Research Square

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Long-term compost application alters carbon stabilization and increases mineral- associated carbon throughout urban soil profiles

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Abstract

Long-term organic matter amendments can increase soil carbon stocks, but how this soil carbon may be distributed and stabilized throughout a profile is less understood. In addition, soil carbon stabilization and processing may differ in soils where baseline carbon is high, which is common in intensively managed urban soils covered in turfgrass, such as recreational parks.. To address these questions, we sampled 14 fields to one meter in recreational parks in Pleasanton, CA; eight with long-term compost application (20 and 30-years) and six that had never received compost application and characterized soil carbon changes using density fractionation, radiocarbon, and DRIFTS. We found that unamended control sites had high carbon stocks (239 Mg C ha^{-1} to 100 cm) relative to other managed systems while compost sites had26% higher soil carbon stocks (302 Mg ha^{-1}, $p < 0.05$).Soil carbon increases were concentrated in the 0–10 cm and 75–100 cm depths, highlighting the ability for surface-applied compost to increase deep soil carbon stabilization over time. The mineral fraction contained most of these deep soil gains, with 70% increases in the mineral fraction carbon content at compost sites ($p < 0.05$) and with a shift in containing more microbial OM. Overall, this study supports the idea that long-term compost application can alter persistent soil carbon to at least 1 m in high organic matter soil systems and that the increases in soil carbon from compost application can occur within the mineral fraction on decadal timescales.

1. Introduction

Organic matter amendments like compost are used extensively as slow-release fertilizers used to increase soil carbon (C) and improve soil health (Goldan et al., 2023; Hargreaves et al., 2008; Malone et al., 2023). However, most research on compost and other exogenous organic matter (OM) additions occur on short timescales (< 5 years) which limits our understanding of long-term and persistent changes in soil C dynamics. Some studies on the long-term (> 20-years) impacts of OM applications have shown soil C increases not only in the topsoil, but also at depth and in the mineral associated pool (Mayer et al., 2022; Tian et al., 2017; Villa et al., 2024). Several mechanisms lead to deep soil transit of C including irrigation, tillage, and deeply rooted plants (Sokol et al., 2022a). Once C arrives at depth, C can be more readily stabilized in the mineral associated pool than in upper layers as the fraction of fine silt, clay, and Fe/Al oxides increases throughout the profile (Rumpel & Kögel-Knabner, 2011; Villa et al., 2024). These findings suggest that repeated OM additions could influence long-term and deep C dynamics, but significant work remains as few replicated studies have been conducted.

There are limited data on the impacts of long-term compost additions throughout a deep soil profile (up to 1 meter) across most managed systems, particularly urban soils. While intensively managed urban soils are a practical place for compost use, previous studies have been short (typically three or fewer years) and limited to shallow soils (Malone et al., 2023). Urban land is approximately 5% of total US land area. A large portion of urban land is intensively managed with turfgrass cover, constituting 2% of total US land area (16,000,000 ha) (Z. Liu et al., 2014; Milesi et al., 2005). Turfgrass systems are a particularly interesting but relatively untested use case for long-term compost additions as they often have high soil C concentrations due to rapid plant growth above and belowground, commonly aided by fertilization and irrigation (Phillips et al., 2022; Qian & Follett, 2012). Soil organic C concentrations in mature turfgrass soils often exceed 5% in the top 10 cm with increases in the first 30-years after establishment, reaching an apparent limit around 50 years post-establishment (Phillips et al., 2022). Given relatively high soil C content, there is uncertainty in how much additional C mature turfgrass soils may store under compost additions. Studying a turfgrass system with long-term exogenous OM application could help to understand soil C saturation, limits to C accrual, and even if limits are soft slowdowns rather than hard caps on soil C storage (Georgiou et al., 2022, 2025).

Studies that find a limit in turfgrass soil C accrual, or limits in C accrual in other high soil C systems, have focused on bulk soil C (Phillips et al., 2022; Qian & Follett, 2012). This omits both how the persistence of soil C may change with long term management as well as changes in the long cycling mineral fraction, despite mineral associated OM constituting 65% of global soil C (Sokol et al., 2022b). One concern is that the C gains observed in turfgrass soils may be due to increases in particulate organic matter (POM), the least persistent pool, due to accumulation of undecomposed root and grass fragments from mowing (Kerek et al., 2002). In one of the few studies of mineral-associated organic matter in urban turfgrass ecosystems, Kerek et al. (2002) observed increases in mineral-associated organic matter of turfgrass soils over time (30-years) in the top 15 cm of the soil, but this was to a lesser extent than gains observed in POM (Kerek et al., 2002). Across land uses, even high quality soils are typically not close to their mineral-associated C capacity, though gains of C in this pool may be more difficult to detect than gains in the POM pool (Georgiou et al., 2022). This makes the study of mineral-associated organic matter in long-term turfgrass systems of particular interest for understanding high C soils at large.

One reason for gaps in understanding limits to soil C accrual in high soil C systems is that deep soil (> 30 cm) C dynamics are greatly understudied compared to top soils (< 30 cm), especially in urban soils (Malone et al., 2023; Pries et al., 2023; Yost & Hartemink, 2020). This gap persists despite knowledge that over 50% of soil C is stored below 30 cm and that deep soils can be actively managed to sequester C. C cycling in deep soils is governed principally by mineralogy and inputs from shallower soil layers. When the topsoil is managed for high plant and microbial biomass, vertical C transit supports organo-mineral formation at depth (Pries et al., 2023). This potential for topsoil organic C translocation is notable in managed systems that receive long-term irrigation, such as urban parks, as irrigation can drastically change the residence time and storage of C (Han et al., 2010; Jobbágy & Jackson, 2000; Skadell et al., 2023). Compost application could further increase deep C storage in soils with long-term irrigation as it allows for additional dissolved organic C (DOC) transit, with Wright et al. (2008) observing increased DOC at 15 cm sustained for 29 months after a one-time compost application in turfgrass soils (Stergiadi et al., 2016; Wright et al., 2008). Thus, deep soil sampling is vital to understanding changing soil C dynamics in high C soil systems with long-term compost applications, as irrigation may significantly transit. Restricting soil sampling at 30 cm or less would lead to an incomplete understanding of deep soil C dynamics in a soil system that has high C translocation potential.

Our objective was to assess the impacts of long-term (20 and 30-years) additions of compost on total soil and mineral-associated organic C in urban turfgrass systems to a 1 m depth. We hypothesized that 1) unamended soils would have high topsoil organic C concentrations (> 3%), with older parks containing more C due to additional years of high levels of plant OM inputs. 2) Turfgrass soils with long-term compost additions would have greater amounts of total soil C, as well as greater amounts of mineral-associated organic matter, relative to unamended soils. Compost itself will supply additional OM into the soil system and support additional plant production that further increases total C input to the soil. 3) Organic matter from compost will be incorporated into mineral associated organic C pools, including at depth, as there is a high ability for translocation due to long-term irrigation.

2. Materials and Methods

2.1 Study sites and experimental design

The study sites are located in the city of Pleasanton, California (37.6604° N, 121.8758° W) on publicly managed athletic parks with turfgrass cover. A total of six unamended parks, one 20-year site, and one 30-year site. Among these parks, there were 14 fields sampled (six unamended, three at the 20-year site, and five 30-year site). Compost

was derived from green waste feedstocks and was sourced from the same vendor (Recology Blossom Valley Organics, Vernalis, CA). The 20-year site is estimated to have received a total of 68.6 Mg C ha^{-1} (2.8 Mg C ha^{-1} y^{-1}) of compost additions while the 30-year site is estimated to have received 42.7 Mg C ha^{-1} (1.4 Mg C ha^{-1} y^{-1}) from compost additions. Unamended parks have no record of receiving compost for at least 30-years. All parks receive additional nutrients either from mineral fertilizer or recycled water. Each park receives irrigation three times a week, herbicides and pesticides annually, and share similar grass species with seed mixes that are applied at similar rates. All parks are used for soccer, except for one which is used for baseball outfields. The park age ranges from approximately 59 years to 15 years (Table 1). These parks are also placed throughout the city, leading to four soil orders and numerous soil textures and taxonomic classes being represented in the sample set (Table 1). The factors within soil taxonomic classes in urban soils are important in determining soil function, but can be less informative in urban systems due to heavy anthropogenic disturbance and management (Ajmone-Marsan et al., 2016).

