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Changes in Soil Properties Following 55 Years of Secondary Forest Succession at Fort Benning, Georgia, U.S.A.

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Abstract

We present results on changes in soil properties following land use change over an approximately 55-year period at Fort Benning, Georgia, U.S.A. Soil cores were taken at 129 locations that were categorized as reforested (field/bare ground in 1944 and forest in 1999), disturbed (field/bare ground in 1944 and 1999), or reference forests (forest in 1944 and 1999). Soil disturbance included historic agriculture (pre-1944) and military training (post-1944). Density in mineral soils exhibited a historic land use legacy effect (reference < reforested < disturbed). Rates of change in bulk density decreased with depth and estimated total times to reach reference forest levels ranged from 83 (0–10 cm) to 165 (30–40 cm) years. A land use legacy effect on C stock was apparent in the O-horizon and in 30- to 40-cm soil increment (reference > reforested > disturbed). Soil C stock in all other increments and in particulate organic matter was affected by disturbance; however, no legacy was apparent (reference = reforested > disturbed). For the entire soil profile (O-horizon to 40 cm), rate of C accrual was 28 g m^{-2} yr^{-1} (1.5%/yr). Nitrogen stocks were affected by disturbance in the O-horizon and 0- to 10-cm increment; however, no legacy effect was detected (reference = reforested > disturbed). Nitrogen accumulated at 0.56 g m^{-2} yr^{-1} (0.6%/yr) for the entire soil profile. At Fort Benning, soil C and N stocks of reforested stands were similar to those of reference forested stands after approximately 55 years. However, soil bulk density was greater on reforested stands than reference forest stands at 55 years and may require an additional century to reach reference levels.

Key words: **carbon, density, historic land use, land use legacy, nitrogen, restoration, soil.**

Introduction

Recovery of ecosystem structure, function, and composition from historic anthropogenic disturbance has received much attention, likely because of the increased recognition of the long-term imprint of historic disturbance on contemporary conditions (Foster et al. 2003; Lunt & Spooner 2005). There is ample evidence illustrating historic land use influences on contemporary forest ecosystems, especially with respect to vegetation patterns. In general, forest canopies in historically disturbed plots are more homogenous than in historically undisturbed sites, showing a higher proportion of shade-intolerant, fast-growing species (Zimmerman et al. 1995; Foster et al. 2003; Grau et al. 2003; Ziegler 2004). Moreover, seed banks within historically disturbed forests may have a greater abundance of early successional seeds than those in nondisturbed forests (Bossuyt & Hermy 2001). Soil microbial communities also may be susceptible to historic land use, exhibiting altered community structure for decades after abandonment (Steenwerth et al. 2003). Because soil properties are highly dependent on resident vegetation and microbial communities, the imprint of historic disturbance also is manifested in forest soils.

Soils on recovering lands often have altered physical, organic, and nutrient properties. For example, soils on historically cultivated sites have greater soil bulk densities than those on respective historically noncultivated sites (Compton et al. 1998). Historically cultivated sites also have lower C and N stocks than comparable noncultivated sites (Knops & Tilman 2000; Foster et al. 2003; Falkengren-Grerup et al. 2006) and old forests (Latty et al. 2004). Soil C:N ratios are also less on historically cultivated sites than under pastures (Compton et al. 1998; Compton & Boone 2000), whereas soil phosphorus levels are greater on historically cultivated sites than on comparable noncultivated sites (Koerner et al. 1997; Verheyen et al. 1999; Falkengren-Grerup et al. 2006). Burke et al. (1995) also reported a land use legacy effect on soil texture with lower silt content on historically cultivated fields (abandoned for 50 years) than on native fields. Because of these often reported long-term legacies of historic agriculture on soils, it is important to document

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and understand changes in soil properties following agricultural abandonment. Unfortunately, considerable variation exists for reported recovery times of soils from historic disturbance, with estimates ranging from decades to millennia (Burke et al. 1995; Verheyen et al. 1999; Knops & Tilman 2000; Dupouey et al. 2002), likely a result of differing climatic, edaphic, and vegetation properties.

In order to adequately address soil changes following abandoned historic land use (i.e., agriculture), study areas need to have (1) well-documented historic and contemporary land use and (2) sufficient variation in restoration of abandoned lands. Military installations provide excellent areas to test the effects of historic land use on contemporary soil properties because they satisfy each of these requirements. First, they usually have well-documented historic and contemporary land use patterns. Second, they often have varied contemporary land use patterns, which results in large areas with contrasting disturbance intensities allowing many sites to recover, whereas other sites are maintained under disturbed conditions by military training. The objective of our study was to determine whether the effects of historic land use (abandoned approximately 55 years ago) still persist in contemporary soils at the Fort Benning Military Installation (FBMI), Georgia, U.S.A. Specifically, we tested whether soils under different land use histories (i.e., disturbed, reforested, and reference forests) had different soil bulk densities and C and N stocks.

