOC and SOILN, and indicated that PEN strongly limited forest recovery. The cause of the N limitation was the consumption of O-horizon C and N by prescribed fires.

Combined effects of prescribed burning and forest harvesting are summarized in Table 5.

Even with burning and harvesting together, predicted AGWB was > 90% of maximum aboveground tree biomass (i.e., TARGET) on soils with high soil N availability. On these latter soils, predicted C stocks ranged from 3256 to 3527 g C m^{-2} and predicted N stocks ranged from 93 to 101 g N m^{-2} (with a 3-year fire return interval). The predictions were within 25% of measured mean C stocks (3847 g C m^{-2}) and measured mean N stocks (118 g N m^{-2}) in more sandy soils at Fort Benning (Garten and Ashwood, 2004b).

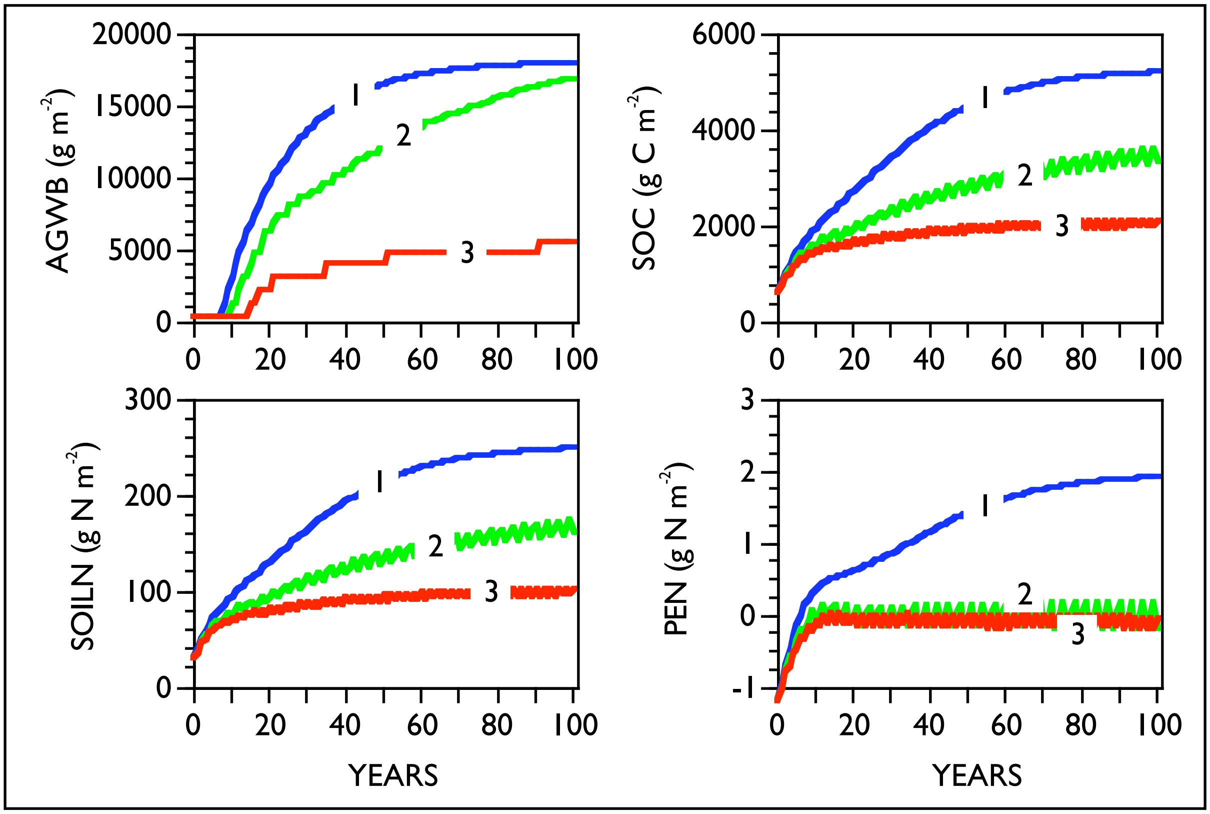

Fig. 5. Effect of prescribed burning on aboveground tree biomass (AGWB), soil C stocks (SOC), soil N stocks (SOILN), and potential excess N (PEN) on less sandy soils. Legend: (1) blue line = no fire; (2) green line = prescribed burn once every 3 years; (3) red line = prescribed burn once every 2 years.

Prescribed burning at 2- and 3-year intervals reduced predicted C and N stocks relative to the "no fire" scenario on soils with both low and high N availability (Table 5). On soils with low N availability, increased fire frequency had more effect on predicted forest recovery than stand thinning or clearcutting (50 year rotation). Prescribed burning at a 2- and 3-year return interval dramatically reduced predicted PEN which impacted predicted forest recovery. When burning occurred every other year, predicted aboveground tree biomass (AGWB) was reduced by ≈70%. On soils with low N availability, predicted C stocks ranged from 3089 to 3334 g C m^{-2} and predicted N stocks ranged from 147 to 159 g N m^{-2} (if the fire return interval was set to 3 years). The predictions were within 20% of measured mean C stocks (3709 g C m^{-2}) and measured mean N stocks (173 g N m^{-2}) in less sandy soils at Fort Benning (Garten and Ashwood, 2004b).

Table 5. Effect of harvesting (0, 50, or 99% removal of AGWB) and frequency of prescribed burning (FIREFREQ) on predicted recovery of aboveground forest biomass (AGWB, g m^{-2}), soil C stocks (SOC, g C m^{-2}), soil N stocks (SOILN, g N m^{-2}), and potential excess N (PEN, g N m^{-2}) on soils with low and high N availability (experiment 1). The time interval between thinning (50% removal) or clearcutting (99% removal) was 50 years. The predicted values were summarized following a 100 year model run.									
REMOVAL %	FIREFREQ (years)	Low soil N availability				High soil N availability			
		AGWB	SOC	SOILN	PEN	AGWB	SOC	SOILN	PEN
0	No fire	17913	5210	248	1.91	17931	5221	149	5.00
	3	16861	3334	159	-0.19	17931	3527	101	1.90
	2	5439	2141	102	-0.04	17927	3336	95	1.56
50	No fire	17414	4982	237	1.76	17420	4987	142	4.71
	3	16125	3218	153	-0.19	17420	3390	97	1.79
	2	4769	2117	101	-0.00	17418	3213	92	1.47
99	No fire	16921	4755	226	1.61	16921	4757	136	4.42
	3	15183	3089	147	-0.19	16921	3256	93	1.68
	2	5258	2113	101	-0.04	16921	3093	88	1.38

3.3 FOREST SUSTAINABILITY

Predicted forest recovery from barren land and sustainability on less sandy soils (with low N availability) with timber management and without prescribed burning is illustrated in Figure 6. The timber management regime was stand thinning (50% removal of AGWB) on a 50 year rotation after the first 100 years of forest recovery. In the absence of prescribed fires, forest recovery was sustainable (i.e., predicted AGWB repeatedly returned to the maximum aboveground tree biomass following forest thinning). Predicted soil C and N stocks exhibited minor fluctuations that were related to changes in soil C inputs following tree removal.

