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Positive long-term impacts of restoration on soils in an experimental urban forest

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Abstract. As urbanization increases worldwide, investments in nature-based solutions that aim to mitigate urban stressors and counter the impacts of global climate change are also on the rise. Tree planting on degraded urban lands—or afforestation—is one form of nature-based solution that has been increasingly implemented in cities around the world. The benefits of afforestation are, however, contingent on the capacity of soils to support the growth of planted trees, which poses a challenge in some urban settings where unfavorable soil conditions limit tree performance. Soil-focused site treatments could help urban areas overcome impediments to afforestation, yet few studies have examined the long-term (>5 yr) effects of site treatments on soils and other management objectives. We analyzed the impacts of compost amendments, interplanting with shrubs, and tree species composition (six species vs. two species) on soil conditions and associated tree growth in 54 experimental afforestation plots in New York City, USA. We compared baseline soil conditions to conditions after 6 yr and examined changes in the treatment effects from 1 to 6 yr. Site treatments and tree planting increased soil microbial biomass, water holding capacity, and total carbon and nitrogen, and reduced soil pH and bulk density relative to baseline conditions. These changes were most pronounced in compost-amended plots, and the effects of the shrub and species composition treatments were minimal. In fact, compost was key to sustaining long-term changes in soil carbon stocks, which increased by 17% in compost-amended plots but declined in unamended plots. Plots amended with compost also had 59% more nitrogen than unamended plots, which was associated with a 20% increase in the basal area of planted trees. Improvements in soil conditions after 6 yr departed from the initial trends observed after 1 yr, highlighting the importance of longer-term studies to quantify restoration success. Altogether, our results show that site treatments and tree planting can have long-lasting impacts on soil conditions and that these changes can support multiple urban land management objectives.

Key words: *afforestation; carbon; compost; nature-based solutions; nitrogen; plant–soil interactions; soil carbon stocks; soil health; urban forestry.*

Introduction

Land-use change from urban and agricultural expansion is the leading driver of terrestrial biodiversity decline and loss of ecosystem services (Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services [IPBES] 2019). Seventy-five percent of the global land surface has been significantly altered by human use (IPBES 2019), and projections of urban expansion predict continued and sustained growth at the expense of forests and other natural areas (Seto et al. 2012, Güneralp and Seto 2013). The expansion of urban areas has been linked to significant losses in habitat, biodiversity, and carbon (C) storage (McKinney 2002, Seto et al. 2012, IPBES 2019), prompting investments in nature-based solutions (i.e., conservation or restoration practices that support ecosystem services, such as C storage and greenhouse gas emission reductions) to offset these negative impacts (Pataki et al. 2011, Griscom et al. 2017).

Globally, reforestation efforts—including afforestation of nonforested land—are suggested to offer one of the largest opportunities for low-cost, nature-based climate mitigation (Griscom et al. 2017, Doelman et al. 2020). As such, large-scale tree-planting programs have become increasingly common in cities around the world (Pincetl et al. 2013). Once established, urban forests

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provide a multitude of social and ecological services and functions, such as C storage and sequestration (Nowak and Crane 2002, Pataki et al. 2011); habitat for native wildlife (MacKay et al. 2011); stormwater mitigation (Xiao et al. 1998, Kuehler et al. 2017, Phillips et al. 2019); local cooling (Loughner et al. 2012); and a host of social, health, and aesthetic benefits (Tyrväinen et al. 2005, McPhearson 2011, Elmqvist et al. 2015). However, the capacity of afforestation projects to fulfill these management goals hinges on whether planted trees ultimately mature into forests that can perform a broad range of ecological functions.

Although trees are the primary focus of afforestation projects, soils are also critical to achieving restoration goals. In addition to supporting the growth and development of planted trees (Hawver and Bassuk 2007), healthy soils can help mitigate urban stressors by improving infiltration of excess stormwater (Phillips et al. 2019), retaining nutrients from runoff (Morel et al. 2015), sequestering and storing C in belowground pools (Pouyat et al. 2002), and promoting native understory plant recruitment over invasive species (Pavao-Zuckerman 2008). However, urban soils can also be comprised of human-transported materials (e.g., topsoil relocation) and/or anthropogenic fill (Effland and Pouyat 1997), which are characterized by high levels of compaction, unfavorable pH values (e.g., extremely acidic in former industrial sites or highly basic from the weathering of impervious surfaces), and insufficient nutrients to support tree growth and forest development (Capra et al. 2015). Restoration site treatments specifically designed to ameliorate soil conditions may therefore be required to support urban land management objectives at sites with anthropogenic soils.

In some circumstances, afforestation alone may be sufficient for improving soil conditions at degraded sites. For instance, afforestation has been shown to lead to a significant increase in soil organic C stocks—one of the most commonly used indicators of soil health (Bünemann et al. 2018)—in highly managed systems, such as former agricultural lands (Laganière et al. 2010), mines (Giménez et al. 2002), landfills (Schwendenmann and Mitchell 2014), and wastelands (Shi and Cui 2010). Likewise, in urban areas, afforestation can reduce compaction and build soil organic matter pools (Cao et al. 2007). Yet the potential benefits of urban afforestation are only attainable if planted trees overcome initial impediments to their survival and growth. This poses a challenge in some urban settings. where inhospitable soil conditions severely limit tree performance (Jim 1993, Jim 1998*b*), including for native species (Pregitzer et al. 2016).