A total of 384 soil samples were collected between March 4 and March 12, 2024. Soil was sampled at six locations within each field at various depths. Sampling locations within each field were randomly selected with the criteria that each sampling location was at least 10 m away from the next closest sampling location. Soil samples were collected with a 5.5 cm diameter auger at the following depth intervals: 0–10, 10–30, 30–50 for three of six sampling locations and 0–10, 10–30, 30–50, 50–75, and 75–100 cm for three additional sampling locations. At each location that was sampling to 100 cm, an additional sample was collected for microbial biomass C determination at depths of 0–10 cm and 10–30 cm. The first sampling location of each field was sampled with a quantitative bulk density corer at depths of 0–10 and 10–30 cm.

Table 1
Description of field site and management practices. Soil sample total represented by n.

Site	Compost Application Rate	Fertilizer	Irrigation	Year Opened	Taxonomic Class	n
Amaral	0	Yes	Regular	1997	Fine-silty, mixed, superactive, nonacid, thermic Mollic Xerofluvents	24
Creekside	0	No	Recycled Water	2002	Fine-silty, mixed, superactive, nonacid, thermic Mollic Endoaquepts	24
Muirwood	0	Yes	Regular	1970	Fine, smectitic, thermic Xeric Endoaquerts	24
Upper Pleasanton	0	Yes	Regular	1965	Fine, smectitic, thermic Mollic Palexeralfs	24
Val Vista	0	Yes	Recycled Water	2004	Fine, smectitic, thermic Xeric Endoaquerts	24
Bernal	0	Yes	Regular	2009	Fine, thermic Typic Calciaquolls	24
Amador	20-years, 204 Mg ha^{-1} total	Yes	Regular	1970	Fine-silty, mixed, superactive, nonacid, thermic Mollic Endoaquepts	72
Ken Mercer	30-years, 153 Mg ha^{-1} total	No	Recycled Water	1970	Fine, thermic Typic Calciaquolls	120

2.2 Soil processing, pH, EC, and soil texture

Soil pH, electrical conductivity, and texture were measured on air dried subsamples that were composited by depth and plot. Soil pH and electrical conductivity were measured using probes in a 1:2 soil:deionized water ratio (Mclean, 1982). Soil texture was assessed using the PARIO plus particle size analyzer, which uses the integral suspension method (PARIO, Meter Group, 2021). On day one 30 ml of 30% hydrogen peroxide (H_2O_2) and 30 ml of DI water was added to 40 g of soil and left to react overnight to destroy organic matter. Soils that had a high SOM% (5–12%) had this process repeated (Acevedo et al., 2021). These samples were placed on a hot plate at 60°C for four to six hours before adding 100 ml of sodium hexametaphosphate and 250 ml of DI water. The following day, the samples were mixed for five minutes and then added into the sedimentation cylinder where the PARIO device was used to monitor the pressure change over 2.5 hours. After this a subsample of the suspension was taken while the remaining solution was wet sieved using 2000–500 µm, 500–250 µm, and 250–53 µm sieves. After drying in an oven at 105°C for 48 hours, these beakers were weighed, had their tare masses subtracted, and the remaining dry solid masses were input into the PARIO software to assess the particle size distribution.

Soils were also air-dried, sieved at 2 mm, and finely grounded with mortar and pestle to prepare for C and nitrogen (N) concentration and isotopic analysis. Carbonate presence was tested by adding 2M HCl to the soil, with no sites exhibiting an effervescent response. An elemental analyzer (Costech ECS 4010 CHNS-O Elemental Analyzer coupled with a ThermoFisher Delta0V Plus continuous flow isotope ratio mass spectrometer, Costech Analytical Technologies, Inc., Valencia, CA) at the Stable Isotope Ecosystem Laboratory of UC Merced was used for C and N analysis. Elemental concentrations were converted to content by using the equivalent soil mass method (ESM) which used a cubic spline function to estimate cumulative soil C content in the absent of bulk density measurements (Wendt & Hauser, 2013). Reference soil layer masses were calculated for each field to control for differences in soil layer masses and soil density between different parks.

2.3 Microbial biomass C

Microbial biomass C (MBC) was performed using the chloroform fumigation method on the 0–10 cm and 10–30 cm depth samples for three of six replicates per site (Brookes et al., 1985). Once sampled, soils were immediately placed into a cooler with dry ice and transported to UC Merced. MBC was extracted using 0.5 M K_2SO_4 and filtered using a Whatman #42 filter paper (2.5µm pore size). The resulting filtrate was frozen at -20°C until analysis. Prior to analysis, samples were thawed and diluted to a 1:5 ratio (sample: water) and analyzed on a Shimadzu total organic C analyzer (Shimadzu TOC-L, Kyoto, Japan) at the UC Merced Environmental Analytical Lab for both TOC and TN. Microbial biomass C and N was calculated as the difference between the C content of the nonfumigated and fumigated soils, using an extraction efficiency factor of 0.45 and 0.54 for MBC and MBN respectively (Beck et al., 1997).

2.4 Density fractionation

Density fractionation was performed on a subset of samples; one sample of the 0–10, 30–50, and 75–100 cm depths was fractionated at each unamended site while three samples per depth was fractionated at each of the long-term composted fields. Density fractionation was done in a similar manner to Swanston 2005 and Villa 2024 with three pools of C fractionated (Swanston et al., 2005; Villa et al., 2024). The free light fraction (FLF) and occluded light fraction (OLF) constitute the POM pool while the heavy fraction (HF) makes up the minerally-associated organic matter pool. Samples were density separated using a 1.85 g cm^{-3} solution of sodium polytungstate (SPT). 75 ml of SPT was added to 20 g of air-dried soil which was then inverted, allowed to sit for 45 minutes, and then centrifuged at 3470 rpm for 1 hour. The FLF was then aspirated, filtered through a 0.8 µm membrane filter under vacuum, and rinsed five times with 100 ml of deionized water through the filter to remove SPT particles from the FLF. FLF was then placed on an aluminum dish in a 55°C oven for 48 hours, weighed, and transferred to a glass vial for storage.

The remaining solution was then mixed at 1700 rpm with a benchtop mixer and sonicated at 280 j ml^{-1} (282 j ml^{-1}, sd = 16.1) to separate the OLF from the HF. Cylinders were placed in an ice bath during sonication to avoid overheating the soil-SPT solution. After sonication, the cylinder sat for 45 minutes and was then centrifuged at 4000 rpm, aspirated, rinsed, and dried similar to the FLF. To remove the HF, 150 ml of deionized water was added to the cylinder and shaken vigorously, centrifuged for 5 minutes at 3470 rpm, and then the sides were rinsed. After this, the solution was centrifuged at 3470 rpm for 20 minutes and then the solution was aspiration down to the remaining soil pellet. This was done 5 times to rinse SPT from the HF, after which the HF was moved to an aluminum dish and placed in a 105°C oven for 24 hours. The overall mass recovery averaged 85% (± 1) and this did not vary by treatment or depth (Table S-1, $p > 0.05$). The estimated C recovery was 70% (± 10) for density fractionated soils, with no variation by treatment (Table S1, $p > 0.05$), but with more mass loss at depth (Table S1, $p < 0.05$). This larger estimated C loss at depth is within reported literature values for deeper soils which contain a greater proportion of mineral-associate organic matter that could be susceptible to physical dispersion during fractionation and rinsing steps, leading to the loss of colloidal or dissolved organic carbon (Crow et al., 2007; Helbling et al., 2021). The soil C stocks for each fraction were calculated by taking the ESM soil layer stocks and multiplying them by the proportion of C within each fraction for the same corresponding field. If a field had not been density fractionated, then it was removed from the average for the purposes of calculating DF C stocks.