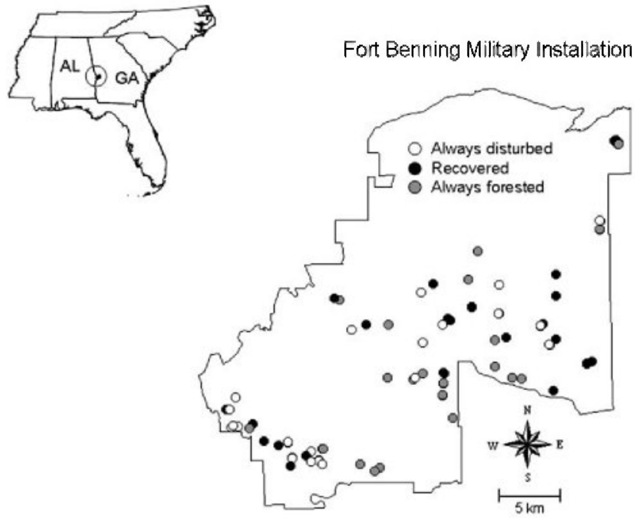

Figure 1. Map of FBMI with soil sampling sites.

Methods

Study Site

The FBMI comprises an area of approximately 735 km^2 in the Chattahoochee and Muscogee counties, Georgia, U.S.A. (Fig. 1). The climate in this region is humid and mild, with precipitation occurring throughout the year ($\bar{X}$ = 105 cm/yr). Vegetation on FBMI is primarily oak–hickory–pine and southern mixed forests. Geology is mainly Cusseta Sand (Lawton 1976). Soils at Fort Benning are highly weathered Ultisols, mainly of Coastal Plain origin but with minor inclusion of alluviums derived from the Piedmont ecoregions to the north (Jones & Davo 1997). Fort Benning lies within the Southeastern Plains ecoregion and the Sand Hills and Southern Hilly Gulf Coastal Plain subecoregions (Griffith et al. 2001). Dominant soil series found within these subecoregions 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 in the upland sites, whereas sandy loams and sandy clay loams are common in valley and riparian areas.

Fort Benning was established in 1918 by the U.S. military with an additional major land acquisition in 1941. Land use prior to military purchase (early 1800s to 1918, 1941) was primarily agriculture (cotton farming) and silviculture (Hilliard 1984; Frost 1993; Kane & Keeton 1998). Since military purchase, the land has been used for training of military personnel, including both infantry and mechanized training. Mechanized training includes heavy equipment vehicles (tanks, armored personnel carriers, light- and heavy-wheeled vehicles), which disturbs the landscape by disrupting both soil and vegetative cover (USAIC 2001; Dale et al. 2002). However, FBMI has large areas that are solely used for infantry training. These latter areas are undisturbed by the heavy equipment used in mechanized training and thus allowed to revegetate and recover from historic disturbance. Moreover, FBMI has established military compartments, which spatially segregate military training activities. This spatial segregation afforded sites that varied in the level of contemporary disturbance from military training as well as sites that were under different levels of soil/vegetation recovery from historic agricultural and silvicultural disturbance. Fort Benning uses prescribed burning to remove understory vegetation on almost approximately 3-year fire return interval. There are minor predicted effects of prescribed burning on forest recovery or sustainability (Garten 2006) as well as measured minor effects on soil A-horizon depth and the composition of understory vegetation (Dilustro et al. 2006). However, an analysis of a subset of sites indicated no difference among land use categories with respect to years since the last burn (p = 0.67, one-way analysis of variance; reference forests = 2.3 years, n = 12; reforested = 1.9 years, n = 13, and disturbed = 1.8 years, n = 13). Further details on the biology, geology, and physical setting of Fort Benning are available in Jones and Davo (1997). An online account of the history of Fort Benning and the region is available at http://www.cr.nps.gov/seac/benning-book/benning-index.htm.

Soil Sampling and Sample Preparation

Replicate mineral soil samples were collected from 1999 to 2002 at 129 sites using a stainless steel soil recovery probe (2.54 cm inner diameter) with a hammer attachment (AMS, American Falls, ID, U.S.A.) to a 40 cm depth and then separated into 10-cm depth increments. Sites were located near Fort Benning's Land Condition–Trend Analysis sampling plots (Diersing et al. 1992). Where present, the O-horizon was sampled by removing a 0.0214-m^2 area directly above the soil sampling locations. Dry mass of the O-horizon was quantified by oven drying (65°C), whereas other soil samples were air dried to a constant weight (22°C). Air-dried samples were then crushed and passed through a 2-mm sieve to remove gravel and coarse organic debris. We then dispersed a 20 g subsample of the sieved soil by shaking overnight in 100 mL of sodium hexametaphosphate (5 g/L), which was then wet sieved through a 0.053-mm sieve for sand ($\geq$0.053 mm) and silt + clay (<0.053 mm) estimation. Soil density (Mg/m^3) was estimated from the air-dried soil mass for each soil segment and the calculated volume of each increment.