In recognition of these barriers to the success of urban afforestation projects, there is a growing body of research that seeks to understand how site treatments can be tailored to generate urban soil conditions that better support tree growth and other management objectives (e.g., Cogger 2005, Layman 2010, Oldfield et al. 2014). For instance, tillage practices can initially improve water holding capacity by decreasing compaction (Oldfield et al. 2014), which, in turn, can increase the root volume of planted trees (Layman 2010). However, heavy tillage is also known to reduce soil C (Six et al. 2000, Haddaway et al. 2017), which poses a potential trade-off with respect to urban land management objectives. Mulching, ground covers (Wang et al. 2017), and nurse shrubs (Gómez-Aparicio et al. 2005) can also improve soil characteristics by regulating soil temperature, increasing soil moisture, and providing a source of C and nutrients to support microbial activity and plant growth. The addition of compost in the form of green waste or biosolids also has well-documented positive impacts on soil conditions (Chen et al. 2013, 2014), and, consequently, on the growth rates of planted trees over both short (Sullivan et al. 2009, Guerrini et al. 2017) and long (Layman et al. 2016) periods of time. Altogether, these findings suggest that soil-focused site treatments could help promote forest development. However, few studies have empirically tested the effects of different site treatments on soil conditions and tree growth over longer time frames (>5 yr; Kuebbing et al. 2018) and within the context of multiple management objectives. As planted trees take several years to respond to soil conditions (Brunner et al. 2015), and soil C stocks take years to accumulate postplanting (Li et al. 2012, Nave et al. 2013), evaluating longer-term changes in soil conditions and tree growth is essential for documenting the value of investments in tree-planting projects and ensuring that this value translates into management practices.

Here, we help to fill this knowledge gap by assessing changes in soil physical, chemical, and biological properties and processes 6 yr after the implementation of site treatments and tree planting at a large-scale afforestation project in New York City, USA. We evaluated soil responses to three commonly used site treatments: compost amendments, interplanting with shrubs, and the species composition of planted trees. Specifically, we ask (Q1) To what extent do the different site treatments alter soil conditions and belowground C stocks after 6 yr relative to baseline conditions? (Q2) How do the effects of the site treatments on soils change over time? (Q3) Are changes in soil conditions resulting from site treatments associated with the growth of the planted trees? By evaluating the effects of site treatments and tree planting on soil conditions, along with the capacity of altered soils to fulfill specific management objectives, we aim to inform the design and implementation of future urban afforestation efforts.

Methods

Site description and experimental design

Our study was conducted within long-term research plots located in Kissena Corridor Park, a 40-ha urban park in Queens, New York, USA (40°44′ N, 73°49′ W;

Fig. 1a). Kissena Corridor Park was planted as part of Million Trees NYC, a citywide initiative established in 2007 to increase New York City's canopy cover (PlaNYC Reforestation Overview 2015). Over the course of 8 yr, Million Trees NYC planted one million trees throughout the city's natural areas and street tree pits (PlaNYC Reforestation Overview 2015). To track the long-term effects of Million Trees NYC, 54 experimental plots—collectively known as the New York City Afforestation Project (NY-CAP)—were installed in Kissena Corridor Park in 2010 (Fig. 1a; Felson et al. 2013). The plots were intentionally established within a heavily trafficked urban park to facilitate both research and public use (Felson et al. 2013). The experimental site has a moist temperate climate with temperature highs and lows in July (mean 24.9°C) and January (mean 0.2°C), respectively, and a mean annual precipitation of 109 cm (National Oceanic and Atmospheric Administration [NOAA] 2016). The soils are classified as Laguardia-Ebbets complex, which is characterized by coarse, well-drained sandy loam with more than 10% human-transported material (Natural Resources Conservation Service [NRCS] 2016). Parent material at Kissena Corridor Park is comprised of human-transported material, which distinguishes the soils at our site from other urban soils of glacial origin (e.g., gneiss, granite, or schist) in this region.

The NY-CAP experimental plots test the impacts of three common site treatments (hereafter referred to as "experimental treatments") on planted tree growth (Oldfield et al. 2015) and soil conditions (Oldfield et al. 2014) over time: tree species composition (two species or six species), compost addition (compost amended or unamended), and interplanting with shrubs (presence or absence; Felson et al. 2013). All 54 plots were randomly planted with either two or six tree species (i.e., the species composition treatment) in pairs of two plots, and each pair was treated with or without compost in a crossed arrangement (Fig. 1). Within each pair, one plot was interplanted with shrubs and one contained no shrubs (Fig. 1). Replication of the eight treatment combinations is uneven and is as follows: two species/+shrub/+compost (n = 5), two species/+shrub/−compost (n = 9), two species/−shrub/+compost (n = 5), two species/−shrub/−compost (n = 9), six species/+shrub/+compost (n = 8), six species/+shrub/−compost (n = 5), six species/−shrub/+compost (n = 8), and six species/−shrub/−compost (n = 5).

FIG. 1. v(a) Map of the 54 experimental plot locations and their compost treatments (amended or unamended) in Kissena Corridor Park, Queens, New York City, USA and (b) the plot layout for the species composition (two and six species) and shrub (with or without shrubs) treatments. All 54 plots were planted with either two or six tree species, and the species composition treatment was crossed with the shrub and soil-amendment treatments. Sampling occurred in the interior plot, which was surrounded by a 2.5-m buffer of planted trees to minimize the influence of edge effects. Figure modified from Felson et al. (2013).