2.5 Radiocarbon analysis

Radiocarbon (^{14}C) analysis was conducted on the HF of density fractionated soils for the depth intervals of 0–10 cm, 30–50 cm, and 75–100 cm. HFs containing 0.5 to 1 mg C were placed in 6 mm quartz tubes with roughly 250 mg of copper oxide to provide oxygen for combustion and a small amount of silver to eliminate sulfur interference. This tube was then placed in a 9 mm quartz tube, placed on a vacuum line to remove atmospheric gases, and flame sealed with a hydrogen torch. After sealing, tubes were placed in an oven at 900°C for at least five hours to convert all soil C to CO_2 gas. The CO_2 gas was graphitized, using H_2 as a reducing agent (Vogel et al., 1984). Samples were then run on the NEC Compact Accelerator Mass Spectrometer at Lawrence Livermore National Lab for ^{14}C analysis (Broek et al., 2021). Radiocarbon isotopic values are corrected for mass dependent fractionation using measured $\delta^{13}C$ values and reported in Fraction Modern (Fm) (Reimer, 2004) and in $\Delta^{14}C$ notation (Stuiver and Polach, 1977).

2.6 Diffuse reflectance infrared Fourier transform spectroscopy (DRIFTS)

Diffuse reflectance infrared Fourier transform spectroscopy (DRIFTS) was performed on the FLF, OLF, and HF samples obtained from the density fraction procedure. This was done using a Fourier transform infrared (FTIR) spectrophotometer with an attached Praying Mantis DRIFTS attachment to capture spectra. Finely ground potassium bromide (KBr) powder was used as a background reference material prior to analysis and ran after every 15 samples. Samples were ground finely with a mortar and pestle and oven dried at 65°C for the light fractions and 105°C for the HF prior to analysis. We performed 300 scans per sample at 4000 to 400 cm^{-1} that were averaged and background corrected. Assessment of the spectra and operation of the instrument was performed using the OPUS software (version 7.5.18; Bruker Optik GmbH, Ettlingen, Germany). The resulting absorbance peak areas within spectra were then integrated within three function groups by wavenumber ranges; C–H stretching compounds representing simple plant OM (2976–2898 cm^{-1} and 2870–2839 cm^{-1}), aromatic C = C compounds representing complex plant OM (1550–1500 cm^{-1}), and a carboxylic C = O stretch group representing microbial OM (1660– 1580 cm^{-1}) (Mainka et al., 2022).

2.7 Statistical analyses

All statistical analyses and data visualization was done using RStudio (R Core Team, 2020). Data was tested for normality using the Shapiro-Wilks test. Pairwise Tukey's Honestly Significant Difference (Tukey HSD) *post hoc* tests were used to discern significances between compost treatments after analysis of variances (ANOVA) were run among all treatments. An alpha value of 0.05 was used for significance testing. Visualization was performed using the "ggplot2" and "ggpubr" packages in R for plots (Kassambara, 2020; Wickman, 2016, p. 2). Code to assess DRIFTS spectra was originally written by Dr. Steven Hall (Hall et al., 2018).

3. Results

3.1 Soil texture, EC, and pH

Across unamended sites, there was no significant change in the content of any size fraction between depths, with large variability overall ($p > 0.05$, Fig. 1, Table S1). For amended samples, there was a decrease in sand content as depth progressed ($p < 0.05$), accompanied by an increase in fine silt and total silt ($p < 0.05$) with subsequent depths (Fig. 1, Table S1). Clay content was higher at the 20-year site for the 10–30 cm ($p < 0.05$), but was higher from 50–100 cm at the 30-year site ($p < 0.1$). Mirroring these depth trends, combined silt and clay content had a 20% relative increase in mass at the 20-year site from 10–50cm ($p < 0.05$) compared to the 30 year site, but had a 10% relative increase in mass from 75–100 cm at the 30-year site compared to the 20-year site ($p < 0.05$).

When comparing control soils to all compost soils, the total sand content was higher in unamended soils at both the 50–75 cm (41% vs 22% total soil mass) and 75–100 cm (37% vs 14% total soil mass) depths ($p < 0.05$). This was accompanied by increased coarse silt (at 0–10 cm and 50–75 cm), fine silt (at 50–75 cm and 75–100 cm), and total silt (at 50–75 cm and 75–100 cm) at compost sites ($p < 0.05$). Clay content had a marginally significant higher value in the 0–10 cm depth at amended sites at 29% compared to 21% for controls ($p < 0.1$). Combined silt and clay content was significantly higher at the amended sites for 50–75 cm (59% vs 77% total soil mass) and 75–100 cm (62% vs 86% total soil mass) compared to unamend samples ($p < 0.05$).

Across all sites, pH increased with soil depth from 7.09 ± 0.07 at 0–10 cm to 8.2 ± 0.09 at 75–100 cm. The 30-year site had significantly higher pH at 10–30 cm (8.27 ± 0.05) than the unamended (7.73 ± 0.11, $p < 0.05$) and the 20-year samples (7.78 ± 0.06, $p < 0.05$) but there were no other differences by treatment (Figure S1A). EC was highly variable and relatively consistent to 1 m within unamended plots ($p > 0.05$, Figure S1B). Within compost treatments, the EC was lower when comparing the 0–10 cm to all lower depths for both compost sites ($p < 0.05$, Figure S1B). The EC within compost sites had a marginally significant decrease when comparing the 30–50 cm depth to the 75–100 cm depth ($p < 0.1$, Figure S1B). Despite these decreases at depth within the compost profile, there was no significant difference by treatment effect for within depth comparisons ($p > 0.05$).

3.3 Soil C and N

Bulk soil C concentrations were significantly higher at 0–10 cm at the 20-year site and amended sites overall and higher in the 75–100 cm at both compost amended sites compared to unamended soils ($p < 0.05$, Table S2). From 10–75 cm the 20-year site had a significantly higher soil C concentration than the 30-year site, and significantly higher compared to unamended soils from 30–75 cm ($p < 0.05$, Table S2).

Soil C stocks were significantly higher at the 30-year site compared to unamended soils in the 0–10 and 75–100 cm depths ($p < 0.05$, Fig. 2). This held true when comparing all composted soils to all unamended soils, as there were significantly more C stocks in the 0–10 cm ($p < 0.05$) and in the 75–100 cm ($p < 0.05$) in composted soils. The remaining depths were not significantly different between treatments. Cumulative soil organic C to 100 cm was least

in the unamended soils (239 ± 15.3 Mg C ha^{-1}), followed by 20-years of compost application (294.1 ± 12.8 Mg C ha^{-1}) and then 30-years of compost (309.7 ± 28.7 Mg C ha^{-1}). These differences were driven both by increased soil C concentrations (Table 2) and changes in soil mass (Figure S2), though bulk density directly measured in the 0–10 and 10–30 cm were not significantly different by treatment ($p > 0.05$). The 0–10 and 75–100 cm depths had a significant increase in C % across all compost plots and in the 30-year soils compared to unamended ($p < 0.05$), while the 30-year site also had significantly increased soil mass in the 0–10 cm over both 20-years and unamended samples ($p < 0.05$). Soil mass was significantly lower in the 75–100 cm depth at both compost sites compared to unamended ($p < 0.05$), despite stocks being significantly higher at the depth for the 30-year site ($p < 0.05$).

Soil N was significantly higher in the sites amended with compost for 30-years compared to the unamended parks at 0–10 cm ($p < 0.05$) and 75–100 cm ($p < 0.05$) depths, but the 30-years park soil N was similar to the 20-year compost parks (Table 2). When comparing all compost soils to unamended soils, compost soils had significantly more N in only the 0–10 cm depth ($p = 0.08$). The δ15N was significantly higher at both the 20 and 30-year site compared to unamended soils in the 0–50 cm depths ($p < 0.05$, Table S2) and marginally higher when comparing all amended samples to unamended samples in the 50–75 cm ($p < 0.1$, Table 2).

