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Author(s): Michael A. Bashkin and Dan Binkley
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CHANGES IN SOIL CARBON FOLLOWING AFFORESTATION IN HAWAII

MICHAEL A. BASHKIN[1] AND DAN BINKLEY

Department of Forest Sciences and Graduate Degree Program in Ecology, Colorado State University, Fort Collins, Colorado 80523 USA

Abstract. Afforestation in the tropics may sequester soil C and has been proposed as a management tool to aid in controlling rising levels of atmospheric CO_2. We measured changes in soil C following afforestation of sugarcane fields with fast-growing *Eucalyptus saligna* (Sm.) plantations in Hawaii. Using stable C isotopes, we estimated the contributions to changes in total soil C that were due to the loss of C from the prior cane cultivation, and to the gain of C from the new *Eucalyptus* plantations. Total soil C 10–13 yr after afforestation was 114 and 113 Mg/ha, respectively, in the *Eucalyptus* and cane plantation. *Eucalyptus* increased total soil C in the 0–10 cm layer by 11.5 Mg/ha, but that was offset by a loss of 10.1 Mg/ha of cane-derived C from the 10–55 cm layer. The net effect on soil C of afforestation of cultivated lands depends not only on new C gained, but also on C lost from the previous management.

Key words: afforestation; Eucalyptus; *global climate change; Hawaii; land use; soil carbon; stable carbon isotope ratios; sugarcane plantations.*

INTRODUCTION

The global soil C pool is 1000–3000 Pg, more than three times the C in land vegetation, and, depending on land use, this pool may be a source or sink of atmospheric CO_2 (Houghton et al. 1987). Practices such as cultivation, forest harvesting, and afforestation have differing effects on soil C input, accretion, and loss, complicating efforts to complete an accurate model of the global carbon cycle (Tans et al. 1990, Lugo 1992). Cultivation may cause up to a 50% loss of soil C, whereas forest harvesting has no detectable effect, on average (Johnson 1992). Afforestation, defined as forest establishment after ≥50 yr without forest (Evans 1992), is expected to cause accretion of soil C (Lundgren 1978, Bouwman and Leemans 1995). The potential for tropical afforestation with tree plantations is large, and is projected to cover 60×10^6 ha in the next few years (Evans 1992). The area of tropical tree plantations is small compared to the total area of closed forest (1%), but their impact may be significant to the global carbon cycle (Brown et al. 1986, Lugo and Brown 1992).

Tropical tree plantations may sequester enough carbon in aboveground biomass to offset the carbon lost to the atmosphere as a result of land use changes in the temperate zone ($0.03–0.11 \times 10^9$ Mg/yr; Brown et al. 1986). In addition, plantations may sequester C in soil, especially in cases of establishment on cultivated lands where soil C has been depleted (Johnson 1992). For example, soil C increased by 0.8–4.0 $Mg \cdot ha^{-1} \cdot yr^{-1}$ during secondary forest succession on land that had been cultivated for 100–300 yr in Puerto Rico (Lugo and Sanchez 1986). One recent estimate (Bouwman and Leemans 1995) assumed that afforestation stores 50 Mg/ha of soil C in 30 yr, giving a tentative estimate of global soil C accumulation rate in tropical tree plantations of 0.07 Pg/yr. These rates are based on very few studies, and most results come from unreplicated, descriptive studies rather than designed experiments.

On the island of Hawaii, a unique opportunity exists to study changes in soil C following afforestation of cultivated land. Sugarcane dominated the island's agricultural activity for most of this century, covering >100 000 ha (Sugarcane Planters' Association 1992). Cane production is now unprofitable, and one potential use for this land is afforestation with fast-growing tree plantations, particularly *Eucalyptus* species (Whitesell et al. 1992). This shift in land use from cane (with a C_4 photosynthetic pathway) to *Eucalyptus* (a C_3 pathway genus) allows the use of stable C isotope ratios to examine changes not only in total soil carbon (TC), but also in its derivation (SOC, soil organic carbon) from cane (SOC_4) and *Eucalyptus* (SOC_3). We estimated the contributions to the change in TC from the loss of C from the antecedent cane agriculture, and the gain of C from the new tree plantations.