Prior to planting and plot installation, the site was dominated by invasive herbaceous species, such as *Artemisia vulgaris* L. (common mugwort) and native species, such as *Solidago canadensis* L. (Canada goldenrod; Oldfield et al. 2014). There were also a few small stands of *Robinia pseudoacacia* L. (black locust), *Rhus typhina* L. (staghorn sumac), *Prunus serotina* Ehrh. (black cherry), and a few individual *Acer saccharinum* L. (silver maple) located on the east side of the park but outside of the afforestation plots (see Fig. 1a). In 2009 all existing vegetation was removed from the planting area and plots were rototilled to prepare the site for planting (hereafter referred to as "site preparation"). This included the removal of brush, vines, stumps, roots, stones, human artifacts (e.g., concrete, scrap metal), and some trees ≤15 cm in diameter at 1.37 m. At this time, compost was incorporated into the compost-amended plots (n = 26) to a depth of 15 cm at a rate of 2.5 m^3 per 100 m^2 using a rototiller. The compost consisted of a commercial blend of nutrient-rich biosolids and clean, ground wood chips and had a pH of 6.3, a bulk density of 0.457 g/cm^3, and elemental concentrations of 60% C, 3.2% N, 3.7% P, and 0.44% K on a dry-weight basis (Oldfield et al. 2014). In 2010, all 54 plots also received a 5-cm surface layer of mulch composed of shredded hardwood trees to minimize drought stress for the planted trees. The pH of the mulch was between 5.8 and 7, but the nutrient concentrations are unknown.

Native trees, shrubs, and forbs were planted in the NY-CAP plots in October 2010 following site preparation. All 54 15 × 15 m plots were planted with 56 3–5-yr-old container-grown (7.58 L) saplings that were 0.6–1.2 m in height. Trees were planted at a spacing of 2.1 m in an offset grid designed to integrate the research plots into the public landscape (Fig. 1; Felson et al. 2013). The plots were either planted with two or six species for the species composition treatments. Two-species plots (n = 28) included 28 *Tilia americana* L. (American basswood) and 28 *Quercus rubra* L. (northern red oak; Fig. 1b). Six-species plots (n = 26) included 8 *T. americana*, 8 *Q. rubra*, 10 *Celtis occidentalis* L. (hackberry), 10 *P. serotina*, 10 *Quercus alba* L. (white oak), and 10 *Carya* Nutt. species (hickory). The two sides of the park were planted with different but functionally similar *Carya* species owing to limited nursery availability; *C. glabra* (P. Mill.) Sweet (pignut hickory) was planted on the west side and *C. laciniosa* (Michx. F.) G. Don (shellbark hickory) was planted on the east side (Fig. 1). Half the plots (n = 27) were also underplanted with an assemblage of native shrubs (41 individuals/plot) and forbs (672 individuals/plot) in addition to the trees (Fig. 1b). We refer to this as the shrub treatment since the forb plantings suffered from high mortality 1-yr postplanting and are therefore unlikely to affect long-term soil conditions. Shrub species included *Sambucus nigra* L. ssp. *canadensis* (L.) R. Bolli (American black elderberry), *Hamamelis virginiana* L. (American witch hazel), *Lindera benzoin* (L.) Blume (northern spicebush), *Cornus racemosa* Lam. (gray dogwood), and *Viburnum dentatum* L. (southern arrowwood; Fig. 1b). See Felson et al. (2013) for additional information on the design and setup of NY-CAP and Oldfield et al. (2014) for the sourcing and selection of native plant materials.

Soil sampling

We sampled soils four times over the course of the experiment to capture baseline conditions in October 2009 (prior to site preparation and planting), conditions postsite preparation (prior to planting) in October 2010, and conditions 1 and 6 yr postplanting in October 2011 and 2016, respectively. Our study supplements the results from the first 3 yr of the project (i.e., 2009–2011; Oldfield et al. 2014) with data collected 6 yr postplanting in 2016 to assess whether the effects of the experimental treatments on soils were sustained over longer time frames.

For each sampling occasion we collected and pooled five 8-cm-diameter soil cores from the four corners and center of a smaller 10 × 10 m area nested within the 15 × 15 m research plots to minimize the influence of edge effects (Fig. 1b). We intentionally sampled from the same locations each year (~50 cm from the base of the planted trees) to capture the effects of the growing trees (e.g., root expansion, leaf litter inputs) on soils over time. In 2009, 2010, and 2016, we sampled soils at two depths (~0–8 and 8–16 cm) and recorded the exact depth to calculate C stocks to a fixed soil mass because we anticipated changes in bulk density owing to site preparation practices such as tilling (i.e., the cumulative mass coordinate approach in Gifford and Roderick 2003). We had planned to collect soils to a cumulative depth of 30 cm but were unable to sample below 16 cm owing to extreme subsurface compaction. In 2011, we only collected 8-cm-deep soil samples because we did not measure soil C stocks that year. We pooled all samples by depth (i.e., 0–8 cm or 8–16 cm) and plot to minimize the influence of fine scale (i.e., within-plot) heterogeneity on the laboratory measurements. This sampling scheme yielded 54 surface (0–8 cm) and 54 subsurface (8–16 cm) samples per year (with the exception of 2011) for a total of 378 samples. We kept soils in coolers following collection and returned them to the lab the same day, where they were stored as field-moist samples in sealed plastic bags at 4°C for up to 8 weeks prior to analysis.