Both O-horizon and mineral soil samples were ground and homogenized and then analyzed for C and N concentrations (Perkin-Elmer 2400 Series II CHNS/O Analyzer; Perkin Elmer Analytical Instruments, Norwalk, CT, U.S.A. or a LECO CN-2000; LECO Corporation, St. Joseph, MI, U.S.A.). Carbon and N stocks for each soil layer were quantified as dry mass per unit area (g/m^2). Because of differences among land use categories in soil density (see Results), we have chosen to report C and N data as stocks (g/m^2), which account for differences in both concentrations and soil density. Moreover, we separated particulate organic matter C (POM-C) by wet sieving methods (Cambardella & Elliott 1992) from organic matter in the 0- to 20-cm soil increment. Particulate organic matter consists of free organic debris in the mineral soil and larger fragments of organic matter ($\geq$0.053 mm) released following dispersion of soil aggregates by shaking with sodium hexametaphosphate (5 g/L). Further details on the methods of soil analysis can be found in prior reports by Garten et al. (2003) and Garten and Ashwood (2004).

Land Use Classification

Land use at sampling sites was visually categorized by superimposing soil sampling locations on 1944 and 1999 aerial photography using ArcView 3.2 software (Environmental Systems Research Institute, Redlands, CA, U.S.A.). For each time period, we classified land use as bare ground/fields or forested. Therefore, we had four categories. Category 1 sites were bare ground or old fields in 1944 and forests in 1999, indicating secondary forest succession from historic disturbance (i.e., reforested sites). Category 2 sites were bare ground or old fields in both 1944 and 1999, suggesting disturbance in both time periods (i.e., disturbed sites). Category 3 sites were forested in both 1944 and 1999, in effect showing no or minimal anthropogenic disturbance during both time periods (i.e., reference forest sites). Category 4 sites were forests in 1944 and bare ground or old fields in 1999, indicating sites that became more disturbed over time (i.e., more disturbed sites); however, we excluded this latter category from analyses due to low frequency of more disturbed sites ($n = 7$). We assumed that sites that were forest in 1944 and 1999 were "reference" conditions to which recovery should attain and identified a land use legacy effect when a soil property differed between these reference sites and reforested sites.

Statistical Analysis

We tested for differences in means among land use categories with a one-way ANOVA. Following a significant ANOVA ($p < 0.05$), we used Tukey's studentized range multiple comparison tests to evaluate differences among land use categories ($p < 0.05$). In addition, for soil properties that exhibited different values between reforested and disturbed sites, we calculated accumulation rates for soil C and N by dividing the difference between mean values at reforested and disturbed sites by 55 (years since abandonment). The calculation assumes that levels of soil C and N at disturbed sites are baseline values against which reforested sites can be compared. Additionally, for soil properties found to differ between reference forests and reforested sites (i.e., exhibited a legacy effect), we estimated the additional time to reach reference forest levels by dividing the difference in mean reference forest and reforested values by an accumulation rate for each soil property. Finally, we calculated total time to reach reference forest levels for these soil properties by adding 55 to the additional time. These latter estimates assume that accumulation of soil C and N during secondary succession is a linear process. Other studies (Switzer et al. 1979; Garten et al. 2003) indicate that accumulation of surface soil organic matter on the southeastern coastal plain, and therefore C and N stocks, is a curvilinear (i.e., logistic) process; therefore, our linear approximation may underestimate accrual rates in the early years of succession and overestimate these rates in later years of succession. Land use data from additional time periods were unavailable and prevented us from parameterizing a curvilinear model of changes in soil properties. Other researchers have had similar problems when calculating rates of soil C and N accrual (Post & Kwon 2000).

Results

Soil texture was similar among land use categories. Mean (± 1 SE) percent sand did not differ among land use categories ($p = 0.98$, one-way ANOVA) and was 69.1 ± 2.6% for reference forest sites, 69.6 ± 3.4% for reforested sites, and 68.6 ± 4.6% for disturbed sites. Similarly, mean (± 1 SE) percent silt and clay did not differ among land uses

($p = 0.96$, one-way ANOVA) and was 31.1 ± 2.6% for reference forest sites, 31.1 ± 3.3% for reforested sites, and 32.4 ± 4.6% for disturbed sites.

Soil density differed among land use categories and exhibited an historic land use effect at all depth increments (all $p < 0.05$; Table 1). In the O-horizon, mean density was highest in the reference forests and reforested sites and lowest in the disturbed sites. However, density in the upper mineral soil (0–10 and 10–20 cm) exhibited an opposite pattern, with reference sites being least dense, followed by reforested sites, and then disturbed sites with the highest density. At the 20- to 30-cm and 30- to 40-cm increments, mean density was less in reference sites than in reforested and disturbed sites (Table 1).

Similar to density, land use disturbance affected C stocks at all soil depths; however, unlike density, the effects of historic land use were apparent only in a few soil increments (Table 1). In the O-horizon, C stock exhibited an historic land use effect, being highest in reference forest sites, intermediate in reforested sites, and least in disturbed sites. However, differences became less clear with increasing depth. No historic land use effect was apparent in the 0- to 10-cm C stocks where reference and reforested sites were not different, but both were higher than disturbed sites (Table 1). No influence of historic land use was apparent on C in the 10- to 20-cm or 20- to 30-cm increment. At 10–20 cm, C stock was highest in reference forest sites and least in disturbed sites, whereas at 20–30 cm, C stock was similar between the reference and the reforested sites, both of which had higher C stocks than disturbed sites. An historic land use effect occurred in the 30- to 40-cm increment as reference sites had higher C than either reforested or disturbed sites (Table 1). No legacy effect occurred at larger soil increments (0–20 cm and O-horizon to 40 cm) where reference forest and reforested sites had higher C stocks than disturbed sites. POM-C was not influenced by historic land use but was higher in reference and reforested sites than in disturbed sites (Table 1).