The soil C:N ratio increased with depth at the unamended and 20-year composted sites, but showed no trend by depth at the 30-year site (Table 2). Between treatments, the C:N was significantly lower from 0–30 cm at 30-years compared to unamended soils ($p < 0.05$) while it was significantly higher from 50–100 cm for the 20-year site compared to unamended soils ($p < 0.05$).

Table 2
Table of C, N, and C:N by depth and years of compost application. SE indicated by ±.

Treatment	Depth (cm)	Carbon (g kg^{-1} soil)	Nitrogen (g kg^{-1} soil)	δ15N (‰)	C:N
	0–10 cm	**49.5 ± 2.1**	**4.8 ± 0.2**	**5.31 ± 0.26**	**10.3 ± 0.13**
	10–30 cm	17.7 ± 1.6	1.7 ± 0.2	5.04 ± 0.15	10.2 ± 0.14
Unamended	30–50 cm	11.1 ± 0.7	1.1 ± 0.1	5.12 ± 0.13	10.6 ± 0.20
	50–75 cm	8.9 ± 1.1	0.8 ± 0.1	4.41 ± 0.28	10.7 ± 0.46
	75–100 cm	7.2 ± 0.8	0.7 ± 0.1	3.82 ± 0.29	10.7 ± 0.73
	0–10 cm	68.4 ± 3.2	6.7 ± 0.3	6.77 ± 0.18	10.1 ± 0.11
	10–30 cm	20.6 ± 0.9	2.1 ± 0.1	6.5 ± 0.14	9.9 ± 0.25
20-years	30–50 cm	14.6 ± 779	1.3 ± 0.1	5.89 ± 0.15	11.3 ± 0.35
	50–75 cm	15.0 ± 1.0	1.2 ± 0.1	4.86 ± 0.28	12.7 ± 0.57
	75–100 cm	12.5 ± 1.0	1.0 ± 0.1	4.27 ± 0.06	13.1 ± 0.79
	0–10 cm	55.0 ± 2.8	5.6 ± 0.3	7.29 ± 0.14	9.7 ± 0.07
	10–30 cm	14.4 ± 1.7	1506 ± 0.2	5.59 ± 0.15	9.6 ± 0.09
30-years	30–50 cm	7.7 ± 0.5	0.7 ± 0.1	5.52 ± 0.13	10.6 ± 0.23
	50–75 cm	9.4 ± 0.7	0.9 ± 0.1	5.08 ± 0.18	10.6 ± 0.22
	75–100 cm	13.2 ± 1.5	1.3 ± 0.1	4.04 ± 0.24	10.1 ± 0.20

3.5 Density fractionation

In the 0–10 cm depth, there was marginally more FLF C ($p < 0.1$) and significantly more N stocks at the 30-year site compared to the unamended and 20-year site ($p < 0.05$, Fig. 5–4) with a slightly higher percentage of the bulk soil mass found in the FLF at the 30-year relative to the 20-year site ($p < 0.1$, Table S3). There was marginally more OLF C stocks across both compost sites compared to unamended soils ($p < 0.1$) and significantly more N stocks compared to unamended soils ($p < 0.05$, Table S4). For the heavy fraction, there was marginally more N stocks at 30-years compared to unamended ($p < 0.1$, Table S4) but there were no differences for the 20-year site.

There were no differences in the FLF, OLF, or HLF in the 30–50 cm among any of the C and N variables measured among treatments (Table S3, S4, S5).

The FLF at the 75–100 cm depth had a significantly higher N concentration compared to both the unamended and 20-year soils ($p < 0.05$, Table S3) but no difference in FLF N stocks or C:N. There was no difference among treatments in the OLF fraction at the 75–100 depth (Table S4), but there were many in the HF (Table S5). The C concentration of the HF was marginally higher in the 30-year vs unamended soils ($p < 0.1$) and significantly higher when all compost samples were compared with unamended ($p < 0.05$, Table S5). The N concentration of the HF was significantly higher in the 30-year vs unamended soils ($p < 0.05$) and when all compost samples were compared with unamended ($p < 0.05$, Table S5). This also led to marginally more N stocks in the 30-year site vs unamended soils ($p < 0.1$) though there was no significant difference in C stocks or C:N among treatments ($p > 0.05$, Table S5, Fig. 3A).

The proportion of C stored within each fraction had no significant differences by treatment within any depth ($p > 0.05$). When averaging the fractions in each depth across all treatments, the proportion of FLF C had a marginally significant increase at 0–10 cm compared to 30–50 cm ($p < 0.1$), and a significant increase when comparing 0–10 cm to 75–100 cm ($p < 0.05$, Fig. 3B). There was no significant change in the proportion of soil C within the OLF fraction by depth, but the proportion of C in the HF had a marginally significant decrease in the 0–10 cm compared to the 30–50 cm and 75–100 cm depths ($p < 0.1$, Fig. 3B).

3.6 Mineral Fraction Analysis: Radiocarbon and DRIFTS FTIR

The fraction modern of the mineral-associated HF did not vary by treatment at any depth (Fig. 4, $p > 0.05$). The fraction modern at 0–10 cm was significantly higher than the fraction modern at the 30–50 and 75–100 cm depths ($p < 0.05$).

To further assess changes to the mineral fraction, DRIFTS was used to assess the molecular composition of the OM within fractions (Within the mineral fraction, the simple plant OM to microbial OM ratio (simple:microbial) increased when assessed over all three depths when comparing the 20-year site to unamended soils ($p < 0.05$) and when comparing all compost sites to unamended soils ($p < 0.05$, Fig. 5, Table S6). No specific depth was significant on their own, however. The simple plant OM: complex plant OM had a marginally significant increase across all compost samples compared to unamended samples ($p < 0.1$, Table S7) when assessed across all depths, but no individual depth was statistically significant. These changes shifted the OM signature of compost samples closer to that of compost alone (Fig. 5). While there was no difference at any level for the complex plant OM: microbial ($p > 0.05$), there was for simple plant + complex plant OM: microbial OM across all depths as the 20-year and compost samples overall had a significantly higher simple plant + complex plant OM: microbial OM ($p < 0.05$), but there was no difference at the 30-year site ($p > 0.05$, Table S7).

Within the light fractions overall compost treatments had significantly lower complex plant OM: microbial OM in FLF at both the 30–50 and 75–100 cm ($p < 0.05$), but were not different for any other OM ratio. The plant OM: microbial OM was lower at the 30–50 cm aggregated across treatments at the 30–50 cm compared to 0–10 and 75–100 cm ($p < 0.05$, Figure S3). For the OLF we found that the 20-year site had significantly different values than the control and 30-year site at 30–50 cm for simple OM: microbial OM (Figure S4), all plant OM: microbial OM, and simple OM: complex plant OM ($p < 0.05$). Compost treatments had significantly lower complex plant OM: microbial OM than the control at 0–10 ($p < 0.05$), while 30 years alone was lower than control samples at 75–100 cm ($p < 0.05$). Compost treatments overall were marginally significantly higher at the 0–10 cm for the simple plant OM: complex plant OM ($p < 0.05$).

3.7 Microbial Biomass

There was no significant change in total MBC or MBN for either the 0–10 cm or 10–30 cm by years of compost additions or combined treatments relative to unamended samples ($p > 0.05$, Figure S5). Across all treatments, there was significantly more MBC and MBN in the 0–10 cm than the 10–30 cm ($p < 0.05$). There was a significant decrease from 5.9 to 5.1 in the MBC:MBN ratio from unamended to 20-year sites in the 0–10 cm ($p < 0.05$), with a marginally significant decrease when comparing all compost sites to unamended samples ($p < 0.1$, Figure S5). Across all treatments, the MBC:MBN ratio had a marginally significant increase of 5.7 to 6 from the 0–10 cm to the 10–30 cm depth ($p < 0.1$).