Planted tree measurements

In August 2011, approximately 10 months after planting, we took measurements for 24 trees planted in each of our experimental plots (1,296 trees total). These 24 trees were located in the center of the full 15 × 15 m plot, allowing for a 2.5-m buffer around plot edges (Fig. 1b). Initially, the planted trees ranged from 0.6 to 1.2 m in height, so they were too small to measure diameter at breast height (DBH, 1.37 m). Instead, we measured

diameter of root collar (DRC) at soil level using calipers (Oldfield et al. 2015), which can be used in lieu of DBH measurements for small-statured or multistemmed trees (Chojnacky and Rogers 1999, Magarik et al. 2020). We remeasured these same planted trees in 2016. Six years postplanting the trees were large enough to have a DBH, so we measured DBH instead of DRC and used the DBH measurements to calculate basal area (m^2/ha). For trees with multiple stems, we used the DBH of the largest stem for basal area calculations.

Laboratory analyses

For the 54 surface soil samples (i.e., 0–8 cm) collected each year, we measured six soil variables indicative of particular ecological functions in temperate urban forests in our study region: bulk density, soil pH, water holding capacity, active microbial biomass, and total C and nitrogen (N) concentrations. For the 2016 surface samples, we also measured soil texture. For the 54 subsurface samples (i.e., 8–16 cm) collected in 2009, 2010, and 2016, we repeated bulk density and total C concentration measurements to calculate belowground C stocks on a per mass basis (Gifford and Roderick 2003).

We weighed and homogenized each sample and air-dried a representative subsample of nonsieved soil (~500 g) to determine the mass and volume of roots and stones greater than 2 mm. We calculated bulk density based on soil dry mass and core volume (i.e., g soil/cm^3) and adjusted our calculations for the mass and volume of roots and stones (Soil Survey Staff 2014). Prior to the chemical analyses, we passed each field-moist sample through a 4-mm sieve. To determine pH, we mixed the soils and water in a 1:1 volumetric ratio and measured the pH of the supernatant after 10 min using a benchtop meter (VWR sympHony Sb70p; Allen 1989). We estimated water-holding capacity by saturating each sample and allowing it to drain freely for 2 h then weighing, drying at 105°C, and reweighing each sample (Paul et al. 2001). For microbial biomass, we used a modified substrate-induced respiration method that measures rates of CO_2 efflux over a 4-h incubation period using an Infra-Red Gas Analyzer (IRGA, Li-COR model Li-7000, West and Sparling 1986, Bradford et al. 2008). This method provides an estimate of active microbial biomass (Wardle and Ghani 1995). We used a ball mill to grind air-dried soil samples to a fine powder and analyzed total C and N concentrations using a Costech ESC 4010 Elemental Analyzer (Costech Analytical Technologies Inc., Valencia, California, USA). We calculated C stocks using a mass-dependent rather than depth-dependent approach because we expected bulk density to decrease following site preparation and the addition of trees (Gifford and Roderick 2003, Wendt and Hauser 2013). Linear interpolation of soil C estimates from cores collected at two depths was used to calculate C stocks for a standard dry soil mass. Finally, we analyzed the percent sand, silt, and clay-sized particles using the hydrometer method. For each plot, we calculated the ratio of coarse to fine soil particles for use in our analysis by dividing percent sand by the sum of percent clay and silt.

Statistical analyses

We used a series of linear mixed effect models (LMMs) to address our three research questions. For all analyses except soil C stocks, we used surface soil data (0–8 cm). For all LMMs, we standardized and centered continuous predictors by subtracting the mean and dividing by two standard deviations so that continuous and binary variables both had means of 0 and standard deviations of 0.5 (Gelman 2008). We observed that the east side of the park (n = 26) was located closer to adjacent forest stands and had less foot traffic than the west side (n = 28; Fig. 1), so we included side as a random effect in all of the LMMs to account for these differences (Doroski et al. 2018). We calculated variance explained by fixed effects (marginal R^2) and by fixed and random effects (conditional R^2) using the method described by Nakagawa and Schielzeth (2013). We normalized the 95% confidence intervals of the means to account for repeated measures of the same plot (Cousineau 2005, Morey 2008). We considered coefficients with $P < 0.05$ to be significant and coefficients with $P < 0.10$ to be marginally significant. We fit all models using a Gaussian error distribution and ran analyses with the "lme4" package in R (version 3.4.2; R Core Team 2020, Bates et al. 2015).

(Q1) Long-term effects of the experimental treatments on soil conditions (2009 versus 2016).—We used a series of LMMs to assess the impacts of the three experimental restoration treatments on pH, microbial biomass, %C, % N, water holding capacity, bulk density, and soil C stocks. Fixed effects included year, the three experimental treatments (i.e., presence or absence of compost, presence or absence of shrubs, and two or six planted tree species), the soil texture index (i.e., ratio of coarse to fine soil particles), and the two-way interaction between year and each of the treatments. For the pH models, we included a second-order term for soil texture because we observed a nonlinear, unimodal relationship. In these models, the fixed effect year compares baseline conditions from 2009 to conditions 6 yr postplanting in 2016 and therefore signifies the restoration effect. Interactions between year and the experimental treatments therefore indicate that the restoration effect was dependent on that particular treatment. All response variables met assumptions of normality and homoscedasticity. We initially analyzed pH data as hydrogen ion concentration (i.e., the inverse log of pH; [H^+]), because it is on a linear rather than logarithmic scale. However, because we needed to natural log-transform [H^+] to meet model assumptions, we ultimately chose to analyze the data as pH because it is more readily interpreted by land managers. We report all relative changes in soil acidity as

[H^+]. We included plot and plot nested within side (east or west) as random effects to account for repeated measurements in the same plot in different years.