Addition of a 3-year schedule of prescribed burning to the timber management regime did not seriously impact predicted AGWB even though predicted soil C and N stocks were dramatically reduced (Table 6). Prescribed burning caused predicted PEN to fluctuate near zero but there was enough N to allow recovery of predicted AGWB following forest thinning. When the fire frequency increased to once every 2 years, predicted AGWB declined after each stand thinning and predicted SOC, SOILN, and PEN declined over time (Fig. 6). The experiment indicated that some combinations of prescribed fire and timber management may preclude sustainable forest ecosystems on soils with low N availability. In particular, a schedule of prescribed burning once every 2 years plus forest thinning or clearcutting on a 50 year rotation could result in a failure of forest stands to recover to their maximum aboveground tree biomass.

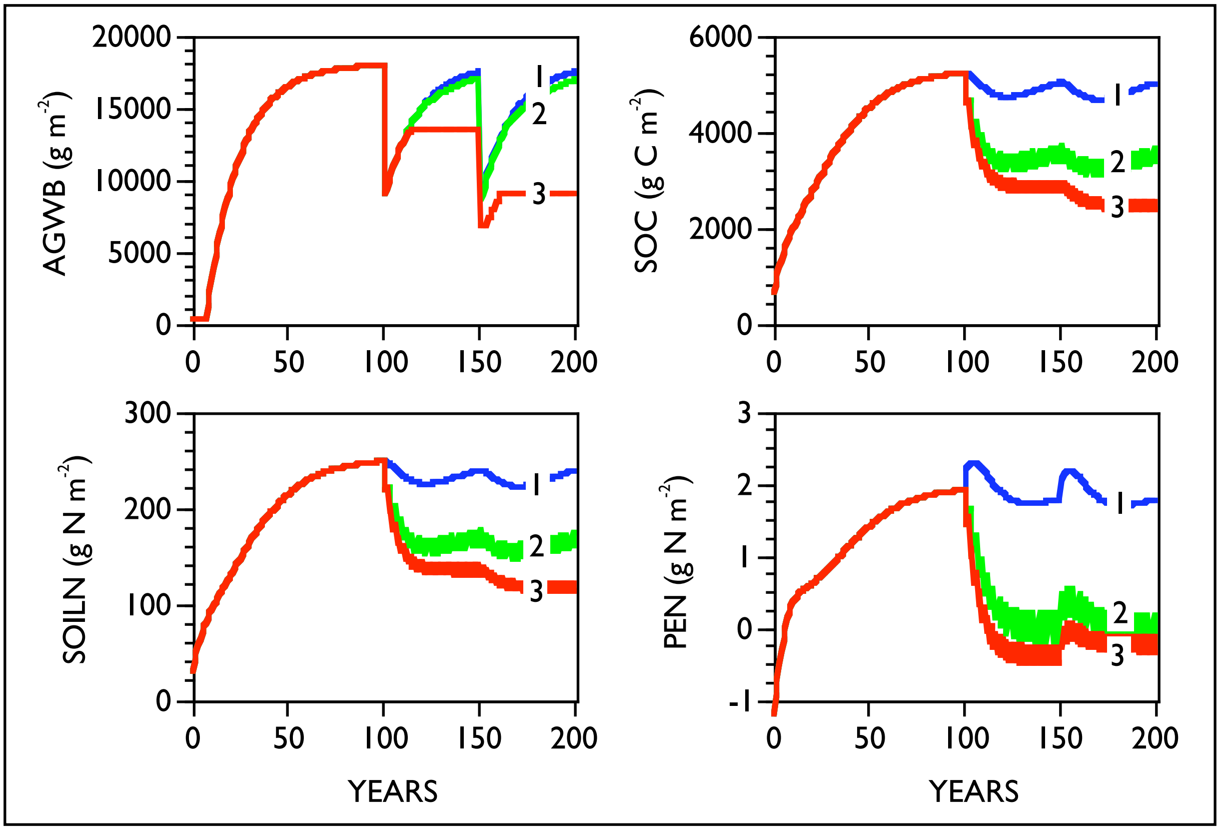

Fig. 6. Effect of prescribed burning and timber management (50% forest thinning at 100 and 150 years), following forest recovery, on aboveground tree biomass (AGWB), soil C stocks (SOC), soil N stocks (SOILN), and potential excess N (PEN) on less sandy soils. Legend: (1) blue line = no fire; (2) green line = prescribed burn once every 3 years, (3) red line = prescribed burn once every 2 years.

4. DISCUSSION

Prior research (Garten and Ashwood, 2004b) indicates there are four factors important to ecosystem recovery on degraded soils at Fort Benning: (1) initial amounts of aboveground biomass, (2) initial soil C stocks, (3) relative recovery rates of aboveground biomass, and (4) soil sand content. These same factors are also important in the current model-based analysis of the effects of prescribed fire and timber management on forest recovery and sustainability. Although other initial conditions are possible, recovery from barren soils was selected as the initial condition for both experiments with the model. The latter scenario represented an extreme type of ecosystem restoration that predicted high demands on soil N supplies by forest growth. Soil C and N stocks are greatly reduced in barren soils at Fort Benning which makes ecosystem recovery more difficult than when initial conditions for soil resemble those in less disturbed environments. The model indicates that if recovery rates are too high, forest growth was down regulated through feedbacks on potential excess N. Soil type and its relationship to soil N availability, as represented by less sandy and more sandy soils, was also a critical determinant of predicted forest recovery and sustainability under different regimes of prescribed burning and timber management.

Table 6. Effect of harvesting (0, 50, or 99% removal of AGWB) and frequency of prescribed burning (FIREFREQ) on predicted sustainability of aboveground forest biomass (AGWB, g m^{-2}), soil C stocks (SOC, g C m^{-2}), soil N stocks (SOILN, g N m^{-2}), and potential excess N (PEN, g N m^{-2}) on soils with low and high soil N availability (experiment 2). The time interval between thinning (50% removal) or clearcutting (99% removal) was 50 years. Treatments did not start until after 100 years of recovery and the predicted values were summarized after 200 years.