4. Discussion

4.1 Baseline Site Properties and Management Control Soil C in Unamended Sites

To have a point of comparison for long-term compost sites, a focus of this study was understanding the soil health variation within parks that have never received compost amendments. The variable with the largest prediction ability for soil carbon stocks at unamended sites was combined silt and clay content, which when aggregated across all depths explained approximately 19% of the variation in whole profile soil carbon stocks ($p < 0.05$), with only marginal explanatory power from pH (4%, $p > 0.05$). When assessing other properties impact on whole profile C stocks there was no impact from park age or other management factors ($p > 0.05$). No overall change in total C stocks by age runs contrary to current literature which finds increasing C stocks with turfgrass age (Phillips et al., 2022; Qian & Follett, 2012). However, these previous studies have focused on the top 30 cm of the soil while our unamended sites show the greatest variation in C stocks across sites below 50 cm (Figure S6).

While bulk C stocks had little impact from management practices before accounting for compost additions, we find that the turfgrass system itself has unique impacts on the proportion of soil C stored within each C fraction. Across a global average approximately 65% of SOC is found in the mineral fraction, with grasslands specifically averaging 70%, while in our study the average was 58% across treatments (Bastida et al., 2021; Sokol et al., 2022a; Sokol et al., 2022b). Our 30–50 cm and 75–100 cm samples were consistent with values from 61% to 71% across all treatment types and both depths. However, our top soil saw a lower proportion of C within the mineral fraction across all treatment types at only 43%, though this measurement had the highest within depth variation (Fig. 3B). Accounting for this difference in C proportion, we found the proportion of C as OLF in our study across all treatments and textures was 38% ± 9 in the top soil (0–10 cm), which is at the upper end of values compared to other managed systems, even among those that also had multiple years of OM amendments applied (10% – 25%) (Ryals et al., 2014;

Villa et al., 2024). This shift in the C proportions from mineral C to occluded fraction C in the top soil of turfgrass systems was independent of compost application ($p > 0.05$, Fig. 3B). This suggests that turfgrass soils may maintain a greater proportion of soil C in less protected C pools than other systems, potentially indicating a greater amount of inputs or turnover of organic matter compared to other systems (Cotrufo et al., 2013).

4.2 Long-Term Compost Additions Alters Surface C stabilization

Organic matter additions such as compost can impact many aspects of soil health, especially when applied annually for decades. SOC is an important soil health indicator, as it influences biological, physical, and chemical soil functions. In the topsoil, we found that long-term compost application increased soil organic C and N stocks in the 0–10 cm depths ($p < 0.05$). The largest total gains in stocks at any depth were within the 0–10 cm depth, which are expected as compost has been surface applied. Without compost, unamended parks averaged 5% C while the 20 and 30-year compost sites averaged over 6% and 7% soil C respectively, which led to significantly higher C stock values at our compost sites (Fig. 2). While there was sizable variation in C stocks the 10–75 cm depth range, there were no significant changes in soil C stocks with compost additions (Fig. 2). This may be due lower silt and clay content in this range compared to the 0–10 cm and 75–100 cm, leading to decreased ability to increase soil C (Fig. 1). Silt and clay content were consistent predictors for soil C stocks across all soil treatments from 10–100cm ($p < 0.1$), while compost is a strong predictor for soil C stocks in the surface soil (0–10 cm, $p < 0.05$) and deep soil (75–100 cm, $p < 0.05$). These results support a model where biological controls dominate SOC cycling in the surface horizons and mineral controls become increasingly important in subsoil layers (Pries et al., 2023).

Despite changes in soil C stocks in the topsoil, there were no significance differences in MBC and MBN contents ($p > 0.05$). However, the MBC:MBN did decrease with compost additions in the 0–10 cm ($p < 0.1$), particularly when comparing the 20-year site to unamended parks ($p < 0.05$, Figure S5). A decrease in MBC:MBN coupled with an observed increase in top 50 cm $\delta^{15}N$ at compost sites is consistent with enhanced microbial N cycling, which could lead to the preferential loss of ^{14}N overtime, suggesting that microbial activity is leading to enrichment of soil $\delta^{15}N$ as opposed to enrichment of $\delta^{15}N$ from direct compost inputs. This increase in N cycling comes despite no change in total MBC or MBN, suggesting there could be changes in microbial community composition. The decrease in MBC:MBN is consistent with an increase in carbon use efficiency (CUE) and decreased N limitation, which could promote increased microbial N cycling (Ros et al., 2006; Sinsabaugh et al., 2016). Increased N cycling and reduced N limitation could also increase microbial activity and turnover, potentially increasing necromass inputs to mineral-associated organic matter, which saw higher mineral C stocks at compost sites, though increases were concentrated at depth (75–100 cm; Fig. 3A) (Sokol & Bradford, 2019). While microbial community was not measured in this study, increases in mineral-associated organic matter and changes to the molecular makeup of the OM within density fractions study support the idea for altered microbial activity and turnover. Assessing both the stocks and molecular makeup of several soil C fractions is a valuable tool for understanding how different pools of C and N are cycling (Lavallee et al., 2020; Swanston et al., 2005). When we assess the mineral associated organic matter and other density fractions, we see that increases were not evenly distributed among the three fractions: the FLF, OLF, and HF, or by depth (Fig. 3A, 3B). We found that there was significantly more C and N stocks in the 0–10 cm FLF at the 30-year site compared to both our unamended and 20-year sites, driven by increased mass of FLF rather than changing C concentration (Fig. 3A, Table S3). A lack of FLF C stock change at the 20-year site or at other depths, could be explained if the FLF from compost application cycled into the other fractions. Turfgrass systems often have high microbial biomass and quickly cycling particulate organic matter pools, driven by turfgrass high allocation to the rhizosphere (Kaye et al., 2005; Thompson & Kao-Kniffin, 2019). While MBC did not vary by compost treatment, MBC was high across all parks, suggesting that these soils had a high capacity of microbially processing organic matter. At sites that received compost, this could lead to increases in the total amount of substrate transformed by

microbes, without increasing MBC or accumulating C within the FLF pool. In addition, compost could increase belowground plant productivity, indirectly providing additional C substrates that could transform into more stable C fractions (Junaidi et al., 2018). While we did not observe clear changes in the molecular makeup of the FLF fractions, we observed a significant increase in the 0–10 cm OLF C and N stocks at both compost sites compared to unamended soils (Table S4, S6). In addition, the compost treatments had lower complex plant OM: microbial OM ratios within the 0–10 cm ($p < 0.05$), suggesting there was both an increase in the topsoil OLF C stock, and in the OLF OM makeup. As OLF cycles on a decadal scale, changes in this fraction may be expected after several decades of applied compost OM break down and have opportunities to form and/or be enclosed by soil aggregates. This OLF pool can also be highly varied depending on how C is incorporated into aggregates, the rate of aggregation formation, and aggregate stability (Wagai et al., 2009). However, there were minimal differences in silt and clay content between unamended and compost sites in the topsoil (Fig. 1, $p > 0.05$), suggesting that differences in topsoil OLF are caused by the input of exogenous OM from compost and altered microbial activity.

When assessing the HF, we saw significantly more C stocks in the mineral pool at the 20-year site compared to unamended and 30-years at both the 0–10 cm and 30–50 cm depths (Fig. 3A). As the compost was surface-applied compost, we might expect large changes in the makeup of the topsoil mineral fraction pool, and the 20-year site may change more in the surface as it received more total compost than the 30-year site. However, we did see that the molecular properties of the mineral fraction changed across both compost sites in the top 50 cm, even when the total stock of mineral C did not. While no individual depth was significant on its own, we saw that the simple plant OM: microbial OM, simple plant OM: complex plant OM, and all plant OM: microbial OM ratios increased across all compost samples compared to unamended samples when pooled for the 0–50 cm depths ($p < 0.1$, Table S7). These values shifted to become more similar to the ratios found within finished compost compared to the ratios found in unamended soils. Thus, despite a lack of uniform change in HF C stocks in the topsoil, such as in the 0–10 cm at the 30-year site, the HF is still being altered through long-term compost application at both timescales. Mayer et al., 2022 also found decreases in the OM decomposition of the mineral fraction with long term organic matter amendments, but they also observed increases in topsoil mineral C content unlike our study (Mayer et al., 2022). Because our top soil mineral C and N stocks had little change at 30-years, despite observing the potential inclusion of compost materials within the mineral fraction DRIFTS OM signature change, there must be mechanism(s) leveling out the exogenous C inputs on a mineral C stocks basis (Fig. 5, S3, S4). Potential explanations could range from an increased turnover rate to the translocation of C into the mineral fraction facilitated by the compost addition. In support of the latter, changes in the C stock and the OM signature were present within the within 75–100 cm depth for the HF at the 30-year site and compost sites overall (Table S4, S6).