Based on model outputs, we ran a post hoc analysis to further our investigation of the strong year by compost interaction observed for all soil response variables with the exception of bulk density (Appendix S1: Table S1). The post hoc models assessed the restoration effect from 2009 to 2016 (as indicated by year) in the subset of plots with (n = 26) and without (n = 28) compost amendments. Fixed effects included year (i.e., restored data from 2016 or unrestored baseline data from 2009) and the soil texture index (i.e., ratio of coarse to fine soil particles). Again, we included a second-order term for soil texture in the pH models. Random effects included plot and plot nested within side.

(Q2) Changes in the experimental treatment effects over time (2011–2016).—Using the same soil response variables described for Q1 (with the exception of soil C stocks which we did not collect in 2011), we created a series of LMMs with year (i.e., 2011 or 2016), the three experimental treatments (presence or absence of compost, presence or absence of shrubs, and two or six planted tree species), the soil texture index, and the two-way interaction between year and each of the treatments as fixed effects. For these models, the fixed effect year compares conditions 1-yr postplanting in 2011 to conditions 6 yr postplanting in 2016 and therefore signifies changes in the restoration effects over time. Again, we included plot and plot nested within side as random effects.

(Q3) Association between soil conditions and tree growth.—We created a series of LMMs using the 2016 surface soil data and tree basal area data from 2011 and 2016. For these models, we limited our analyses to the two species (*Q. rubra* and *T. americana*) present in both the two- and six-species plots because we would otherwise be unable to make cross-treatment comparisons. Over the course of the experiment, 137 *Q. rubra* and 95 *T. americana* either died, were reported missing, or resprouted, and detailed field notes suggested that these instances were primarily the result of extreme herbivory or vandalism (e.g., fire; see Oldfield et al. 2015). Because our objective was to examine the potential influence of soil conditions on tree growth, we removed dead, missing, or resprouted individuals from our tree-growth data set, because these factors (e.g., vandalism) were beyond the scope of this study. The final data set included 299 *Q. rubra* and 345 *T. americana*.

We built our tree growth LMMs with 2011 DRC (hereafter referred to as "initial DRC") as a predictor variable and 2016 tree basal area as the response. This model structure accounted for the fact that trees with larger DRC measurements at the time of planting would also be expected to have larger DBH values in 2016 independent of differences in soil conditions. To select the soil variables included as predictors in the LMMs, we created a correlation matrix with all 2016 surface (0–8 cm) soil variables to ensure that we would not violate the assumptions of no multicollinearity (Appendix S1: Fig. S1). Through this process, we selected [H^+], %N, and microbial biomass, all of which had correlation coefficients <0.50. Because we observed a nonlinear, unimodal relationship between microbial biomass and basal area, we added a second-order microbial biomass term to improve the model fit. We used this model structure to evaluate the relationships between soil conditions and the basal area of *Q. rubra* and *T. americana* combined, as well as *Q. rubra* and *T. americana* alone, because we expected the two species to have different growth rates. Based on strong effects of the compost treatments (as opposed to the shrub and composition treatments) on soil conditions (Appendix S1: Table S1), we further investigated the impacts of compost on tree growth by creating additional LMMs (with the same model structure as the full tree growth model) using data from amended plots only (n = 26) and unamended plots only (n = 28). This approach allowed us to compare the relative effects of soil variables on tree growth in amended and unamended plots without violating model assumptions (e.g., multicollinearity between compost treatments and soil variables). Variance inflation factors (VIF) <1.7 suggested that multicollinearity was sufficiently low among these predictor variables.

Results

(Q1) Long-term effects of the experimental treatments on soil conditions (2009 versus 2016)

Overall, we found significant differences in all measured soil variables 6 yr postplanting relative to baseline conditions. These changes included increased mean %N, microbial biomass, water holding capacity, and %C and reduced soil pH and bulk density (Fig. 2; Appendix S1: Tables S1, S2). The positive effect across all 54 plots was largest for %N, which increased by 54% in 2016 relative to baseline conditions, followed by microbial biomass, water holding capacity, and %C, which increased by 23%, 20%, and 15%, respectively (Fig. 2; Appendix S1: Tables S1, S2). Mean soil pH (+SE) across the 54 plots declined from 7.36 + 0.07 to 7.21 + 0.06, which is equivalent to a 41% increase in soil acidity (calculated using [H^+]), and mean bulk density (±SE) decreased from 1.37 ± 0.06 to 0.93 ± 0.02 g/cm^3, which is a 32% reduction in soil compaction (Fig. 2; Appendix S1: Tables S1, S2).

The magnitude of the effects of site preparation and tree planting also varied substantially by the experimental treatments (Table 1; Appendix S1: Table S1). Most notably, we observed a strong interaction between year and the compost treatments for all soil variables except bulk density (Appendix S1: Table S1), indicating that the soil responses to site preparation and tree planting were dependent on compost addition. In contrast, we