METHODS

Study sites

The study area is located on the island of Hawaii's Hamakua coast (19°30′ N, 155°15′ W). Average annual precipitation of 3.0–4.6 m is evenly distributed throughout the year, and average annual temperature is 21°C, with little seasonal variation (Debell et al. 1989).

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[1] Present address: Natural Resource Ecology Laboratory, Colorado State University, Fort Collins, Colorado 80523 USA.

TABLE 1. Site history and characteristics of afforested study plots in Hawaii.

Site†	Elevation (m)	Start cane cultivation	End cane cultivation	Cane yield ($Mg \cdot ha^{-1} \cdot yr^{-1}$)	*Eucalyptus* planted
Chinchuck	395	1915	1980	10–13	1982
Kamae‡	395	1940	1977	10–13	1982
Peepkeo 1	300	1900	1984	18–20	1984
Peepkeo 2	325	1920	1985	18–20	1985
Wailea	30	1900	1985	18–20	1985
Amaulu§	400				1982§

† Every two years from 1920, 85, 75, and 110 kg/ha of N, P, and K were added, respectively; since 1955, all sites were treated with $CaCO_3$ at 700 kg/ha. In addition, all sites were ripped to 40 cm depth every 4–6 yr. Data were supplied by T. Crabb (*personal communication*).

‡ Contains a paired wildland forest site.

§ *Eucalyptus* plantation established on wildland forest.

Elevation ranges from 15 to 400 m, with gentle slopes of 0–15%. Soils are classified as isothermic Typic Hydrudands of the Akaka and Kaiwiki series (USDA Soil Conservation Service 1973). These volcanic ash soils are characterized by amorphous noncrystalline hydrous alumina and iron oxides mineralogy (Mizota and van Reeuwijk 1989). Typically, these soils have low bulk density (~0.35 kg/L, oven-dried basis) and a very fine structure that adsorbs large amounts of water and humus (Wada and Wada 1976).

Five paired sites were located containing *Eucalyptus saligna* (Sm.) plantations established on, and adjacent to, abandoned cane fields. Since 1920, the cane fields were fertilized with 85, 75, and 110 $kg \cdot ha^{-1} \cdot yr^{-1}$ of N, P, and K, respectively, and were tilled to 40 cm every 4–6 yr. Since 1955, cane fields were also limed with $CaCO_3$ at 700 kg/ha every two years. The *Eucalyptus* plantations were planted at 3 × 3 m spacing and were fertilized for the first three years (Whitesell et al. 1992). Understory species in all *Eucalyptus* sites contained a mixture of C_3 and C_4 vegetation. Two wildland forest sites (never under management), dominated by introduced *Guava* and native *Koa acacia,* were also found adjacent to cane plantations at Amaulu and Kamae (Table 1). At Amaulu, the adjacent *Eucalyptus* plantation was established on soils that were never in cane production; it was analyzed separately from the five paired cane and *Eucalyptus* sites. The other wildland forest site (Kamae) had an adjacent *Eucalyptus* plantation on a former cane site. All sites were within 17 km of each other, with similar history and characteristics (Table 1).

Soil sampling

In each site, three soil cores were taken every 5 m along a transect 10 m from the plantation edge. A parallel transect was located in the cane field 10 m from the edge of the plantation. Each core was 6 cm in diameter and, in most cases, 1 m in length. The cores were divided into 0–5, 5–10, 10–15, 15–20, 20–25, 25–40, 40–55, 55–70, 70–85, and 85–100 cm depths, and then were air-dried and transported to Colorado State University for analyses.

Soil samples were passed through a 2-mm sieve and oven-dried at 70°C to a constant mass. Bulk density was calculated by dividing the oven-dry mass of the <2 mm fraction by the volume of the core segment. From each site, bulk density was averaged over three cores for each layer in abandoned cane fields, *Eucalyptus* plantation, or wildland forest.