REMOVAL %	FIREFREQ (years)	Low soil N availability				High soil N availability			
		AGWB	SOC	SOILN	PEN	AGWB	SOC	SOILN	PEN
0	No fire	18000	5265	251	1.95	18000	5265	146	4.79
	3	17999	3549	169	-0.17	18000	3549	99	1.74
	2	17945	3352	160	-0.41	18000	3357	93	1.40
50	No fire	17438	5002	238	1.78	17438	5002	139	4.48
	3	16852	3500	167	0.01	17438	3397	94	1.62
	2	6003	2232	106	-0.04	17438	3220	89	1.31
99	No fire	16920	4759	227	1.62	16920	4759	132	4.18
	3	15464	3128	149	-0.19	16920	3631	90	1.51
	2	7301	2351	112	-0.08	16920	3094	86	1.22

The sensitivity analysis indicated that potential excess N, which was the contributing feedback from soil C and N dynamics to forest growth, was most sensitive to changes in soil C:N ratios and net soil N mineralization. Differences in these two soil properties were captured by considering two broad soil categories. More sandy soils (i.e., those with > 70% sand content) exhibit higher relative rates of net soil N mineralization than less sandy soils (i.e., those with < 70% sand content) at Fort Benning (Garten and Ashwood, 2004b). Even though soil N stocks are less on more sandy soils (due to their higher soil C:N ratios), more sandy soils have a higher estimated annual flux of net soil N mineralization than less sandy soils. The mineralization process contributed to increased levels of predicted PEN on more sandy soils (see Table 5). With a 3-year fire return interval, the predicted annual flux of net soil N mineralization in the model was ≈4 g N m^{-2} which is several times greater than *in situ* measurements of net soil N mineralization (0.5 to 1.2 g N m^{-2} yr^{-1}) under longleaf pine in southwestern Georgia (Wilson et al., 1999).

More sandy soils under perennial vegetation at Fort Benning have a significantly greater amount of soil C in particulate organic matter (Garten and Ashwood, 2004b) which is a highly labile C pool that is important to N availability, particularly in sandy soils (Hook and Burke, 2000; Willson et al., 2001). Greater amounts of labile soil organic matter may be one factor contributing to higher potential net soil N mineralization and elevated predictions of PEN in more sandy soils at Fort Benning. The sensitivity analysis indicated that model predictions could be further improved through more accurate measurements of net soil N mineralization and soil C:N ratios. As indicated above, *in situ*, site-specific measurements may provide different estimates of net soil N mineralization than the aerobic laboratory incubations on which NMINRATE was based for the purposes of the model. Several other variables also exerted an important control on PEN and thus potentially affect predictions of forest recovery and sustainability. Some, like plant tissue N concentrations, are more easily measured than others, like soil C turnover times and within plant N translocation.

Numerous studies (Neary et al., 1999; Wan et al., 2001) have examined the effects of prescribed burning on the sustainability of forest soil C and N reserves. Prescribed fires can substantially reduce O-horizon C and N stocks but they generally have no significant effect on mineral soil C and N. For example, Binkley et al. (1992) reported that the cumulative effects of 30 years of prescribed burning in Coastal Plain pine forests were generally limited to reduced C and N stocks in the forest floor. Prescribed burning may temporarily increase soil N availability and thereby promote establishment of herbaceous ground covers that can eventually stabilize burned areas. However, as the current model indicates, N losses from the forest floor as a consequence of prescribed fire may be significant to forest recovery when soils are nutrient poor and plant N demands are approximately in balance with soil N supply. Under such circumstances, prescribed fires may lower soil C and N stocks and create N deficiencies that limit forest recovery. By comparison, forest harvesting in the absence of prescribed burning had only a minor effect on soil C and N (Table 5). This result is similar to that from a model-based analysis of the effects of prescribed fire and forest harvesting on regrowth of Eucalyptus stands that indicated fire frequency had a greater effect on stand N balance than forest harvesting (McMurtrie and Dewar, 1997). Experiments with the current model also indicate that forest recovery and sustainability are more sensitive prescribed fire regimes than to forest thinning or clearcutting.

In summary, predictions of forest recovery and sustainability were directly influenced by how prescribed fire affected potential excess N or the difference between soil N supply and plant N demand. In the model, prescribed fire impacted soil N supplies by lowering soil C and N stocks which reduced the soil N pool that contributed to the predicted annual flux of net soil N mineralization. On soils with high soil N availability, increasing fire frequency in combination with stand thinning or clearcutting had little effect on predictions of forest recovery and sustainability. However, the model indicated that combined effects of stand thinning and frequent prescribed burning could have adverse effects on forest recovery and sustainability when soil N availability was just at the point of limiting forest growth. Model predictions indicated that prescribed burning with a 3-year return interval would decrease soil C and N stocks, but not adversely affect forest recovery from barren soils or forest sustainability following ecosystem recovery. On soils with low N availability, prescribed burning with a 2-

year return interval depressed predicted soil C and N stocks to the point where soil N deficiencies precluded forest recovery as well as forest sustainability following ecosystem recovery.

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APPENDIX E.
Effects of Heavy, Tracked-Vehicle Disturbance on Forest Soil Properties at Fort Benning, Georgia

ORNL/TM-2004/76

Effects of Heavy, Tracked-Vehicle Disturbance on Forest Soil Properties at Fort Benning, Georgia

May 2004

C.T. Garten, Jr., and T.L. Ashwood

Environmental Sciences Division

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EFFECTS OF HEAVY, TRACKED-VEHICLE DISTURBANCE ON FOREST SOIL PROPERTIES AT FORT BENNING, GEORGIA

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ABSTRACT

The purpose of this report is to describe the effects of heavy, tracked-vehicle disturbance on various measures of soil quality in training compartment K-11 at Fort Benning, Georgia. Pre-disturbance soil sampling in April and October of 2002 indicated statistically significant differences in soil properties between upland and riparian sites. Soil density was less at riparian sites, but riparian soils had significantly greater C and N concentrations and stocks than upland soils. Most of the C stock in riparian soils was associated with mineral-associated organic matter (i.e., the silt + clay fraction physically separated from whole mineral soil). Topographic differences in soil N availability were highly dependent on the time of sampling. Riparian soils had higher concentrations of extractable inorganic N than upland soils and also exhibited significantly greater soil N availability during the spring sampling.

The disturbance experiment was performed in May 2003 by driving a D7 bulldozer through the mixed pine/hardwood forest. Post-disturbance sampling was limited to upland sites because training with heavy, tracked vehicles at Fort Benning is generally confined to upland soils. Soil sampling approximately one month after the experiment indicated that effects of the bulldozer were limited primarily to the forest floor (O-horizon) and the surface (0-10 cm) mineral soil. O-horizon dry mass and C stocks were significantly reduced, relative to undisturbed sites, and there was an indication of reduced mineral soil C stocks in the disturbance zone. Differences in the surface (0-10 cm) mineral soil also indicated a significant increase in soil density as a result of disturbance by the bulldozer. Although there was some tendency for greater soil N availability in disturbed soils, the changes were not significantly different from undisturbed controls. It is expected that repeated soil disturbance over time, which will normally occur in a military training area, would simply intensify the changes in soil properties that were measured following a one-time soil disturbance at the K-11 training compartment.