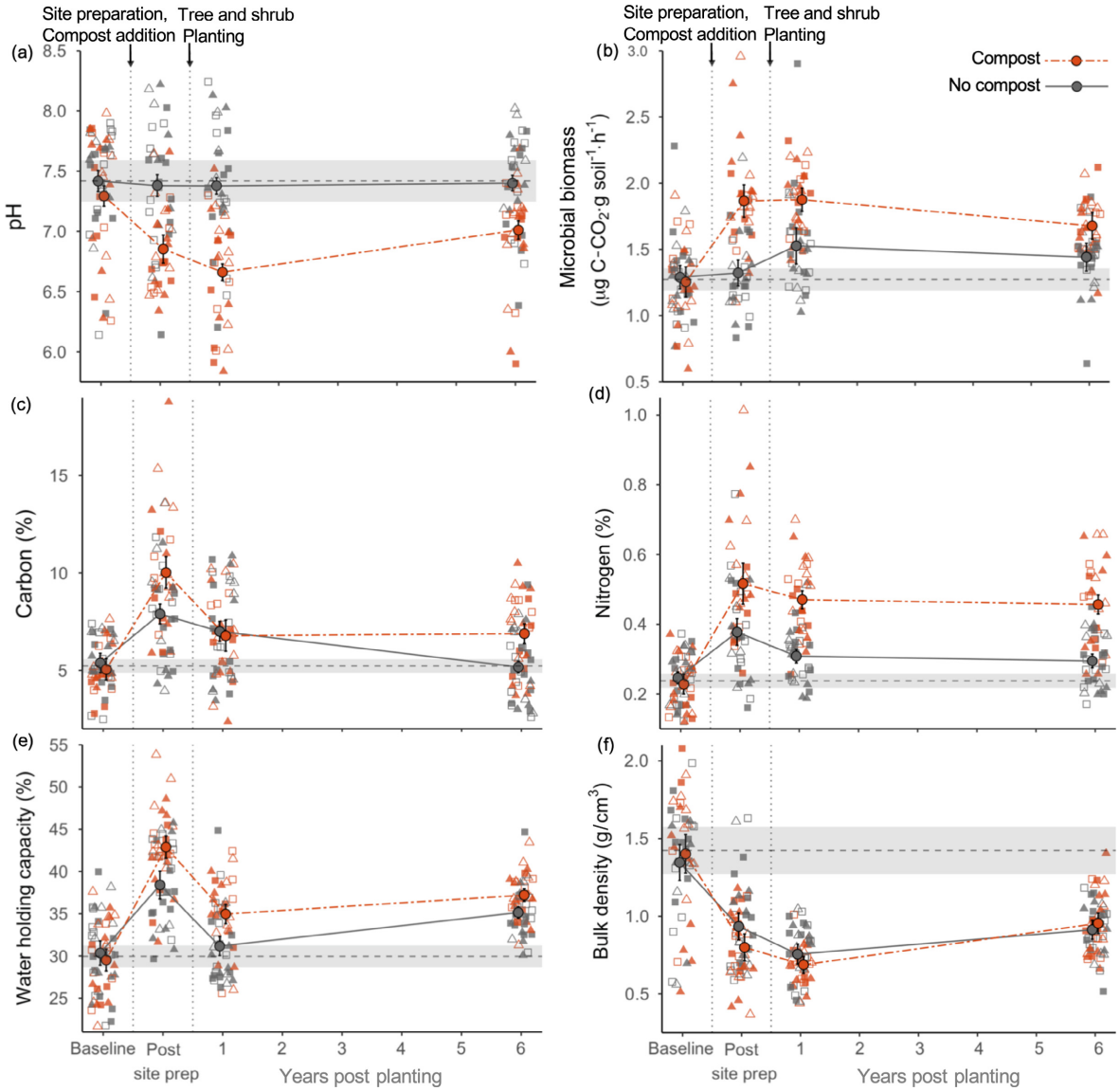

FIG. 2. Effects of site preparation (i.e., tilling, weeding, mulch application), afforestation, and the three experimental treatments (compost amendments, interplanting with shrubs, and tree species composition) on soil conditions over time. Compost-amended plots sustained larger soil responses than unamended plots for all measured variables (a)–(e) except bulk density (f). The horizontal dashed lines and shaded areas show mean baseline values and 95% confidence intervals. The double-dashed (red) and solid (gray) lines are used to help visualize directional trends in mean values over time for amended and unamended plots, respectively, and are not intended to represent interpolated data. Circles indicate the mean values for each year, and the error bars show the 95% confidence intervals, which are normalized to account for repeated measures of the same plot (see *Methods*). Compost-amended plots are shown in red and unamended plots are gray; six-species plots are triangular and two-species plots are square-shaped; and plots with shrub plantings are filled and plots without shrubs are unfilled.

found no evidence that the shrub or composition treatments influenced how soils responded to site preparation and tree planting since interactions between year and these treatments were weak (Appendix S1: Tables S1, S2). For pH and %C, only the compost-amended plots showed differences 6 yr postplanting relative to baseline conditions (Table 1). Compost-amended plots (n = 26) in 2016 were two times more acidic (7.01 + 0.08 mean pH + SE) than unamended plots (n = 28, 7.40 + 0.08), which were unchanged relative to baseline levels from 2009 (7.42 + 0.084; Table 1, Fig. 2). Likewise, mean %C (±SE) increased by 35% in compost-amended plots from 2009 (5.1 ± 0.2%) to 2016 (6.9 ± 0.3%) but was unchanged in the unamended plots (Table 1, Fig. 2). For microbial biomass, %N, and water holding capacity, both the amended and unamended plots showed significant differences in 2016 relative to baseline conditions, but the changes were more pronounced in compost-

amended plots (Table 1, Fig. 2). Mean (±SE) %N increased by 92% in amended plots (0.46 ± 0.025%) but only by 21% in unamended plots (0.29 ± 0.014%; Table 1, Fig. 2). Similarly, water holding capacity and microbial biomass increased by 24% and 32% in compost-amended plots but only by 17% and 13% in unamended plots, respectively.