Although N stock was affected by disturbance, this effect was limited to the upper soil layer and exhibited no effect of historic land use (Table 1). In the O-horizon, N was similar in reference forests and reforested sites, and both had higher N than disturbed sites. In the 0- to 10-cm increment, N was higher in reference sites than in disturbed sites. No differences in N were observed at 10- to 20-cm, 20- to 30-cm, and 30- to 40-cm soil increments (Table 1). At the larger increments (0–20 cm and O-horizon to 40 cm) reference sites had higher N than disturbed sites (Table 1).

The estimated rate of change in soil density decreased with depth, being highest in the 0- to 10-cm layer and lowest in the 30- to 40-cm layer (Table 2). The accumulation rate of C was highest in the 0- to 10-cm increment (13.9 g C m^{-2} yr^{-1}) and lowest in the 30- to 40-cm increment (0.9 g C m^{-2} yr^{-1}). The accumulation rates of C for the 0- to 20-cm and O-horizon to 40-cm cumulative increments

Table 1. Mean (± 1 SD) for each soil property by land use category.

Soil Property	*Land Use Category* Disturbed	Reforested	Reference Forests	*p Value*
Density (Mg/m^3)				
O-horizon*	0.04 (0.09)^a	0.09 (0.06)^b	0.12 (0.05)^b	<0.0001
00–10 cm	1.48 (0.19)^a	1.23 (0.21)^b	1.09 (0.13)^c	<0.0001
10–20 cm	1.61 (0.15)^a	1.45 (0.20)^b	1.32 (0.13)^c	<0.0001
20–30 cm	1.61 (0.17)^a	1.50 (0.20)^a	1.39 (0.14)^b	<0.0001
30–40 cm	1.60 (0.18)^a	1.55 (0.18)^a	1.44 (0.13)^b	0.0004
Carbon (g/m^2)				
O-horizon	92.9 (175.9)^a	353.6 (176.1)^b	480.5 (175.6)^c	<0.0001
00–10 cm	960.3 (778.0)^a	1,726.0 (733.8)^b	1,953.2 (739.8)^b	<0.0001
10–20 cm	556.5 (421.3)^a	784.3 (356.6)^{ab}	983.1 (501.9)^b	0.0003
20–30 cm	359.1 (254.9)^a	571.5 (309.8)^b	677.4 (446.1)^b	0.0013
30–40 cm	313.9 (241.5)^a	364.1 (198.7)^a	622.6 (443.3)^b	<0.0001
00–20 cm	1,516.7 (1,140.5)^a	2,510.4 (1,031.6)^b	2,936.3 (1,144.6)^b	<0.0001
O-horizon to 40 cm	1,904.1 (1,510.2)^a	3,435.5 (1,392.0)^b	4,084.0 (1,489.2)^b	<0.0001
00–20 cm POM-C	424.7 (371.1)^a	847.6 (435.1)^b	960.8 (387.7)^b	<0.0001
Nitrogen (g/m^2)				
O-horizon	2.4 (5.3)^a	6.6 (5.2)^b	8.9 (4.5)^b	<0.0001
00–10 cm	45.8 (44.6)^a	64.4 (30.8)^{ab}	72.7 (35.2)^b	0.0082
10–20 cm	28.3 (24.0)	29.6 (13.7)	38.4 (23.0)	0.0530
20–30 cm	21.8 (20.7)	25.9 (14.6)	28.2 (22.1)	0.3762
30–40 cm	19.3 (16.45)	21.2 (14.5)	24.0 (16.7)	0.4486
00–20 cm	74.1 (67.6)^a	94.0 (42.7)^{ab}	111.1 (52.5)^b	0.0150
O-horizon to 40 cm	95.8 (93.3)^a	126.5 (55.5)^{ab}	147.9 (68.0)^b	0.0109

Values across rows with different superscript letters indicate that means were significantly different (Tukey's studentized range test, $p < 0.05$).
*O-horizon units were Mg/m^2.

Table 2. Estimated accumulation rates for soil density (Mg m^{-3} yr^{-1}) and C and N stocks (g m^{-2} yr^{-1}).