The experiment was also useful for identifying soil measurements that are particularly sensitive to disturbance and therefore can be used successfully as indicators of a change in soil properties as a result of heavy, tracked-vehicle traffic at Fort Benning. Measurements related to total O-horizon mass and C concentrations or stocks exhibited changes that ranged from ≈25 to 75% following the one-time disturbance. Changes in surface (0-10 cm) mineral soil density or measures of surface soil C and N following the disturbance were less remarkable and ranged from ≈15 to 45% (relative to undisturbed controls). Soil N availability (measured as initial extractable soil N or N production in laboratory incubations) was the least sensitive and the least useful indicator for detecting a change in soil quality. Collectively, the results suggest that the best indicators of a change in soil quality will be found at the soil surface because there were no statistically significant effects of bulldozer disturbance at soil depths below 10 cm.

Key words: soil disturbance, tracked-vehicle training, soil C, soil N, particulate organic matter (POM), soil quality, ecological indicators, military land management

1. INTRODUCTION

Disturbance of soil physical properties and/or structure are commonly reported effects associated with the use of heavy, tracked vehicles in military training (Iverson et al., 1981; Prose, 1985; Braunack, 1986; Thurow et al., 1993; Ayers, 1994; Prosser et al., 2000; Belnap and Warren, 2002). In some environments, it has been shown that the effects of soil disturbance by tracked military vehicles can persist for decades (e.g., Iverson et al., 1981; Belnap and Warren, 2002). At Fort Benning, Georgia, field training with tracked vehicles has resulted in an overall loss of soil quality at some training sites where heavily disturbed, barren soils have negligible O-horizons, lower soil N availability, and lower soil C and N stocks than soils subject to minimal military use (Garten et al., 2003; Garten and Ashwood, 2004).

The purpose of this report is to describe the effects of heavy, tracked-vehicle disturbance on various measures of soil quality in training compartment K-11 at Fort Benning, Georgia. An experiment was performed by driving a D7 bulldozer through a mixed pine/hardwood forest. Pre-disturbance soil sampling was performed in both spring and autumn of the year preceding the disturbance to determine how site-specific topographic differences in soil quality would potentially affect post-disturbance sampling. The null hypothesis for the experiment was that disturbance caused by a heavy, tracked vehicle would not affect overall forest soil quality by changing soil C and N or by changing soil N availability. The bulldozer was considered here as a surrogate for a military, tracked vehicle of similar weight.

2. METHODS

2.1 STUDY SITE

The study site was located in the northeast corner of Fort Benning in a mature, second-growth, mixed evergreen-hardwood forest in training compartment K-11. The topography at the site included both poorly drained riparian zones adjacent to intermittent streams and well-drained upland forest stands. Forest management practices at the site have been implemented to promote growth and development of long-leaf pine (*Pinus palustris*).

The site's prior history included light military activity (i.e., infantry training) in addition to stand thinning and prescribed burning as part of the installation's timber management program. Prescribed burning to remove understory shrubs and hardwood saplings occurred in May 2002 and thinning occurred in late October 2002 (6 months prior to the experimental disturbance).

The disturbance was performed in May 2003 with several passes of a D7 bulldozer. The weight of the equipment was approximately 23,000 kg. The disturbance removed existing vegetation and forest floor organic matter from two rectangular areas (approximately 5 x 50 m in size). Most surface debris was piled at one end of the bulldozer cut, however a visual inspection of the site revealed that a minor part of the surface debris was buried by the disturbance.

2.2 SOIL SAMPLING

Pre-disturbance sampling was conducted in April and early October 2002 to characterize seasonal and topographic differences in soil C and N and soil N availability. In April, 16 sampling stations were established along transects that traversed both upland ($n = 8$) and riparian ($n = 8$) areas. The October sampling was performed at 17 stations (7 riparian and 10 upland). Post-disturbance sampling was conducted in June 2003 to characterize the effect of disturbance on soil quality. Results from the pre-disturbance sampling indicated topographic differences in soil properties (see results), therefore post-disturbance soil sampling was limited to upland locations. Seven sampling stations were randomly chosen along the disturbance zone created by the bulldozer. For each upland station in the disturbance zone, a control sampling station was selected in an undisturbed area ≈5 m from the disturbance.

In both pre- and post-disturbance soil sampling, replicate samples of the O-horizon (when present) were removed with a knife from a 214 cm^2 area above the mineral soil. Replicate mineral soil samples (0-30 cm) were then collected in butyrate plastic tubes using a soil probe (2.4 cm diameter) with hammer attachment (AMS, American Falls, ID). The distance between replicate samples was 1 to 2 m. A third sample (0-20 cm) was also collected at each sampling station by hammering a PVC pipe (5.1 cm diam x 25 cm long) into the mineral soil. The ends of the tubes and the pipes were capped to prevent soil loss during transport. Samples were transported to the laboratory and stored in a refrigerator (5 °C) prior to analysis.

2.3 SAMPLE ANALYSIS

2.3.1 Soil Carbon and Nitrogen

The dry mass of O-horizon material was determined after oven-drying at 75 °C. O-horizon samples were ground and homogenized in a sample mill and stored in airtight glass bottles prior to elemental analysis. Mineral soil samples collected in the butyrate tubes were cut into 10 cm increments and equivalent depth increments from the same sampling station were composited. Soil samples were dried to a constant weight at room temperature (21 °C) in a laboratory equipped with a dehumidifier. The air-dry soil samples were crushed with a rubber mallet and passed through a 2-mm sieve to remove gravel and coarse debris. A subsample of the soil (< 2 mm) was ground and homogenized in a ball mill and stored in an airtight container prior to elemental analysis.

2.3.2 Physical Fractionation of Soils

Part of each surface mineral soil sample (0-20 cm) collected in a PVC pipe was physically separated into particulate organic matter (POM) and mineral-associated organic matter (MOM) (Cambardella and Elliott, 1992). Twenty grams of air-dry soil were dispersed by shaking overnight in a 100 mL solution of sodium hexametaphosphate (5 g L^{-1}). The mixture was wet-sieved through a 0.053 mm sieve. POM was recovered by back washing the sieve, filtering (Whatman 541) the POM from the wash solution, and oven drying. The mixture that passed the 0.053 mm sieve was also oven dried to recover MOM (i.e., silt + clay). Both POM and MOM from each soil sample were weighed after oven drying (75 °C) and stored in airtight containers prior to elemental analysis.