Carbon analyses

Because soils had been limed to increase cane production, we acidified soil samples to remove any residual $CaCO_3$. A composite 10-g sample from the three cores for each depth and each site of *Eucalyptus,* cane, or wildland forest soil was placed into a 40-mL centrifuge tube. We added 15 mL 0.1 mol/L HCl and placed the tubes on a reciprocating shaker for 12 h. Tubes were then centrifuged for 20 min at 15 000 rpm and the supernatant was aspirated off. We added 25 mL deionized water to each tube, centrifuged, and then aspirated off the supernatant. This sequence was repeated twice. Finally, the entire soil pellet was transferred to a tin container, was oven-dried at 70°C to a constant mass, and then was ground to a fine powder.

Percent carbon was determined on a CHN analyzer by dry combustion (LECO-1000, LECO Corporation, St. Joseph, Michigan, USA). Total soil C (in megagrams per hectare) was calculated by multiplying the C concentration by the bulk density and by depth of the soil layer.

The C isotope composition was determined utilizing the VG isochrom-NA stable isotope ratio mass spectrometer (VG, Middlewich, UK). The C isotopic composition was expressed as δ values relative to the PDB standard and was calculated as follows:

$$\delta^{13}C\ (‰) = (R_{\text{sample}} - R_{\text{standard}}) \times 10^3 \qquad (1)$$

where $\delta^{13}C$ is the per mil (‰) difference between the ^{13}C content of the sample and the standard, and R is the mass $^{13}CO_2/^{12}CO_2$ ratio of the sample or standard. Each sample was run twice and the two values were averaged. Fewer than 5% of the sample duplicates were not within 0.4‰; these were run a third time and all three values were averaged.

The TC pool was divided into carbon derived from

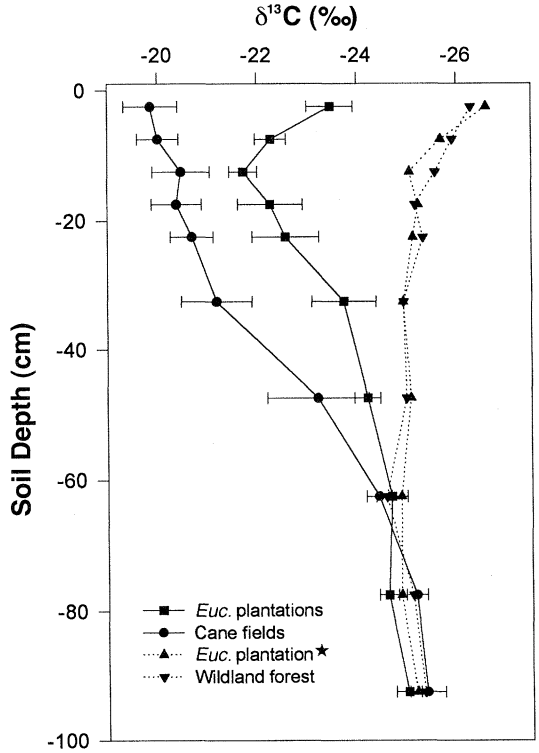

FIG. 1. The $\delta^{13}C$ values, by depth, of wildland forest ($n = 2$), cane fields ($n = 5$), and *Eucalyptus* plantations ($n = 5$) in Hawaii; bars indicate the standard error of the mean. The starred *Eucalyptus* plantation (▲) was established on wildland forest with no former cane cultivation.

C_3 or C_4 vegetation. Percentage TC derived from cane (% SOC_4) and percentage TC derived from C_3 (% SOC_3) plants for each layer were calculated as follows (Vitorello et al. 1989):

$$\% \, SOC_4 = (\delta - \delta_o/\delta_c - \delta_o)100 \quad (2)$$

$$\% \, SOC_3 = 100 - \% \, SOC_4 \quad (3)$$

where δ is the $\delta^{13}C$ value of the soil sample, δ_o is the $\delta^{13}C$ value of soil from the wildland forest soil, and δ_c is the $\delta^{13}C$ value of a composite sample of cane litter and roots from all sites (−11.52‰; M. Bashkin, *unpublished data*). A sensitivity analysis using −14.52‰ and −8.52‰ for δ_c showed a shift in percentage of SOC_4 in cane fields of +3.3% and −2.3%, respectively.