We found no significant main effects of the experimental treatments on belowground C stocks 6 yr postplanting relative to baseline conditions across all 54 plots (Appendix S1: Fig. S2, Tables S1, S2). However, interpretation of the main effects is confounded by the opposing effects of the compost treatments as revealed by the significant year by compost interaction (Table 1; Appendix S1: Fig. S2, Table S1). Whereas amended plots had significantly higher soil C stocks relative to baseline conditions, unamended plots had significantly lower stocks (Table 1; Appendix S1: Fig. S2, Table S2). Mean (±SE) belowground C in compost-amended plots was 16.7% higher in 2016 (5.6 ± 0.29 kg C/m^2 at 2.5 kg soil depth) than in 2009 (4.8 ± 0.18 kg C/m^2 at 2.5 kg soil depth; Table 1; Appendix S1: Fig. S2, Table S2). In contrast, we observed a 10.4% decline in mean (± SE) soil C stocks in unamended plots in 2016 (4.3 ± 0.28 kg C/m^2 at 2.5 kg soil depth) relative to baseline conditions (Table 1; Appendix S1: Fig. S2, Table S2). In both instances, these changes marked a precipitous decline in soil C relative to peak conditions in 2010 immediately following site preparation (Appendix S1: Fig. S2).

(Q2) Changes in the experimental treatment effects over time (2011–2016)

Overall, the magnitude of the effects of site preparation, tree planting, and the experimental treatments diminished from 1 to 6 yr postplanting (Fig. 2; see Oldfield et al. 2014 for results for 2009–2011). Changes in soil conditions from 2011 to 2016 ranged from positive to neutral to negative, indicating that for some variables the effects of site preparation and planting continued to increase after 6 yr whereas for others it plateaued or began to decline (Table 2, Fig. 2). For instance, there was a positive association between water holding capacity and year, indicating that this variable continued to increase from 1 to 6 yr postplanting (Table 2, Fig. 2). However, we observed no changes in microbial biomass or %N from 2011 to 2016, suggesting that these variables plateaued after 1 yr (Table 2, Fig. 2). Moreover, the rise in %C and reduction in bulk density observed during the first 3 yr of this study diminished from 2011 to 2016. Percent C declined by 13% and bulk density increased

Table 1. Soil responses to site preparation and tree planting 6 yr postplanting (2016) relative to baseline (2009) conditions (Q1).

	Standardized model coefficients (±SE)				
	Intercept	Year (2009 vs. 2016)	Texture†	(Texture†)^2	Fixed R^2 (full)
pH					
Compost	**7.6 ± 0.093**	**−0.28 ± 0.060**	*−0.24 ± 0.13*	**−1.2 ± 0.28**	60 (82)
No compost	**7.5 ± 0.099**	−0.017 ± 0.055	−0.010 ± 0.16	*−0.42 ± 0.24*	10 (78)
Microbial biomass (μg C-CO_2·g·soil^{-1}·h^{-1})					
Compost	**1.3 ± 0.053**	**0.42 ± 0.074**	0.026 ± 0.074	–	40 (40)
No compost	**1.3 ± 0.052**	**0.15 ± 0.073**	0.025 ± 0.074	–	7.4 (7.4)
% Carbon					
Compost	**5.1 ± 0.33**	**1.8 ± 0.34**	−0.66 ± 0.56	–	26 (60)
No compost	**5.4 ± 0.31**	−0.21 ± 0.26	−0.26 ± 0.56	–	1.0 (63)
% Nitrogen					
Compost	**0.23 ± 0.020**	**0.23 ± 0.018**	−0.048 ± 0.035	–	58 (83)
No compost	**0.25 ± 0.014**	**0.049 ± 0.012**	−0.023 ± 0.026	–	12 (69)
% Water holding capacity					
Compost	**30 ± 0.71**	**7.6 ± 0.77**	−2.0 ± 1.20	–	55 (74)
No compost	**30 ± 0.78**	**4.8 ± 0.82**	−0.58 ± 1.3	–	26 (59)
Bulk density (g/cm^3)					
Compost	**1.4 ± 0.070**	**−0.44 ± 0.076**	0.063 ± 0.11	–	31 (58)
No compost	**1.3 ± 0.055**	**−0.43 ± 0.068**	0.017 ± 0.085	–	38 (50)
Soil carbon stock (kg C/m^2 at 2.5 kg soil depth)					
Compost	**4.8 ± 0.27**	**0.80 ± 0.24**	−0.39 ± 0.48	–	10 (66)
No compost	**4.8 ± 0.28**	**−0.55 ± 0.20**	0.00050 ± 0.53	–	3.4 (76)

Notes: Here we report standardized coefficients (±SE) to examine the compost by year (i.e., 2009 or 2016) interaction by testing the effect of year in plots with (n = 28) and without (n = 26) compost. We ran separate models for the two compost treatments because the strong interaction effects between this treatment and year suggested that soil responses to site preparation and tree planting were dependent on the addition of compost (Appendix S1: Table S1). Significant (P < 0.05) and marginally significant (P < 0.1) coefficients are shown in bold and italic fonts, respectively.

†The soil texture metric used is the ratio of coarse to fine particles calculated as % sand/(% clay + % silt).

by 29%, indicating higher levels of soil compaction in 2016 relative to 2011 (Table 2, Fig. 2).