2.3.3 Soil Nitrogen Availability

Part of each surface mineral soil sample (0-20 cm) collected in a PVC pipe was used for the determination of potential net soil N mineralization and nitrification in 12-week aerobic laboratory incubations. The fresh soil was passed through a 6.3 mm sieve to remove coarse debris and rocks. A subsample of the sieved soil was air-dried to determine the dry mass-to-fresh mass conversion factor. A second subsample of sieved soil (≈ 12 g) was extracted by shaking for 2 hours in a 100 mL solution of 2 M KCl to determine initial extractable NH_4-N and NO_3-N. The remaining soil (< 6.3 mm) was placed in a plastic jar and incubated in the dark at room temperature (21 °C). Once a week, the lids were removed briefly from the jars to aerate the soil samples. Extractions of the incubating soils were repeated after 6 and 12 weeks to determine the production of NH_4-N and NO_3-N. Extractions were allowed to settle overnight in a refrigerator (5 °C). Potential net soil N mineralization was calculated as the difference between extractable inorganic N (NH_4-N + NO_3-N) at 6 or 12 weeks and the initial extractable inorganic N. Potential net soil nitrification was calculated in a similar manner from concentrations of extractable NO_3-N. In each case, the units were µg N g^{-1} air-dry soil, based the dry mass-to-fresh mass conversion factor from the initial soil sample.

2.3.4 Elemental Analysis

Samples were analyzed for total C and N using a LECO CN-2000 (LECO Corporation, St. Joseph, MI). The elemental analyzer was calibrated using LECO standards traceable to the National Institute of Standards and Technology (NIST), Gaithersburg, MD. Soil extracts were analyzed for NH_4-N and NO_3-N concentrations by digital colorimetry using a Bran+Luebbe AutoAnalyzer 3.

2.3.5 Calculations

Carbon and N stocks (g element m^{-2}) in the O-horizon were calculated as the product of concentration (g element g^{-1}) and dry mass per unit area (g m^{-2}). Carbon and N stocks in each increment of mineral soil were calculated as the product of concentration (g element g^{-1} soil), soil density (g m^{-3}), and increment length (m). Soil density was calculated on the basis of air-dry mass (< 2 mm) and the known volume of soil collected in the butyrate plastic tubes.

Soil C in POM (g POM-C g^{-1} soil) or MOM (g MOM-C g^{-1} soil) was calculated by multiplying the dry mass of the POM or MOM part (g part g^{-1} soil) by the respective C concentration (g C g^{-1} part). The fraction of soil C in POM (fPOM) was calculated based on the total C measured in the POM and MOM. The C stock in surface mineral soil that was associated with POM (POM-C) was calculated as the product of soil C stock (g C m^{-2}) and fPOM. The C stock in surface mineral soil that was associated with MOM (MOM-C) was calculated as the product of soil C stock (g C m^{-2}) and (1 - fPOM). Following appropriate substitutions in the equations, similar calculations were performed for soil N.

The annual potential rate of net soil N mineralization was calculated by extrapolating the net N mineralization in 12-week aerobic laboratory incubations to 52 weeks (g N produced g^{-1} soil) and dividing by the surface (0-20 cm) soil N concentration (g N g^{-1}). This calculation provides an estimate of the fraction of organic soil N that is potentially mineralized each year.

2.3.6 Statistical Analysis

Although pre- and post-disturbance soil sampling occurred in the same general area, the two data sets were not directly comparable because sampling was not undertaken at precisely the same locations. Measurements on pre-disturbance soil samples were analyzed for seasonal (April vs. October) and topographic (riparian vs. upland) differences using two-way ANOVA. If one of the main effects and the interaction were not statistically significant, the analysis was simplified to a one-way ANOVA in which the statistically significant main effect was retained. Post-hoc tests of differences between means were performed using Fisher's protected least significant difference (LSD). The results of pre-disturbance soil sampling was used for planning post-disturbance sampling. Post-disturbance soil samples were limited to upland sites and were analyzed for the effects of bulldozer disturbance using a paired t-test (each disturbance sampling station was paired with an undisturbed control station). Unless stated otherwise, statistical significance was indicated by $P \leq 0.05$.

3. RESULTS

3.1 PRE-DISTURBANCE SOIL SAMPLING

3.1.1 O-Horizon

There were no significant differences between sampling dates (April and October) for O-horizon measurements in training compartment K-11. Mean ±SE O-horizon dry mass in the pre-disturbance soil sampling was 1246 ±78 g m^{-2} (n = 33). The mean ±SE O-horizon C and N stocks were 490 ±35 (n = 33) and 9.1 ±0.8 (n = 33) g m^{-2}, respectively. There were no significant topographic differences in O-horizon C and N stocks despite a significant difference between riparian and upland sampling stations in O-horizon N concentrations and C:N ratios (Table 1). The O-horizon N concentration was significantly less and the C:N ratio was significantly greater at upland sampling stations.

Table 1. Mean (±SE) pre-disturbance O-horizon N concentrations and C:N ratios at riparian and upland sampling stations in training compartment K-11. The number of sampling stations is shown in parenthesis.			
Variable	Sampling stations		F-value[a]
	Riparian (n = 15)	Upland (n = 18)	
N concentration (%)	0.814 ±0.044	0.656 ±0.051	5.2*
C:N ratio	49.4 ±1.8	70.1 ±8.0	5.4*

[a]Degrees of freedom = 1,31
*$P \leq 0.05$

3.1.2 Whole Mineral Soil

Sampling date had no significant effect on soil C or soil N concentrations in any of the mineral soil depth increments examined (0-10, 10-20, and 20-30 cm). However, there were significant differences in soil density, soil C and N concentrations, and soil C and N stocks between upland and riparian sites in training compartment K-11 (Table 2). Although riparian zones had significantly lower soil densities than upland areas, there was significantly more soil C and N in riparian zones because of large differences in soil C and N concentrations.

In general, soil C and N concentrations at riparian sampling stations were a factor of 2 or more greater than those at upland sampling stations. Mean ±SE C stocks over the top 30 cm of mineral soil were 5609 ±367 and 2748 ±131 g C m^{-2} in riparian and upland soils, respectively. Mean ±SE soil N stocks over the top 30 cm of mineral soil were 174 ±14 g N m^{-2} at the riparian stations and 85.4 ±7.2 g N m^{-2} at the upland stations. Topographic position had no significant effect on soil C:N ratios at any soil depth. The mean ±SE C:N ratios for the 0-10, 10-20, and 20-30 cm soil increments (n = 33 samples for each depth) were, respectively, 33.6 ±1.2, 37.4 ±2.3, and 38.7 ±5.0.

3.1.3 Physical Fractionation of Soils

None of the measurements associated with the physical separation of whole soil C or N between POM and MOM were significantly affected by sampling date. There were significant differences between riparian and upland sampling stations for measured amounts of POM and MOM, the fraction of soil C in POM (fPOM) and MOM (fMOM), and concentrations of C and N in MOM (Table 3). Soils from upland stations had greater amounts of POM and a greater fraction of soil C in POM. Riparian soils had greater C and N concentrations in MOM than soils from upland stations.