Soil Property	*Density ($Mg\ m^{-3}\ yr^{-1}$)*	*Carbon ($g\ m^{-2}\ yr^{-1}$)*	*Nitrogen ($g\ m^{-2}\ yr^{-1}$)*
O-horizon	0.001*	4.7	0.08
00–10 cm	−0.005	13.9	0.34
10–20 cm	−0.003	4.1	—
20–30 cm	−0.002	3.9	—
30–40 cm	−0.001	0.9	—
00–20 cm	—	18.1	0.36
O-horizon to 40 cm	—	27.8	0.56

An "em dash" (—) indicates that no differences were found among land use categories (N in 10- to 20-cm, 20- to 30-cm, and 30- to 40-cm increments) or no data were available (soil density at 0–20 cm and O-horizon to 40 cm).
*Density O-horizon units were g cm^{-2} yr^{-1}.

were 18.1 and 27.8 g C m^{-2} yr^{-1}, respectively (Table 2). POM-C stocks in the 0- to 20-cm increment increased at a rate of 7.7 g C m^{-2} yr^{-1}. Significant differences among land use categories in N were only observed in the O-horizon, 0–10 cm, 0–20 cm, and the entire soil profile to a depth of 40 cm. Accumulation rates of N in the O-horizon and 0- to 10-cm layer were 0.08 and 0.34 g N m^{-2} yr^{-1}, respectively (Table 2). Estimated accumulation rates of N for the 0- to 20-cm and O-horizon to 40-cm cumulative soil increments were 0.36 and 0.56 g N m^{-2} yr^{-1}, respectively.

Regardless of depth, soil density had not fully reached reference forest levels (reference sites $\neq$ reforested sites) following 55 years of secondary forest succession. Estimated additional years for soil density to reach reference forest levels ranged from 28 (0- to 10-cm layer) to 110 (30- to 40-cm layer) years, resulting in an estimated range of total times of 83–165 years (Table 3). At the time of our study, C stocks in the O-horizon and 30- to 40-cm increment also had not fully reached reference forest levels, and this attainment was estimated to take an additional 27 years (total time of approximately 82 years) and 287 years (total time of approximately 342 years), respectively (Table 3).

Table 3. Estimated additional and total years for soil properties that were influenced by historic land use (reference forests $\neq$ reforested sites) to attain levels comparable to reference forests.

	Time (years) to Attain Reference Forest Levels	
Soil Property	*Additional*	*Total*
Density		
O-horizon	30	85
00–10 cm	28	83
10–20 cm	43	98
20–30 cm	55	110
30–40 cm	110	165
Carbon		
O-horizon	27	82
30–40 cm	287	342

Discussion

Our study suggests that, in general, soil C and N stocks have reached levels measured under minimally disturbed forest stands during 55 years of secondary forest succession at Fort Benning, Georgia. Only C in the O-horizon and 30- to 40-cm soil increment exhibited a land use legacy effect that was manifested as lower C stocks at reforested sites compared with reference forest sites. Nonetheless, regardless of soil depth (up to 40 cm), there was a strong land use legacy effect for soil density at Fort Benning. At all mineral soil increments, density was higher on reforested than on forest reference sites. Our results suggest that soil density in potentially reforested sites will take an additional 28–110 years to reach reference forest levels, resulting in total restoration times of 83–165 years depending on soil depth. However, we must note that our estimated restoration times were based on the assumption that land use was constant over time. The nature of past disturbances, especially on the disturbed sites, has changed over time (from heavy agriculture to mainly military land use), which may affect estimated accumulation rates and thus restoration times. Moreover, an important caveat is the possibility that our contemporary disturbed sites are still being impacted by military maneuvers and therefore may be in a continual state of soil decline. Although unlikely, if true, this would result in an overestimation of accumulation rates and therefore an underestimation of restoration times.

The strong imprint of historic land use on soil density at Fort Benning agrees with findings of Compton et al. (1998) for abandoned, sandy soils in Massachusetts. They reported greater bulk density for soils historically plowed than those unplowed 40–60 years after abandonment of agriculture. However, our results differ from other reports of no historic land use effect on soil bulk density in Kentucky, 60 years post-abandonment (Kalisz 1986) and, in Sweden, 40–80 years post-abandonment (Falkengren-Grerup et al. 2006), which suggests a strong effect of other local factors (like vegetation and soil type) on soil density. Surprisingly, density in the 30- to 40-cm soil increment was significantly less in reforested or reference sites than in disturbed sites, which suggests a long-term effect of forest cover on deeper soil layers at Fort Benning. This latter difference is counter to assumed cultivation effects being limited to the upper 30 cm soil (Falkengren-Grerup et al. 2006) and may be indicative of the high intensity of historic agricultural and/or military training activities. This effect of historic land use on soil density likely results in degraded soil quality (Unger & Kaspar 1994) as soil compaction leads to a reduction in infiltration capacity (Iverson et al. 1981), availability of organic matter to microorganisms, decomposition of soil organic matter, and availability of soil N (Breland & Hansen 1996). The potential long-term impacts of land use practices on soil density are important to land managers who must account for possible legacy effects in restoration and conservation practices to fully realize soil and forest recovery at historically disturbed sites.

The rate of change for O-horizon density (dry mass) was 0.001 g cm^{-2} yr^{-1}. Based on this estimated rate, we predict that the O-horizon dry mass will reach reference forest levels in approximately 88 years. Switzer et al. (1979) reported that recovery of the forest floor reached maximum amounts of organic matter during the middle stage of secondary forest succession or approximately 45–100 years following abandonment of old fields on the Gulf Coastal Plain. Thus, our linearly estimated time for restoration of the forest O-horizon agrees reasonably well with prior studies. Further, our reported rates of change in mineral soil density (−0.005 to −0.001 Mg m^{-3} yr^{-1}) produce a range in estimated total restoration times from 83 (0–10 cm) to 165 (30–40 cm) years, which agree with other reports of changes in surface mineral soil density on the Gulf Coastal Plain (Switzer et al. 1979). Our results also indicate that restoration times are a function of soil depth and suggest that following agricultural abandonment, changes in density of deeper mineral soils of the Gulf Coastal Plain may take longer than 100 years. Despite the agreement of our estimated rates of soil C and N accrual with other published data, we again caution the reader that they were estimated as linear rates between two points in time and thus are considered approximations to more complex curvilinear rates of recovery in measures of soil quality (Garten et al. 2003).