4.3 Effects of Long-Term Compost on Subsoils

Long-term compost additions led to dramatic changes in the deep soil (50 cm+), particularly in the 75–100 cm depth range. Bulk soil C and N stocks at 75–100 cm were significantly higher at the 30-year site and compost sites overall than unamended sites ($p < 0.05$), but not at the 20-year site compared to unamended soils ($p > 0.5$, Fig. 2). This change was not evenly distributed among the three fractions of soil C. The FLF had double the N concentration, but no C change, at the 30-year site compared to unamended and 20-year samples at 75–100 cm (Table S6, S3). This increase at depth could point to the translocation of some FLF from compost additions, but all other FLF factors at depth were unchanged. The N concentration could change alone if the compost had a low C:N but at an estimated 16:1, it did not. Another potential explanation is if there was sizable nitrate leaching. We lack data on the compost nitrate or overall N mineralization for confirmation, but the 30-year site has begun to receive recycled water irrigation while the 20-year site does not, which could explain the increase in N stocks at depth (Geisseler et al., 2021).

While slower cycling than the FLF pool, we also observed changes in the OLF pool at 75–100 cm. OLF stocks were increased when comparing both compost sites and the 30-year site alone to unamended soils (Fig. 3A). The OLF likely increased more than the FLF at depth as these soils have sizable silt and clay content within the 75–100 cm depth (Fig. 1), leading to a large capacity for aggregate protection if C where to reach the subsoil (Cotrufo et al., 2013). There was no change in the OLF from the 10–75 cm depth range, likely due to significantly lower levels of silt and clay content hampering the capacity for aggregate formation (Fig. 1).While the mineral fraction is typically described as the slowest-cycling and slowest-changing pool, there is recent evidence that short-term changes can impact the pool greatly (Jilling et al., 2025; Torn et al., 2013). This expectation of slow-cycling is exaggerated when assessed at depth, as deeper soil C pools are traditionally thought of as slower cycling (Pries et al., 2023). However, we found large changes in soil C and N concentrations and stocks in the mineral fraction, specifically in the 75–100 depth range (Fig. 3A, Table S4, S6). While compost additions ($p < 0.1$), pH ($p < 0.05$), and combined silt and clay were strong predictors at the profile scale ($p < 0.05$), relationships were depth dependent. Within depths, the only significant predictor of HF stocks was found within the 75–100 cm range and was compost application ($p < 0.05$), which was independent of silt and clay ($p > 0.05$) or pH impacts ($p > 0.05$). This is in contrast to surface C stocks (0–10 cm) where compost application did not significantly relate to mineral fraction C ($p > 0.05$).

Despite these depth-specific effects in the mineral fraction, bulk C stocks drive much of the variation in HF C ($p < 0.05$), as the proportion of C distributed between the three fractions did not change with compost additions (Fig. 3B). This indicates that increases in mineral C from compost additions are primarily reflecting proportional increases in total SOC, rather than shifts in the partitioning of carbon within fractions. While HF C stocks changed they were not accompanied by a change in the HF organic matter molecular signature by treatment within any one depth (Fig. 3A, Table S7). This is consistent with the idea that deep soil C stocks may be fueled both by direct compost C and primed pre-existing soil organic matter that is subsequently transformed or translocated downward, increasing deep soil C stocks while having minimal impact on the OM signature (Borken et al., 2004; Sun et al., 2019). This loss of C through translocation may explain the relatively sustained mineral stocks of C and N in the topsoil despite compost additions. This translocation with frequent irrigation could also explain why gains in the subsoil are greatest in the 30-year site, which received less cumulative compost than the 20-year site, but has 10 additional years of potential compost C translocation. However, we do not see a change in HF radiocarbon values in top soil that one may expect if the pool is continually receiving younger C via compost and new plant production while leaching other “native” C to depth (Fig. 4). To better understand this phenomenon, future studies should assess how DOC amount and DOC molecular makeup throughout the profile may change with long term compost additions. This would allow for better understanding of how compost-derived DOC, recently produced plant DOC, or native soil C that may be displaced and become DOC are contributing to changes in the mineral fraction makeup. In addition, running these DOC pool(s) and other fraction samples for radiocarbon could be used in conjunction to assess the age/source of DOC being moved and altered throughout the profile as radiocarbon in the mineral fraction alone was not definitive in this study for piecing together how the makeup of mineral fraction C changed.

Whether by DOC or another mechanism, compost is likely moving both itself and native soil C downward. This can help explain the relative lack of top soil mineral C increase, gain of C in the mineral fraction at depth, and why the deep C in composted sites has started to more closely resemble topsoil mineral C and/or compost properties (such as in bulk $\delta^{15}N$, Table 2). We also observed that while the mid-profile (10–75 cm) saw little change in overall soil C stocks or C fractions, the mid-profile contents evidence for the translocation of compost C in the bulk soil’s $\delta^{15}N$ content. We observed a significant increase in the soil $\delta^{15}N$ at both compost sites relative to unamended sites in the 0–50 cm ($p < 0.05$), with a marginally significant increase at 50–75 cm ($p < 0.1$, Table 2). This suggests that while compost is not altering C stocks in the mid-profile it is, directly or indirectly, translocating compounds into this

section of the soil. The change in $\delta^{15}N$ can be combined with our C data in the top (0–10 cm) and deep soil (75–100 cm) to further suggest that soil profile changes are due to compost additions, not from baseline soil differences that originated from before compost additions started (Choi et al., 2003; Lynch et al., 2006). Thus, studying both the top and subsoils are vital for understanding whole soil profile dynamics, as ending sampling at 50cm would have missed additional signs both of C movement (50–100 cm) and changes in C stocks (75–100 cm). Understanding these impacts of long-term organic matter additions is vital for both land managers and our climate.

4.4 Implications for Carbon Stabilization in High-Carbon Soils

In a high C baseline system of turfgrass, we still observed increases to one meter depth with 30-years of annual compost additions, even in deep mineral fraction. Similar to our results, Villa et al. (2024) found that 20-years of biosolids applications in irrigated pastures increased soil C throughout the whole profile, and specifically in the mineral fraction at the 75–100 cm depth. The majority of soil studies, especially those in urban systems which often have high baseline soil C, have focused on top soil C where a broader consensus exists for the increase in soil C that occurs with compost additions (Malone et al., 2023). Few urban soil studies have assessed soil C stocks to 1 m, and most of these are short-term, assessing soil responses to 5 years or less of OM additions. Contextualizing soil C dynamics presented in both the unamended and compost amended sites is important for understanding how C rich soils respond to long-term exogenous C additions. Pouyat et al. (2009) measured SOC stocks without OM additions to 1 m in urban turfgrass systems in Baltimore, MD and extrapolated SOC stocks to 1 m in Denver, CO (soil sampling ended at 30 cm in Denver, CO). They estimated turfgrass soil C stocks to be 110 and 127 Mg C ha^{-1} to 1 m respectively, while our unamended sites averaged 239 Mg C ha^{-1}. When assessing C distribution throughout the soil, we found that the proportion of C below 20 cm at our sites were significantly more than estimated in the aforementioned paper. We estimate that across all sites and treatments approximately 65% of soil C stocks are below 20 cm (53% below 30 cm) in our study compared to 53.5% below 20 cm in Pouyat et al. (2009). This could indicate a large degree of C translocation occurs in irrigated systems with high C, and that deep soil C is driving some of the difference between study estimates of soil C stocks, independent of differences in quality or management of the system (Pouyat et al., 2009).