Soils from riparian sampling stations had significantly greater total soil C stocks and more C in POM and MOM than soils from upland sampling stations (Table 4). Carbon in the O-horizon and POM was summed to approximate labile soil C which was significantly greater in riparian soils than in upland soils (due to differences in amounts of POM-C). At riparian and upland sites, respectively, ≈62 and 49% of the total soil C was associated with MOM indicating greater relative amounts of stabilized soil C pool in areas adjacent to streams.

Table 2. Mean (±SE) pre-disturbance soil density, C and N concentrations, and C and N stocks at riparian and upland sites in training compartment K-11.
The number of sampling stations is shown in parenthesis.

Variable	Soil depth (cm)	Sampling stations		F-value[a]
		Riparian (n = 15)	Upland (n = 18)	
Soil density (g cm^{-3})	0-10	0.972 ±0.025	1.143 ±0.034	15.6***
	10-20	1.245 ±0.028	1.382 ±0.020	16.8***
	20-30	1.367 ±0.035	1.441 ±0.022	NS
Soil C concentration (%)	0-10	2.97 ±0.25	1.45 ±0.09	37.1***
	10-20	1.32 ±0.12	0.50 ±0.03	49.9***
	20-30	0.89 ±0.12	0.29 ±0.02	28.5***
Soil C stock (g C m^{-2})	0-10	2821 ±196	1638 ±96	32.5***
	10-20	1612 ±123	694 ±43	56.9***
	20-30	1176 ±129	415 ±31	39.1***
Soil N concentration (%)	0-10	0.092 ±0.008	0.044 ±0.015	32.2***
	10-20	0.039 ±0.004	0.015 ±0.001	34.2***
	20-30	0.030 ±0.006	0.009 ±0.001	13.9***
Soil N stock (g N m^{-2})	0-10	88.4 ±6.9	50.7 ±4.2	23.3***
	10-20	47.1 ±4.1	21.0 ±2.1	34.8***
	20-30	38.9 ±6.3	13.7 ±1.7	17.3***

[a]Degrees of freedom = 1,31
***$P \leq 0.001$; NS = not significantly different

3.1.4 Soil Nitrogen Availability

There was no effect of sampling date on initial extractable NH_4-N, NO_3-N and inorganic N in pre-disturbance soil samples from training compartment K-11. For this reason, extraction data from April and October were combined prior to a comparison of riparian and upland soils. Extractable NH_4-N and inorganic-N were significantly greater in soils from riparian sampling stations (Table 5). At both riparian and upland sites, extractable NO_3-N was a small percentage (≤ 3%) of initial extractable inorganic soil N.

There were complex interactions between time of sampling and sampling location for measurements of both potential net soil N mineralization and nitrification (Table 5). In April, riparian soils exhibited more potential net soil N mineralization than upland soils, but there was no significant difference in October soil samples. Potential net nitrification was significantly greater in riparian soils than upland soils in both April and October (although there was a 10% probability that the difference in April samples was due to chance alone). When data from the April sampling period were used, calculated potential rates of net N mineralization in riparian soils were significantly greater than those in upland soils. However, when data from the October sampling period were used, there was no statistically significant difference between riparian and upland sites.

Table 3. Mean (±SE) pre-disturbance particulate organic matter (POM), fraction of soil C in POM and mineral-associated organic matter (MOM), and C and N concentrations in MOM at riparian and upland sites in K-11.
The number of sampling stations is shown in parenthesis.

Variable	Sampling stations		F-value[a]
	Riparian (n = 15)	Upland (n = 18)	
POM (g POM g^{-1} soil)	0.772 ±0.024	0.841 ±0.008	8.3**
Fraction of soil C in POM	0.317 ±0.018	0.406 ±0.014	15.2**
Fraction of soil C in MOM	0.683 ±0.018	0.594 ±0.014	15.2**
C concentration in MOM (%)	6.44 ±0.51	4.08 ±0.28	17.9***
N concentration in MOM (%)	0.268 ±0.020	0.174 ±0.013	16.4***

[a]Degrees of freedom = 1,31
$^{**}P \leq 0.01$; $^{***}P \leq 0.001$

Table 4. Mean (±SE) pre-disturbance C stocks (g C m^{-2}) in different soil pools at riparian and upland sites in training compartment K-11.
The number of sampling stations is shown in parenthesis.

Soil C pool	Sampling stations		F-value[a]
	Riparian (n = 15)	Upland (n = 18)	
O-horizon	455 ±56	519 ±46	NS
Particulate organic matter (POM)	1404 ±117	950 ±62	12.9**
Labile organic matter	1859 ±146	1469 ±93	5.4*
Mineral-associated organic matter	3028 ±215	1382 ±76	59.9***
Total[b]	4888 ±311	2851 ±132	40.9***

[a]Degress of freedom = 1,31
[b]20 cm soil depth
$^{*}P \leq 0.05$; $^{**}P \leq 0.01$; $^{***}P \leq 0.001$; NS = not significantly different

Table 5. Mean (±SE) pre-disturbance concentrations of extractable NH_4-N, NO_3-N and inorganic N, potential net N mineralization and nitrification during a 12-week aerobic laboratory incubations, and the calculated annual rate of potential net N mineralization in surface (0-20 cm) mineral soils from riparian and upland sites at training compartment K-11.
The number of sampling stations is shown in parenthesis.

Variable	Sampling Date	Sampling stations: Riparian	Sampling stations: Upland	F-value
Extractable NH_4-N (μg g^{-1})	--	3.8 ±0.5 (15)	1.7 ±0.3 (18)	14.3***
Extractable NO_3-N (μg g^{-1})	--	0.09 ±0.030 (15)	0.05 ±0.02 (18)	NS
Extractable inorganic N (μg g^{-1})	--	3.9 ±0.5 (15)	1.8 ±0.3 (18)	14.9***
Net soil N mineralization (μg N g^{-1})	April	13.3 ±3.9 (8)	0.8 ±0.5 (8)	10.0**
	October	1.3 ±1.3 (7)	1.8 ±0.5 (10)	NS
Net nitrification (μg N g^{-1})	April	9.1 ±4.0 (8)	1.3 ±0.6 (8)	3.7†
	October	2.1 ±0.9 (7)	0.3 ±0.2 (10)	5.0*
Net N mineralization rate (yr^{-1})	April	0.080 ±0.015 (8)	0.014 ±0.008 (8)	14.9**
	October	0.008 ±0.008 (7)	0.027 ±0.009 (10)	NS

†$P \leq 0.10$; *$P \leq 0.05$; **$P \leq 0.01$; ***$P \leq 0.001$; NS = not significantly different

3.2. POST-DISTURBANCE SOIL SAMPLING

3.2.1 O-Horizon

Multiple properties of the forest floor were affected by the experimental disturbance (Table 6). Measurements of O-horizon dry mass, C concentrations and stocks, N stocks, and C:N ratios were significantly reduced at sampling points in the disturbance zone relative to paired controls. The disturbance reduced O-horizon dry mass by ≈60% and C stocks by ≈71%. O-horizon N concentration was the only forest floor measurement not significantly affected by the experimental disturbance.