Although the results suggest a strong effect of disturbance on soil C stocks at Fort Benning, no consistent effect of historic land use was evident, which is contrary to many reports of lower C on historically cultivated sites (Kalisz 1986; Falkengren-Grerup et al. 2006) and logged sites (Latty et al. 2004). However, our results agree with those reported by Compton et al. (1998), where there was no difference in soil C stocks on historically plowed and unplowed sites in more than 50 years following abandonment. Similar recovery times (40–50 years) also are reported for tropical soils in Puerto Rico (Brown & Lugo 1990) and Australian soils under secondary succession (>30 years; Paul et al. 2002). Switzer et al. (1979) also reported restoration of C in surface soils of the Gulf Coastal Plain during the middle stage of succession (45–100 years), and Wigginton et al. (2000) reported that recovery of soil C to 75% of comparable reference conditions may take more than 50 years for soils in a forested floodplain at the Savannah River site, National Environmental Research Park, South Carolina, U.S.A. A potential mechanism behind such contrasting reports on C recovery may be soil texture because there is a commonly inferred relationship between clay content and soil C stocks (Jobbagy & Jackson 2000). However, recent studies have suggested that C accrual is affected by many factors in addition to soil texture, including site preparation, previous land use, climate, and site management (Paul et al. 2002, McLauchlan 2006), making inferences on the exact mechanisms behind such contrasting reports difficult. Nevertheless, our results, taken together with the findings of others (Switzer et al. 1979; Wigginton et al. 2000), suggest that restoration of C stocks in Gulf Coastal Plain soils following agricultural abandonment may take less than 100 years.

Additionally, although we found an effect of contemporary land use on POM-C, no effect of historic land use was detected, suggesting that soil quality, as measured by POM-C, has recovered in approximately 55 years following abandonment of disturbance. The effect of contemporary disturbance was expected as POM-C is an indicator of soil quality (Gregorich et al. 1994) and decreases with soil disturbance (Garten et al. 2003). That we found no effect of historic disturbance is also not surprising because POM-C would be expected to exhibit a relatively short turnover time (measured in years; Bird et al. 1996).

At Fort Benning, C accumulation rate within the O-horizon to 40-cm depth increment was estimated at 27.8 g C m^{-2} yr^{-1}, which is only slightly below previously reported rates for soil C accrual of 30 g C m^{-2} yr^{-1} (Schlesinger 1990) and 34 g C m^{-2} yr^{-1} (Post & Kwon 2000). Our results also suggest a decrease in accumulation rate of C with depth in the mineral soil, resulting in longer recovery times for deeper soil increments. A possible mechanism limiting the accumulation rate of both soil C stocks and POM-C stocks (Garten et al. 2003) at Fort Benning may be the rapid decomposition of fresh organic matter inputs to mineral soils in sandy, coastal plain settings (Richter et al. 1999). Such inputs probably decrease with increasing depth because root biomass declines strongly with soil depth (Jackson et al. 1996), and this may account for the decreased accumulation rate of C in deeper soil increments.

The effects of disturbance on N stocks were limited to the upper layers of the soil profile. However, similar to C stock, no effect of historic land use was found on N stock, which is counter to other studies indicating a strong legacy effect of historic land use on soil N in surface soils (Burke et al. 1995; Latty et al. 2004; Falkengren-Grerup et al. 2006). Our findings, however, agree with recovery times of N in mineral soils in Mississippi (45–100 years; Switzer et al. 1979) and Puerto Rico and Virgin Islands (15–20 years, Brown & Lugo 1990). Nitrogen accumulation rates for our study ranged from 0.08 g N m^{-2} yr^{-1} in the O-horizon to 0.34 g N m^{-2} yr^{-1} in the 0- to 10-cm increment, which are considerably less than the 1.23 g N m^{-2} yr^{-1} accumulation rate reported by Knops & Tilman (2000) for a 0- to 10-cm increment in a Minnesota sand plain. Moreover, the accumulation rate for the entire soil profile was calculated at 0.56 g N m^{-2} yr^{-1}, which also falls below estimates given by Knops and Tilman. Our results also suggest that N accumulated at a slower rate than C, a pattern counter to other studies (Burke et al. 1995; Knops & Tilman 2000). For example, C stocks at the reforested sites increased 80% from 1944 to 1999 (1.5%/yr), whereas N stocks increased 32% (0.6%/yr) over the same time frame. The slow accumulation of N may be due, in part, to the relatively minor input of N from N-fixing legumes on

Fort Benning soils (LaJeunesse et al. 2006) and/or a tightly closed N cycle in forests on the sandy, nutrient-poor soils. Finally, no significant differences in N stock in deeper soil layers (10–20, 20–30, and 30–40 cm) among reference forests, reforested, and disturbed sites indicate no strong effect of land use (either historic or contemporary) on soil N at these depths.