The percentages of SOC_4 and SOC_3 were multiplied by TC to obtain C (in megagrams per hectare) derived from *Eucalyptus* (SOC_3) and cane (SOC_4) vegetation in each soil layer. Net carbon change of SOC_3, SOC_4, and TC due to 10–13 yr tree growth was calculated as the difference between the pools in the *Eucalyptus* plantation minus respective pools from the adjacent cane site for each layer. A key assumption was that no substantial losses occurred from the wildland soil C prior to cane cultivation. We did not correct for changes in bulk density (Veldkamp 1994) because there was no difference in bulk density between cane and *Eucalyptus* ($P = 0.98$, paired t test).

Statistical analyses

Land use had no effect on $\delta^{13}C$ values below 55 cm (Fig. 1), so only values from 0–55 cm soil depth were used for analyses. Similarly, Detwiler (1986) reported that most studies found no effect of land use on soil C below 40 cm in the tropics. TC, SOC_4, and SOC_3 were tested for significant differences by using a split-plot ANOVA with site × land use × depth as error terms (SAS 1987). For statistical comparisons, the land use term refers only to cane vs. *Eucalyptus* management; wildland forests were not included because of insufficient replication. Data were tested for differences based on the entire measured layer (0–55 cm), or were

TABLE 2. Average $\delta^{13}C$ values for wildland forest, cane, and *Eucalyptus* soils, and percentage SOC_4 and percentage SOC_3 carbon calculated from Eqs. 2 and 3 for cane and *Eucalyptus* plantations.

	Wildland forest†	Cane‡			*Eucalyptus*‡		
Depth (cm)	$\delta^{13}C$ (‰)	$\delta^{13}C$ (‰)	SOC_4 (%)	SOC_3 (%)	$\delta^{13}C$ (‰)	SOC_4 (%)	SOC_3 (%)
0–5	−26.27	−19.87	43	57	−23.24	19	81
5–10	−26.03	−20.01	41	59	−22.28	26	74
10–15	−25.54	−20.48	36	64	−21.73	27	73
15–20	−25.29	−20.39	36	64	−22.27	22	78
20–25	−25.30	−20.70	33	67	−22.58	20	80
25–40	−25.07	−21.19	29	71	23.74	10	90
40–55	−25.12	−23.22	14	86	−24.21	7	93
55–70	−24.88	−24.43	3	97	−24.69	1	99
70–85	−24.88	−25.18	0	100	−24.62	2	98
85–100	−25.19	−25.39	0	100	−25.01	1	99

Note: SOC_4 and SOC_3 refer to the percentages of total soil carbon (TC) derived from C_4 and C_3 vegetation, respectively. SOC is soil organic carbon.

† Wildland forest data are the mean of two sites.

‡ Data are the mean of five sites.

TABLE 3. Mean soil carbon (Mg/ha) as total soil carbon (TC), SOC_4, and SOC_3, by depth, in wildland forest, *Eucalyptus*, and cane plantations.