Mean O-horizon dry mass measured at upland control points during post-disturbance sampling (640 g m^{-2}) was approximately half that measured in pre-disturbance sampling (1246 g m^{-2}). Similarly, mean post-disturbance stocks of C (310 g C m^{-2}) and N (3.19 g N m^{-2}) in the O-horizon at control sampling points were ≈37 and 65% less, respectively, than those measured during pre-disturbance soil sampling. The pre- and post-disturbance differences were not unexpected because a prescribed fire removed understory vegetation and O-horizons from the K-11 training compartment prior to the experimental disturbance. Lower O-horizon N concentrations (0.51 ±0.03%) in post-disturbance samples indicated that N losses from fire partly contributed to an observed post-disturbance increase in O-horizon C:N ratios.

Table 6. Mean (±SE) O-horizon properties at upland sites disturbed by a bulldozer and at paired, undisturbed (control) sites in K-11.
The number of sampling stations is shown in parenthesis.

Measurement	Treatment		Mean difference	Paired t-value
	Control (n = 14)	Disturbed (n = 14)		
Dry mass (g cm^{-2})	640 ±57	257 ±112	-384	3.26**
C concentration (%)	47.8 ±0.9	36.7 ±1.5	-12.9	9.19***
C:N ratio	100.5 ±7.5	73.5 ±8.7	-36.2	4.5**
C stock (g C m^{-2})	310 ±30	91 ±38	-219	4.9***
N stock (g N m^{-2})	3.19 ±0.34	1.39 ±0.60	-1.81	2.9*
*$P \leq 0.05$; **$P \leq 0.01$; ***$P \leq 0.001$				

3.2.2 Whole Mineral Soil

The effects of bulldozer disturbance on mineral soil properties are presented in Table 7. Disturbance significantly increased soil density in the 0-10 cm depth increment, but differences between control and disturbed sites were not statistically significant for the 10-20 and 20-30 cm soil increments. Throughout the soil profile, there was a decrease in soil C concentrations in the disturbance zone, and for each depth increment there was a 10% probability that the differences occurred by chance alone. Surface (0-10 cm) soil C stocks and N concentrations in the disturbance zone were also reduced by ≈37 and 40%, respectively, relative to control sampling points. Although there was a tendency for lower soil N stocks at disturbed sites relative to control sites, the differences were not statistically significant.

Post-disturbance soil density at control sites (1.13 g cm^{-3}) was similar pre-disturbance soil density at upland sites (1.14 g cm^{-3}) in the K-11 training compartment. Although not directly comparable, pre-disturbance soil C concentrations (1.45%) and stocks (1638 g C m^{-2}) were more similar to soil C concentrations (1.50%) and stocks (1931 g C m^{-2}) in the disturbance zone and less than soil C concentrations (2.71%) and stocks (3041 g C m^{-2}) measured at control sites during post-disturbance soil sampling (Table 7).

3.2.3 Physical Fractionation of Soil

Post-disturbance sampling indicated no differences between control and bulldozer disturbed sites in the amount of POM in surface mineral soils (Table 8). The amount of POM present was similar to that measured in pre-disturbance surface mineral soil samples from upland sampling sites (i.e., 0.841 g POM g^{-1} soil). Although concentrations of C and N in both POM and MOM tended to be less in soils from the area disturbed by the bulldozer, the differences were not significantly different. However, the fraction of soil C and N in POM (i.e., POM C and N expressed relative to total soil C and N) was significantly reduced in surface mineral soils under the disturbance (Table 8). Compared to controls, the fraction of soil C and N in the POM part was reduced by ≈20% and ≈32%, respectively, in soils from the disturbance zone.

Table 7. Mean (±SE) soil density, C and N concentrations, and C and N stocks at upland sites disturbed by a bulldozer and at paired, undisturbed (control) sites in training compartment K-11.
The number of sampling stations is shown in parenthesis.

Measurement	Soil depth (cm)	Treatment		Mean difference	Paired t-value
		Control (n = 7)	Disturbed (n = 7)		
Soil density (g cm^{-3})	0-10	1.13 ±0.03	1.32 ±0.05	0.18	2.74*
	10-20	1.41 ±0.06	1.48 ±0.05	0.08	NS
	20-30	1.54 ±0.05	1.57 ±0.02	0.03	NS
C concentration (%)	0-10	2.71 ±0.41	1.50 ±0.31	-1.20	2.35†
	10-20	0.66 ±0.11	0.45 ±0.37	-0.21	2.37†
	20-30	0.43 ±0.09	0.23 ±0.03	-0.20	2.10†
N concentration (%)	0-10	0.055 ±0.008	0.033 ±0.007	-0.022	1.99†
	10-20	0.010 ±0.003	0.006 ±0.003	-0.004	NS
	20-30	0.004 ±0.001	0.003 ±0.001	-0.001	NS
C stock (g C m^{-2})	0-10	3041 ±450	1931 ±356	-1110	2.05†
	10-20	909 ±114	654 ±184	-255	NS
	20-30	658 ±140	366 ±53	-292	1.98†
N stock (g N m^{-2})	0-10	62.0 ±9.4	41.5 ±7.9	-20.5	NS
	10-20	13.4 ±3.8	8.5 ±4.2	-4.9	NS
	20-30	5.2 ±2.1	4.1 ±1.5	-1.1	NS

$^{\dagger}P \leq 0.10$; $*P \leq 0.05$; NS = not significantly different

3.2.4 Soil Nitrogen Availability

The effects of bulldozer disturbance on measures of soil N availability are presented in Table 9. Although there was a tendency for greater amounts of extractable soil N and greater amounts of net soil N mineralization and nitrification in samples from the disturbance zone, the differences were not significantly different from undisturbed (control) samples. There was a high degree of variability that overshadowed differences in post-disturbance measures of surface (0-20