Over 55 years of secondary forest succession, C and N stocks have largely attained levels measured for reference forest (>55 years) soils at Fort Benning. However, soil density exhibited a land use legacy effect throughout the entire soil profile (up to 40 cm depth), with estimated times to reach reference forest levels ranging from approximately 83 to 165 years. The long-term effects of land use practices on soil density are important to effective land management and restoration practices at Fort Benning and other locations on the southeastern coastal plain. Although soil C and N stocks may indicate recovery from historic disturbance, contemporary soil quality can be significantly impacted through legacy effects on soil density. Therefore, to fully reach restoration goals, land managers need to account for not only soil C and N stocks, which are primary determinants of soil quality, but also potential impacts of ecosystem disturbance on soil density.

Implications for Practice

- Soil C and N stocks on revegetated stands reached levels measured in reference forest stands following 55 years of abandonment, suggesting that forest succession may be a satisfactory restoration practice for these two soil properties.
- Soil bulk density on revegetated stands had not reached levels on reference forested sites after 55 years, and results indicate that an additional century may be necessary to reach reference site levels. This suggests that additional restoration practices, in addition to natural succession, are required to restore formerly plowed soils in a timely manner.
- Land managers and restoration ecologists should include an assessment of historic land use practices in contemporary restoration projects. Failure to account for these practices can potentially lead to unidentified legacies, which may result in unrealistic restoration goals.

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LITERATURE CITED

Bird, M. I., A. R. Chivas, and J. Head. 1996. A latitudinal gradient in carbon turnover times in forest soils. Nature **381:**143–146.

Bossuyt, B., and M. Hermy. 2001. Influence of land use history on seed banks in European temperate forest ecosystems: a review. Ecography **24:**225–238.

Breland, T. A., and S. Hansen. 1996. Nitrogen mineralization and microbial biomass as affected by soil compaction. Soil Biology and Biochemistry **28:**655–663.

Brown, S., and A. E. Lugo. 1990. Effects of forest clearing and succession on the carbon and nitrogen-content of soils in Puerto-Rico and US Virgin Islands. Plant and Soil **124:**53–64.

Burke, I. C., W. K. Lauenroth, and D. P. Coffin. 1995. Soil organic matter recovery in semiarid grasslands: implications for the conservation reserve program. Ecological Applications **5:**793–801.

Cambardella, C. A., and E. T. Elliott. 1992. Particulate soil organic-matter changes across a grassland cultivation sequence. Soil Science Society of America Journal **56:**777–783.

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

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

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

Diersing, V. E., R. B. Shaw, and D. J. Tazik. 1992. US Army Land Condition-Trend analysis (LCTA) program. Environmental Management **16:**405–414.

Dilustro, J., B. Collins, and L. Duncan. 2006. Land use history effects in mixed pine-hardwood forests at Fort Benning. Journal of the Torrey Botanical Society **133:**460–467.

Dupouey, J. L., E. Dambrine, J. D. Laffite, and C. Moares. 2002. Irreversible impact of past land use on forest soils and biodiversity. Ecology **83:**2978–2984.

Falkengren-Grerup, U., D. J. ten Brink, and J. Brunet. 2006. Land use effects on soil N, P, C and pH persist over 40–80 years of forest growth on agricultural soils. Forest Ecology and Management **225:**74–81.

Foster, D., F. Swanson, J. Aber, I. Burke, N. Brokaw, D. Tilman, and A. Knapp. 2003. The importance of land-use legacies to ecology and conservation. BioScience **53:**77–88.

Frost, C. C. 1993. Four centuries of changing landscape patterns in the longleaf pine ecosystem. Pages 17–43 in S. M. Hermann, editor. 18th Tall Timbers Fire Ecology Conference, 30 May–2 June 1991. Tall Timbers Research Inc., Tallahassee, Florida.

Garten, C. T. Jr. 2006. Predicted effects of prescribed burning and harvesting on forest recovery and sustainability in southwest Georgia, USA. Journal of Environmental Management **81:**323–332.

Garten, C. T. Jr, and T. L. Ashwood. 2004. Modeling soil quality thresholds to ecosystem recovery at Fort Benning, GA, USA. Ecological Engineering **23:**351–369.

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.

Grau, H. R., T. M. Aide, J. K. Zimmerman, J. R. Thomlinson, E. Helmer, and X. Zou. 2003. The ecological consequences of socioeconomic and land-use changes in postagriculture Puerto Rico. BioScience **53:**1159–1168.

Gregorich, E. G., M. R. Carter, D. A. Angers, C. M. Monreal, and B. H. Ellert. 1994. Towards a minimum data set to assess soil organic matter quality in agricultural soils. Canadian Journal of Soil Science **74:**367–385.

Griffith, G. E., J. M. Omernik, J. A. Comstock, S. Lawrence, G. Martin, A. Goddard, V. J. Hulcher, and T. Foster. 2001. Ecoregions of Alabama and Georgia (color poster with map, descriptive text, summary tables, and photographs). US Geological Survey (map scale 1:1,700,000), Reston, Virginia.