In addition to observing an increased proportion of soil C in deep layers, we observed that in the mineral fraction, we observed $\Delta^{14}C$ values across all treatments in our high C soils were divergent from global averages suggesting management is strongly impacting soil C cycling (Fig. 4). In our samples were found $\Delta^{14}C$ values in the mineral fraction of -63 ± 16‰, -186 ± 14‰, and − 247 ± 28‰ in our 0–10 cm, 30–50 cm, and 75–100 cm depths respectively while an analysis of the internation soil radiocarbon database by Li et al. (G. Li et al., 2023) found that average values over 295 diverse soil profiles were − 3 ± 5‰, − 168.03 ± 14.91‰ and − 358 ± 29‰ for the 0–30 cm, 30–60 cm, and 60–100 cm depths respectively. Our measured values had significantly more negative $\Delta^{14}C$ in the surface soil and significantly increased $\Delta^{14}C$ values in the subsoil (75–100 cm) compared to the global assessment. This suggests that our high C soils may have older top soil C and younger C at depth relative to most other ecosystems (Heckman et al., 2022; Torn et al., 1997). Through the use of irrigation, and other management strategies aiming to maximize turfgrass growth, the priming or loss of “younger” HF C occurs in the top soil. Irrigation may then facilitate the translocation of this C to depth, leading to both older top soil C and younger subsoil HF C – a phenomenon that has been observed in other ecosystems with high root inputs (Schiedung et al., 2023). As we saw no difference in $\Delta^{14}C$ values across treatments in the mineral fraction at any depth, despite changes in mineral fraction C stocks across treatments, this phenomenon appears to be driven by shared management practices. While deep soil is often thought of as slow changing, all sites regardless of age (ranging from 17 to 61 years) followed this trend,

highlighting how quickly intense management strategies can alter C translocation and stabilization even to one meter (McClelland et al., 2022).

5. Conclusions

We assessed the impacts on C cycling from long-term organic matter additions to urban soils, where we observed large increases in bulk soil profile C stocks, stabilization of carbon within the mineral fraction at depth, and changes in the molecular composition of mineral-associated organic matter. Soil carbon impacts were concentrated in the topsoil and at the bottom of the soil profile, highlighting the capacity for surface-applied compost to impact deep soil C on a decadal timescale. Soil C gains were largest in the mineral fraction, particularly at depth, highlighting the ability for fast incorporation of C into the mineral fraction. Within the mineral fraction, we saw a shift from microbial to simple plant OM, potentially from the direct incorporation of compost C into the mineral fraction at depth. Despite high levels of initial C managed turfgrass ecosystems, decades of compost application significantly increased and altered the makeup of soil C in urban soils. These findings suggest that long-term compost additions have the ability to increase whole profile soil carbon stocks.

Declarations

CRediT Authorship Contributions Statement

Zachary Malone: Conceptualization, Writing – original draft, Writing – review & editing, Formal analysis, Investigation, Data curation, Visualization, Software, Methodology. **Samuel Lee**: Writing – review & editing, Investigation, Methodology. **Karis McFarlane:** Investigation, Methodology, Resources, Supervision, Writing – review & editing. **Rebecca Ryals**: Conceptualization, Writing – review & editing, Supervision, Funding acquisition, Methodology, Project administration.

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Conflict of Interest Statement

The authors declare no conflicts of interest.

Data Availability Statement

Data and code available are available forthcoming on Dryad using the following doi: https://doi.org/10.5061/dryad.t4b8gtjjh. Prior to publication, the data and code are available at this link: http://datadryad.org/share/LINK_NOT_FOR_PUBLICATION/mmINRm_lR6y25TZ1BOhPqkCzGuGFeh4iV7359NzbFHw.

Supporting Information

Additional supporting information and figures can be found online in the supporting information section of this article.

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Figures

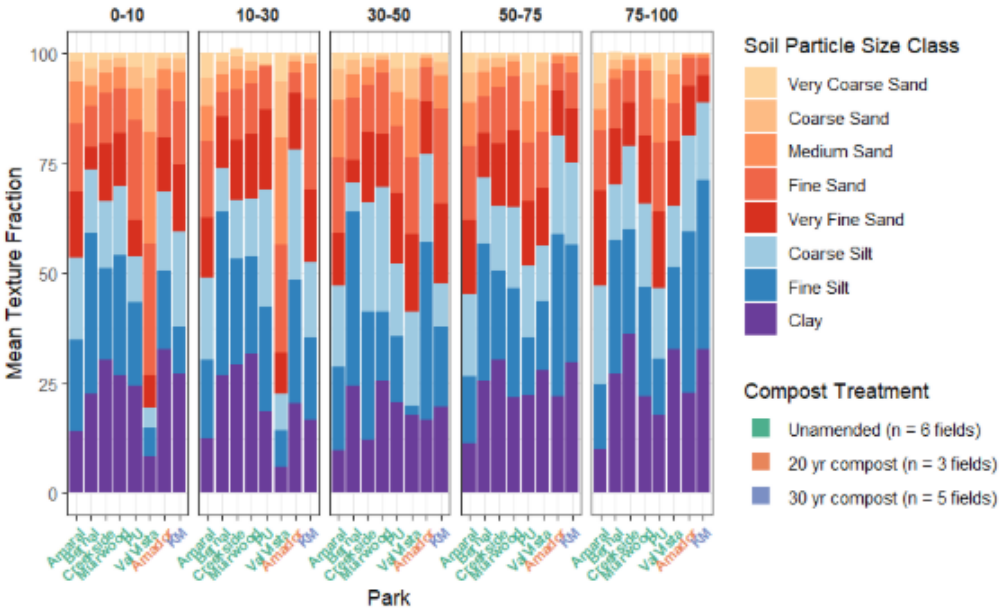


Figure 1

Soil particle size class distribution by park, compost treatment, and depth. Color on the x-axis tick labels correspond to green as unamended, red as 20-years of compost, and blue as 30-years of compost additions. Color of soil particle sizes have reds corresponding to sand classes, blues as silts, and purple as clay.

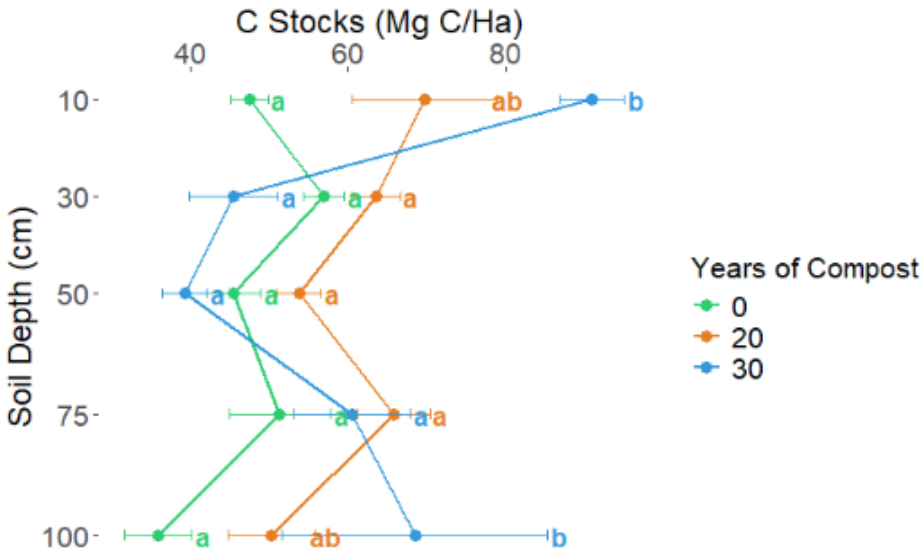


Figure 2

Soil C stock (Mg C ha^{-1}) by years of compost application and depth. Letters indicate Tukey HSD significances while error bars represent SE.