Depth	Wildland forest†	*Eucalyptus*‡			Cane‡			SOC change (*Eucalyptus* − cane)		
(cm)	TC	TC	SOC_4	SOC_3	TC	SOC_4	SOC_3	TC	SOC_4	SOC_3
0–5	25.7	23.2 ± 3.9	4.7 ± 1.1	18.6 ± 4.4	16.2 ± 3.5	7.4 ± 2.9	8.7 ± 0.7	7.0	−2.7	9.9
5–10	22.9	16.2 ± 4.4	3.8 ± 0.9	21.3 ± 3.8	11.7 ± 2.7	4.8 ± 1.8	6.9 ± 1.0	4.5	−1.0	5.4
10–15	16.4	11.8 ± 3.1	2.9 ± 1.0	8.9 ± 2.2	12.0 ± 3.0	4.2 ± 1.9	7.8 ± 1.5	−0.2	−1.3	1.1
15–20	13.3	11.2 ± 3.2	2.6 ± 1.6	8.6 ± 2.1	12.4 ± 1.7	4.5 ± 1.6	7.9 ± 0.7	−1.2	−1.9	0.7
20–25	10.4	9.6 ± 2.9	2.0 ± 1.6	7.6 ± 1.8	11.3 ± 1.9	3.6 ± 0.8	7.6 ± 1.7	−1.7	−1.6	0.0
25–40	21.6	22.8 ± 5.8	2.3 ± 3.4	20.5 ± 4.2	31.5 ± 6.1	9.4 ± 1.8	22.1 ± 0.9	−8.7	−7.1	−1.6
40–55	19.5	19.2 ± 4.1	1.1 ± 0.7	18.1 ± 4.2	17.5 ± 1.0	3.3 ± 5.7	14.2 ± 5.8	1.7	−2.2	3.9
Total	129.8	114	19.4	94.6	112.6	37.2	75.2	1.4	−17.8	19.4

† Mean of two sites.
‡ Mean of five sites ± 1 SD.

partitioned into the surface layer (0–10 cm) and lower layer (10–55 cm), although data for each depth are reported. When there was a significant interaction between land use × depth, the main effect of land use was determined with Bonferroni-adjusted, paired *t* tests to control for Type I error. Two-tailed tests were used for TC and one-tailed tests were used for SOC_4 (decreased under *Eucalyptus*) and SOC_3 (increased under *Eucalyptus*).

RESULTS

$\delta^{13}C$ values of soil profiles

The $\delta^{13}C$ values for the soils ranged from −19.9 to −26.3‰ in the surface layers, converged to −24.7 below 55 cm, and then decreased slightly to −25.2‰ at 1 m depth (Fig. 1). For the wildland forest, soil $\delta^{13}C$ values ranged from −26.27 at the surface to −25.2‰ at 1 m depth; these are typical for soil under C_3 vegetation (Boutton 1991). The *Eucalyptus* plantation established on wildland forest soil (Amaulu site) shows similar $\delta^{13}C$ values, even though C_4 plants are present in the understory. This suggests that errors in estimates attributed to both vegetation types in the plantation understory are minimal. The $\delta^{13}C$ values of soil C ranged from −19.9 to −25.4‰ and −21.7 to −25.0‰ for the cane and *Eucalyptus* plantations at the surface and at 1 m, respectively (Fig. 1, Table 2).

On average, SOC_4 in the cane fields accounted for 42% of TC in the surface 0–10 cm layer and decreased steadily to 14% in the 40–55 cm layer. With *Eucalyptus* afforestation, the proportion of SOC_4 decreased to 22% and 7%, and the proportion of SOC_3 increased to 78% and 93% in the surface layer and the 40–55 cm layer, respectively (Table 2).

TABLE 4. *P* values for effects of land use and the interaction of land use × depth on TC, SOC_4, and SOC_3, detected by paired *t* test and split-plot ANOVA.

Source	Soil organic C pool		
	TC	SOC_4	SOC_3
Land use†			
0–55 cm	0.73	0.02	0.02
0–10 cm	0.04	0.09	0.01
10–55 cm	0.25	0.02	0.17
Land use × depth‡	0.04	0.06	0.02

† Paired *t* test, $n = 5$, df = 4.
‡ Error term = land use × site × depth, df = 6.