Hilliard, S. B. 1984. Atlas of Antebellum Southern Agriculture. Louisiana State University Press, Baton Rouge, Louisiana.

Iverson, R. M., B. S. Hinckley, R. M. Webb, and B. Hallet. 1981. Physical effects of vehicular disturbances on arid landscapes. Science **212:**915–917.

Jackson, R. B., J. Canadell, J. R. Ehleringer, H. A. Mooney, O. E. Sala, and E. D. Schulz. 1996. A global analysis of root distributions for terrestrial biomes. Oecologia **108:**389–411.

Jobbagy, E. G., and R. B. Jackson. 2000. The vertical distribution of soil organic carbon and its relation to climate and vegetation. Ecological Applications **10:**423–436.

Jones, D. S., and T. Davo. 1997. Land Condition-Trend Analysis Program Summary, Fort Benning, Georgia: 1991–1995. Center for Ecological Management of Military Lands, Colorado State University, Fort Collins, Colorado.

Kalisz, P. J. 1986. Soil properties of steep Appalachian old fields. Ecology **67:**1011–1023.

Kane, S., and R. Keeton. 1998. Fort Benning: the Land and the People. Prepared by the National Park Service, Southeast Archeological Center, Tallahassee, Florida for the U.S. Army Infantry Center, Directorate of Public Works, Environmental Management Division, Fort Benning, Georgia.

Knops, J. M. H., and D. Tilman. 2000. Dynamics of soil nitrogen and carbon accumulation for 61 years after agricultural abandonment. Ecology **81:**88–98.

Koerner, W., J. L. Dupouey, E. Dambrine, and M. Benoit. 1997. Influence of past land use on the vegetation and soils of present day forest in the Vosges mountains, France. Journal of Ecology **85:** 351–358.

LaJeunesse, S. D., J. J. Dilustro, R. R. Sharitz, and B. S. Collins. 2006. Ground layer carbon and nitrogen cycling and legume nitrogen inputs following fire in mixed pine forests. American Journal of Botany **93:**84–93.

Latty, E. F., C. D. Canham, and P. L. Marks. 2004. The effects of land-use history on soil properties and nutrient dynamics in northern hardwood forests of the Adirondack Mountains. Ecosystems **7:**193–207.

Lawton, D. E. 1976. Geologic Map of Georgia. Georgia Department of Natural Resources, Atlanta, Georgia.

Lunt, I. D., and P. G. Spooner. 2005. Using historical ecology to understand patterns of biodiversity in fragmented agricultural landscapes. Journal of Biogeography **32:**1859–1873.

McLauchlan, K. 2006. The nature and longevity of agricultural impacts on soil carbon and nutrients: a review. Ecosystems **9:**1364–1382.

Paul, K. I., P. J. Polglase, J. G. Nyakuengama, and P. K. Khanna. 2002. Change in soil carbon following afforestation. Forest Ecology and Management **168:**241–257.

Post, W. M., and K. C. Kwon. 2000. Soil carbon sequestration and land-use change: processes and potential. Global Change Biology **6:**317–327.

Richter, D. D., D. Markewitz, S. E. Trumbore, and C. G. Wells. 1999. Rapid accumulation and turnover of soil carbon in a re-establishing forest. Nature **400:**56–58.

Schlesinger, W. H. 1990. Evidence from chronosequence studies for a low carbon-storage potential of soils. Nature **348:**232–234.

Steenwerth, K. L., L. E. Jackson, F. J. Calderón, M. R. Stromberg, and K. M. Scow. 2003. Soil microbial community composition and land use history in cultivated and grassland ecosystems of coastal California. Soil Biology and Biochemistry **35:**489–500.

Switzer, G. L., M. G. Shelton, and L. E. Nelson. 1979. Successional development of the forest floor and soil surface on upland sites of the East Gulf Coastal Plain. Ecology **60:**1162–1171.

Unger, P. W., and T. C. Kaspar. 1994. Soil compaction and root growth: a review. Agronomy Journal **86:**759–766.

USAIC (United States Army Infantry Center). 2001. Integrated Natural Resources Management Plan, Fort Benning Army Installation 2001-2005. U.S. Army Infantry Center, Fort Benning Army Installation, Georgia/Alabama.

Verheyen, K., B. Bossuyt, M. Hermy, and G. Tack. 1999. The land use history (1278–1990) of a mixed hardwood forest in western Belgium and its relationship with chemical soil characteristics. Journal of Biogeography **26:**1115–1128.

Wigginton, J. D., B. G. Lockaby, and C. C. Trettin. 2000. Soil organic matter formation and sequestration across a forested floodplain chronosequence. Ecological Engineering **15:**S141–S155.

Ziegler, S. S. 2004. Composition, structure, and disturbance history of old-growth and second-growth forests in Adirondack Park, New York. Physical Geography **25:**152–169.

Zimmerman, J. K., T. M. Aide, M. Rosario, M. Serrano, and L. Herrera. 1995. Effects of land management and a recent hurricane on forest structure and composition in the Luquillo Experimental Forest, Puerto Rico. Forest Ecology and Management **77:**65–76.