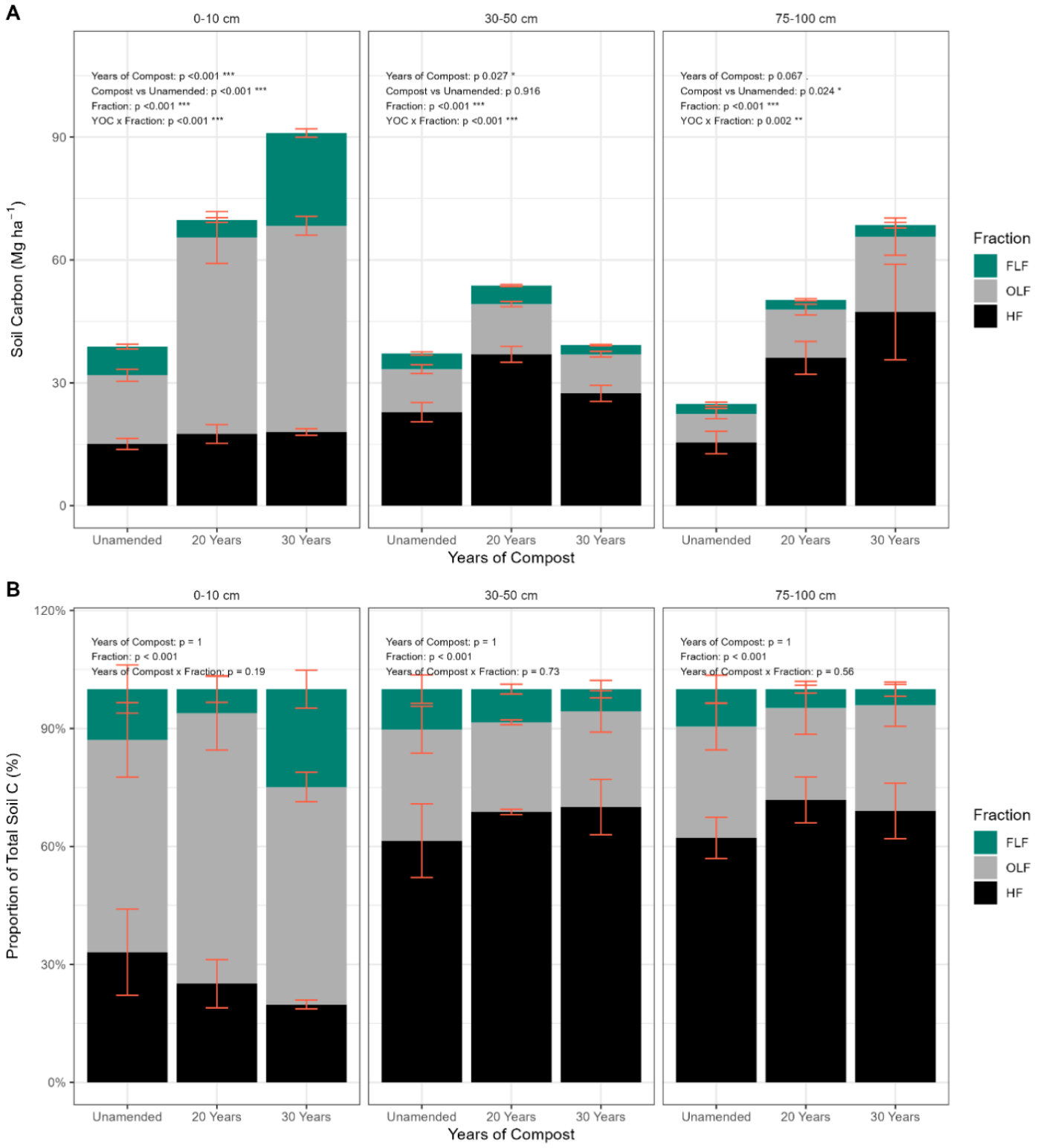


Figure 3

A) Soil C stocks (Mg C ha^{-1}) by years of compost, depth, and density fraction. YOC, FLF, OLF, and HF represent years of compost, free light fraction, occluded light fraction, and heavy fraction respectively. B) Proportion of density fractions within the FLF, OLF, and HF by depth and years of compost.

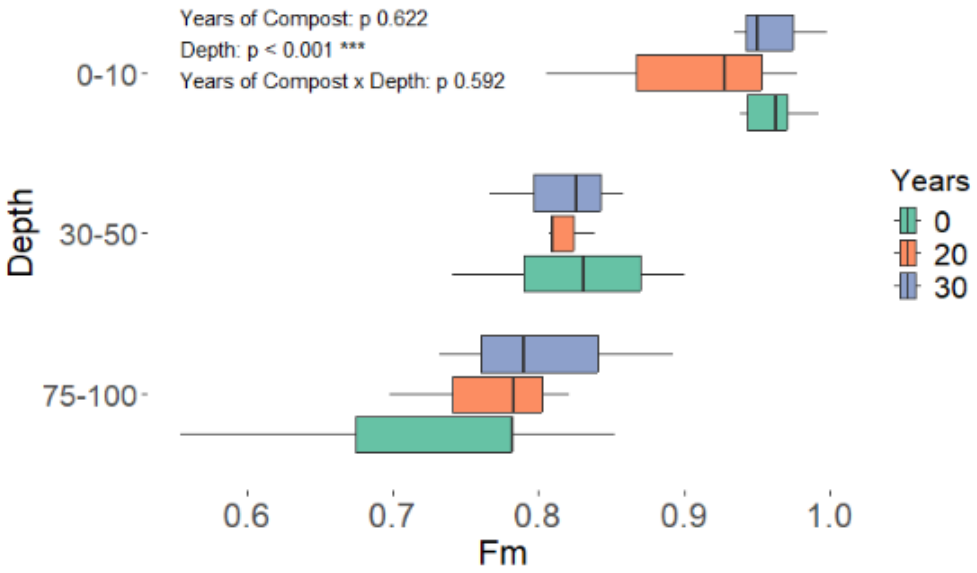


Figure 4

Soil fraction modern by depth and years of compost application.

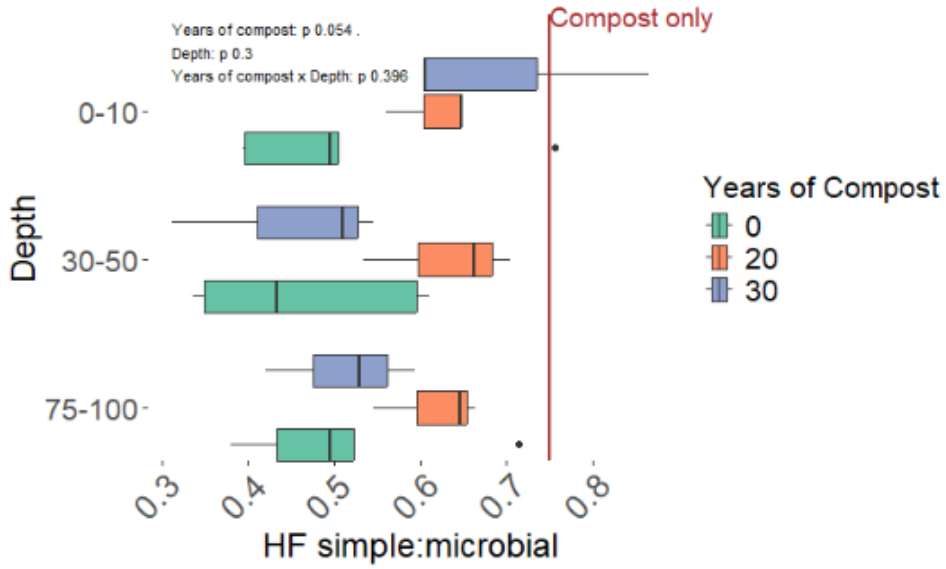


Figure 5

DRIFTS FTIR Simple OM to Microbial OM ratio between treatments and depths for the mineral C fraction. The brown line represents the value of compost (n=3) by itself. P-values represent statistical significance of Tukey's HSD posthoc test between unamended (n=5), 20-years of compost (n=3), and 30-years of compost (n=3) treatments. Dots represent outliers in the respective treatment.

Supplementary Files

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