Table 8. Mean (±SE) amounts of particulate organic matter (POM), concentrations of C and N in POM and mineral-associated organic matter (MOM), and the fraction of soil C or N in POM at upland sites disturbed by a bulldozer and at paired, undisturbed (control) sites in K-11. The number of sampling stations is shown in parenthesis.				
Measurement	Treatment		Mean difference	Paired t-value
	Control (n = 7)	Disturbed (n = 7)		
Particulate organic matter (g POM g^{-1} soil)	0.847 ±0.005	0.831 ±0.016	-0.016	NS
POM C concentration (%)	0.79 ±0.05	0.61 ±0.16	-0.18	NS
MOM C concentration (%)	5.19 ±0.30	4.91 ±1.01	-0.28	NS
Fraction soil C in POM	0.457 ±0.016	0.368 ±0.032	-0.090	4.3**
POM N concentration (%)	0.014 ±0.002	0.009 ±0.003	-0.004	NS
MOM N concentration (%)	0.220 ±0.019	0.200 ±0.041	-0.020	NS
Fraction soil N in POM	0.248 ±0.027	0.168 ±0.033	-0.080	2.8*
*$P \leq 0.05$; **$P \leq 0.01$; NS = not significantly different				

Table 9. Mean (±SE) post-disturbance concentrations of extractable NH_4-N, NO_3-N, and inorganic N, potential net N mineralization and nitrification during a 12-week aerobic laboratory incubations, and the calculated annual rate of potential net N mineralization in surface (0-20 cm) mineral soils from control and disturbed sites in training compartment K-11. The number of sampling stations is shown in parenthesis.				
Measurement	Treatment		Mean difference	Paired t-value
	Control (n = 7)	Disturbed (n = 7)		
Extractable NH_4-N (μg g^{-1})	2.9 ± 0.9	3.0 ±1.3	0.1	NS
Extractable NO_3-N (μg g^{-1})	0.0 ±0.0	0.6 ±0.5	0.6	NS
Extractable inorganic N (μg g^{-1})	2.9 ±0.9	3.6 ±1.6	0.7	NS
Net soil N mineralization (μg N g^{-1})	6.0 ±2.4	9.6 ±5.8	3.6	NS
Net nitrification (μg N g^{-1})	8.1 ±3.3	12.1 ±7.0	4.0	NS
Net N mineralization rate (yr^{-1})	0.058 ±0.022	0.067 ±0.036	0.013	NS
NS = not significantly different				

cm) soil N availability. The calculated annual rates of net soil N mineralization, which were normalized for the amount of soil N present in each sample, were intermediate between rates measured for riparian (0.08 yr^{-1}) and upland (0.014 yr^{-1}) soils during pre-disturbance soil sampling in April 2002.

4. DISCUSSION

Pre-disturbance sampling revealed that site-specific, topographic differences in soil properties could potentially influence the interpretation of data from the disturbance experiment. In particular, there were major differences between riparian and upland sites that existed prior to the experimental disturbance. Soils from riparian areas in compartment K-11 had greater total C stocks and greater amounts of C in different soil parts (e.g., POM and MOM). In addition, a larger amount of C was associated with MOM in riparian soils than in upland soils. Topographic differences in N availability were also indicated by the incubation of pre-disturbance samples with a tendency toward higher N availability in riparian soils. The latter difference was, however, highly dependent on the time of soil sampling. Other than the effect on soil N availability, time of sampling made no difference to the interpretation of data from pre-disturbance soil samples. Greater amounts of C and N in riparian soils may be caused by depositional processes that move organic matter and nutrients from upland areas to riparian zones as well as higher levels of soil moisture in riparian zones that can inhibit decomposition of soil organic matter. Greater soil N availability in riparian zones at training compartment K-11 is consistent with results from other research that has examined topographic variation in forest soil N dynamics (Garten et al., 1994).

The forest at the experimental site was both thinned of trees and subjected to a prescribed burn prior to disturbance by the D7 bulldozer. In general, forest harvesting and prescribed fires have little or no effect on forest mineral soil C and N (Johnson and Curtis, 2001; Wan et al. 2001). Thinning and burning complicated planned pre- and post-disturbance comparisons of the bulldozer's effect on measures of soil quality, however their occurrence probably had no effect on paired comparisons between undisturbed and disturbed soils at upland sampling sites where soils were subjected to the same pre-disturbance forest management practices. Post-disturbance sampling was limited to upland sites in the K-11 training compartment because training with heavy, tracked vehicles at Fort Benning is generally confined to upland soils.

Soil sampling approximately one month after the experimental disturbance indicated that effects of the bulldozer were limited primarily to the forest floor (O-horizon) and the surface (0-10 cm) mineral soil. O-horizon dry mass and C stocks were significantly reduced, relative to undisturbed sites, and there was an indication of reduced mineral soil C stocks in the disturbance zone. Differences in the surface (0-10 cm) mineral soil also indicated a significant increase in soil compaction (i.e., soil density) as a result of disturbance with the bulldozer. However, the effects of soil compaction were not observed below the 0-10 cm depth increment. There was also a reduction in POM-C and N in the disturbance zone but no measurable effect on N availability due to a high degree of variation in measurements associated with the soil incubations for determination of potential net soil N mineralization.

Overall, effects of the bulldozer on measures of soil quality were consistent with reported differences among sites subject to minimal, light, moderate, and heavy training regimes at Fort Benning, Georgia. Greater soil density, less soil C, and less C and N in surface POM has been reported at sites where soils have been repeatedly impacted by tracked-vehicle training (Garten et al., 2003). In the present study, differences between undisturbed and disturbed forest soils were detectable at one month after only a few passes with the D7 bulldozer. The null hypothesis for the experiment was, therefore, partially rejected because there were declines ($P \leq 0.10$) in surface soil C and N as a consequence of heavy vehicle traffic. The removal of surface mineral soil by the bulldozer caused lower surface soil C and N concentrations in the disturbance zone. Although there was some tendency for greater soil N availability in disturbed soils, the changes were not significantly different from undisturbed controls. Thus, the null hypothesis with respect to soil N availability was not rejected. It is expected that repeated soil disturbance over time, which normally occurs in a military training area, would simply intensify the changes in soil properties that were measured following a one-time soil disturbance at the K-11 training compartment.

Finally, the experiment was also useful for identifying soil measurements that are particularly sensitive to disturbance and therefore can be used successfully as indicators of a change in soil properties as a result of heavy, tracked-vehicle traffic at Fort Benning. Measurements related to total O-horizon mass and C concentrations or stocks exhibited changes that ranged from ≈25 to 75% following the one-time disturbance. Changes in surface (0-10 cm) mineral soil density or measures of surface soil C and N following the disturbance were less remarkable and ranged from ≈15 to 45% (relative to undisturbed controls). Soil N availability (measured as initial extractable soil N or N production in laboratory incubations) was the least sensitive and the least useful indicator for detecting a change in soil quality as a result of heavy, tracked-vehicle disturbance. Collectively, the results suggest that the best indicators of a change in soil quality will be found at the soil surface because there were no statistically significant effects of bulldozer disturbance at soil depths below 10 cm.

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