Effect of land use on TC, SOC_4, and SOC_3

In the top 55 cm of soil, land use (referring only to cane vs. *Eucalyptus* management) had no effect on TC ($P = 0.73$; Tables 3 and 4). Total soil C averaged 114 and 113 Mg/ha for the *Eucalyptus* and cane plantations, respectively. However, land use interacted with soil depth ($P = 0.04$); total C for *Eucalyptus* plantations increased by 11.5 Mg/ha in the surface layer ($P = 0.04$), but this appeared to be offset by a net C loss of 10.1 Mg/ha in the 10–55 cm layer ($P = 0.25$).

After 10–15 years of *Eucalyptus* growth, SOC_4 declined from an average across sites of 37.2 to 19.4 Mg/ha in the top 55 cm of soil ($P = 0.03$). Land use again interacted with soil depth ($P = 0.06$). Soil SOC_4 decreased by only 3.7 Mg/ha in the surface layer ($P = 0.09$), but declined by 14.1 Mg/ha in the lower layer ($P = 0.02$). SOC_3 increased from 75.2 to 94.6 Mg/ha in the top 55 cm of soil ($P = 0.02$). However, this effect was different between the surface and lower layers (land use × depth; $P = 0.02$). SOC_3 increased by 15.3 Mg/ha in the surface layer ($P < 0.01$) and did not significantly change in the lower layer ($P = 0.17$) (Table 3, Fig. 2).

DISCUSSION

Effect of cultivation on wildland forest soil C

Our best estimate for the loss of native soil C is 54.6 Mg/ha, or a 42% loss after 65 years. Because the sugarcane added 37.2 Mg/ha, TC declined by 17.2 Mg/ha from levels in the wildland forest. This represents a 13% net loss of TC less than other studies have reported. For example, on native forest in Brazil, Vitorello et al. (1989) reported a loss of 48 and 34 Mg/ha (38% and 27% loss) from the 0–70 cm layer after cul-

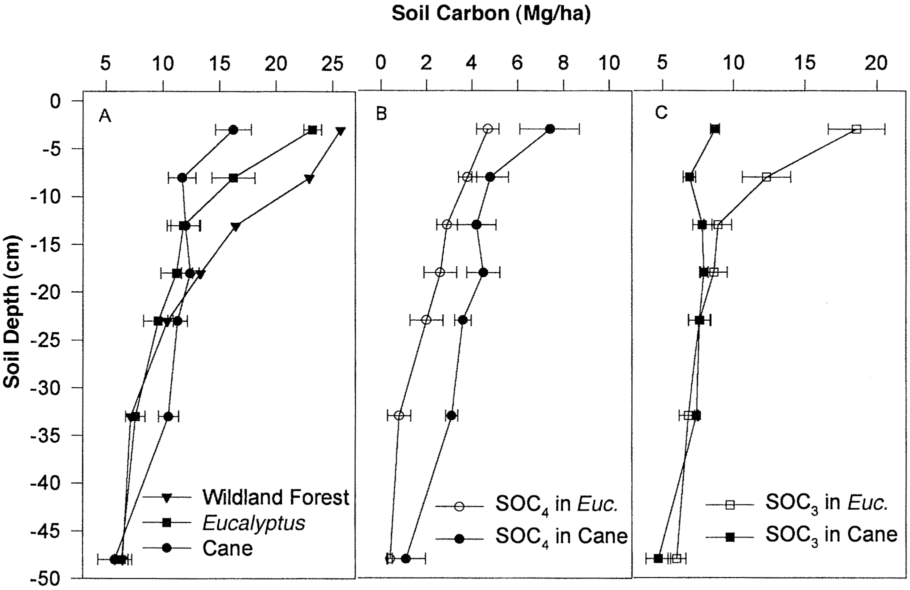

FIG. 2. (A) Total soil carbon (TC) by depth for native wildland forest (n = 2), cane fields (n = 5), and *Eucalyptus* plantations (n = 5) in Hawaii; bars indicate the standard error of the mean. (B) SOC$_4$ in cane fields and after 10–13 years of afforestation with *Eucalyptus*. (C) SOC$_3$ in cane fields and after 10–13 years of afforestation with *Eucalyptus*.

tivation in sugarcane for 12 and 50 yr, respectively. More generally, Schlesinger (1986) reported a mean loss of 21% soil C from 19 postcultivation studies, with a range of 2–69% loss. Detwiler (1986) estimated a loss of 40% compiled from 48 tropical sites with a range of 1–81%. From a compilation of reviews and other studies, Johnson (1992) reported a >50% mean loss in soil C following cultivation worldwide. Davidson and Ackerman (1993) reported a loss of 20–40% from 56 paired plots from 18 studies worldwide.

The soil C losses may be less in our study for two reasons. First, the productivity of these cane fields was very high (10–20 Mg·ha^{-1}·yr^{-1}). Even though harvest removed much of the biomass, C inputs from roots would have been large. Second, andic soils stabilize organic matter and retard decomposition by complexing with Al and Fe (Mizota and van Reeuwijk 1989, Veldkamp 1994).

Effect of afforestation on soil C

Johnson (1992) found that most studies reported a >35% increase in soil C following reforestation and afforestation of cultivated lands. However, the only study from the tropics reported that soil C increased 0.8 Mg·ha^{-1}·yr^{-1} in the wet-tropical life zone (Lugo and Sanchez 1986). Brown and Lugo (1990) (not included in Johnson 1992) found a similar soil C increase of 1–2 Mg·ha^{-1}·yr^{-1} in the wet-tropical life zone. Zou and Bashkin (1997) reported an increase of 2.8 Mg·ha^{-1}·yr^{-1} following reforestation of cane fields in Hawaii; however they examined only a composite value for the top 25 cm of soil. Alternatively, Richter et al. (1994) reported a negligible increase of 0.072 Mg·ha^{-1}·yr^{-1} in the top 7.5 cm of soil after 28 years of loblolly pine growth on historically cultivated Ultisols in the South Carolina piedmont. In the present study, TC showed no net increase after 10–13 years of afforestation in the top 55 cm of soil. However, TC in the surface layer did increase by 11.5 Mg/ha, but this was offset by a decrease of 10.4 Mg/ha in the lower layer (Fig. 2).

The loss of TC in the lower layer probably resulted from the change in land use. Tilling homogenized the soil C content in the 5–40 cm depth, resulting in less soil C at the surface, but more in the lower layer compared to the wildland forest soil (Fig. 2A). Other studies have also found similar redistributions of soil C in cultivated soils (Detwiler 1986). In contrast, the wildland forest soil C showed a typical distribution, with an exponential decrease from the surface to the lower layers. Following afforestation, soil C inputs to the lower layers dropped substantially and C was lost from this portion of the profile. Partitioning the TC pool into SOC$_4$ and SOC$_3$ supports this scenario (Fig. 2B, C).

The TC gained in the surface layer is SOC$_3$ and the C lost in the lower layer is largely SOC$_4$. The content of SOC$_3$ increased by 0.82 Mg/ha in the lower layer, compared with a loss of 3.7 Mg/ha of SOC$_4$ from the surface layer. However, these changes are small compared to the 15.3 Mg/ha SOC$_3$ gain in the surface and a 14.1 Mg/ha SOC$_4$ loss in the lower layer.

The prediction that TC will increase after 10–13 years of afforestation was not met, and future sampling is required to detect if this condition will persist. The trends in SOC$_4$ and SOC$_3$ may be curvilinear or linear, and only subsequent sampling can characterize the dynamics of these pools over time. The majority of SOC_4 may have been lost within the first few years following afforestation, and the remainder may have moved into slower cycling pools. Conversely, SOC_3 may not have stabilized and could continue to increase throughout the soil profile.

The net effect on total soil carbon of afforestation after cultivation is dependent not only on new C gained, but also on C lost from previous management. Afforestation may not lead to substantial increases in soil C, and the current assumption that afforestation can sequester 50 Mg/ha in 30 years (Bouwman and Leemans 1995) may be a large overestimate. Additional experimentation is needed on soil C following afforestation over longer time scales, with different tree species and cultivation regimes and in other soil types and environments.

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