In some instances, compost amendments sustained altered soil conditions even when the trend from 1 to 6 yr postplanting across all 54 plots was neutral or negative. For example, despite the overall decline in %C from 2011 to 2016, compost-amended plots had constant %C values during this time frame, indicating that the compost treatment counteracted the decline in %C observed in unamended plots (Table 2, Fig. 2). We also found significant interactions between year and compost for pH, water holding capacity, and bulk density. Although soil acidity decreased by 55% in amended plots between 2011 (6.66 + 0.09, mean pH + SE) and 2016 (7.01 + 0.08), the compost treatment still sustained significantly higher soil acidity (i.e., lower pH values) than unamended plots in 2016 (Table 1, Fig. 2; Appendix S1: Table S2). The increase in mean water holding capacity from 2011 to 2016 was higher in unamended plots than in compost-amended plots, but amended plots still retained higher overall water holding capacities in 2016 (Table 1, Fig. 2; Appendix S1: Fig. S2). For bulk density, the positive effect of year was more pronounced in amended plots than in unamended plots, but this trend did not result in significant differences in bulk density between the compost treatments in 2016 (Tables 1, 2, Fig. 2; Appendix S1: Table S2). There were no significant interactions between year and the shrub or composition treatments for any of the measured soil variables (Table 2).

(Q3) Association between soil conditions and tree growth

Mean basal area for planted trees in 2016 was 0.116 m^2/ha (±SE 0.04), but this value ranged from 0.002 to 0.513 m^2/ha among the plots. *Tilia americana* had higher mean basal area (0.512 ± 0.006) than *Quercus rubra* (0.063 ± 0.003). When *T. americana* and *Q. rubra* were evaluated together, soil %N had a significant positive association with basal area, and microbial biomass had a marginally significant negative association but only once it exceeded ~1.5 μg $C\text{-}CO_2 \cdot g\ soil^{-1} \cdot h^{-1}$ (Appendix S1: Table S3). These associations were, however, driven entirely by the *T. americana* response to soil characteristics. When analyzed separately, %N continued to have a significant positive association with *T. americana* basal area, and microbial biomass had a marginally negative association. However, none of the three tested soil variables were significant predictors of *Q. rubra* basal area (Appendix S1: Table S3). When plots amended with compost and unamended plots were analyzed separately, none of the soil variables tested were significant predictors of basal area in amended plots (Table 3; Appendix S1: Fig. S3). In contrast, unamended plots had significantly higher basal area as %N increased and significantly lower basal area as microbial biomass exceeded 1.5 μg $C\text{-}CO_2 \cdot g\ soil^{-1} \cdot h^{-1}$ (Table 3; Appendix S1: Fig. S3).

TABLE 2. Standardized coefficients (±SE) for the linear mixed models used to assess soil responses to the experimental treatments from 1 (2011) to 6 (2016) years postplanting (Q2).

	Soil response variable					
Variable	pH	Microbial biomass ($\mu g\ C\text{-}CO_2 \cdot g \cdot soil^{-1} \cdot h^{-1}$)	% Carbon	% Nitrogen	% Water holding capacity	Bulk density (g/cm^3)
Intercept	**7.5 ± 0.13**	**1.4 ± 0.058**	**6.2 ± 0.46**	**0.31 ± 0.023**	**34 ± 0.85**	**0.81 ± 0.040**
Year	0.033 ± 0.071	0.025 ± 0.11	**−2.0 ± 0.54**	−0.019 ± 0.016	**3.5 ± 0.78**	**0.14 ± 0.043**
Compost	**−0.55 ± 0.12**	**0.32 ± 0.056**	*0.89 ± 0.53*	**0.17 ± 0.026**	**3.4 ± 0.97**	−0.033 ± 0.046
Shrub	−0.11 ± 0.11	0.0037 ± 0.052	−0.48 ± 0.50	−0.033 ± 0.024	−0.95 ± 0.92	0.020 ± 0.043
Composition†	0.16 ± 0.12	*−0.095 ± 0.055*	0.014 ± 0.52	0.0019 ± 0.026	−0.88 ± 0.97	0.041 ± 0.045
Year × Compost	**0.33 ± 0.068**	−0.13 ± 0.11	**1.9 ± 0.60**	−0.00014 ± 0.018	**−2.1 ± 0.87**	**0.11 ± 0.048**
Year × Shrub	−0.054 ± 0.066	−0.13 ± 0.11	0.29 ± 0.58	0.010 ± 0.017	0.069 ± 0.84	0.039 ± 0.046
Year × Composition†	0.032 ± 0.068	0.070 ± 0.11	0.036 ± 0.60	0.00086 ± 0.018	1.4 ± 0.87	−0.0044 ± 0.048
Texture‡	*−0.23 ± 0.12*	−0.0046 ± 0.054	−0.83 ± 0.52	*−0.048 ± 0.025*	−1.4 ± 0.95	0.049 ± 0.045
Fixed R^2 (Full)	38 (91)	29 (29)	15 (57)	45 (88)	29 (76)	29 (68)

Notes: The three experimental treatments include compost amendments (presence or absence), interplanting with shrubs (presence or absence), and tree species composition (two or six species). Soil compost amendments consistently improved soil conditions through time in the afforestation site, whereas the long-term effects of the shrub and composition treatments were minimal. Significant ($P < 0.05$) and marginally significant ($P < 0.1$) coefficients are shown in bold and italic fonts, respectively.

†The categorical composition variable is analyzed as a shift from two- to six-species plots in the regression analysis.

‡The soil texture metric used is the ratio of coarse to fine particles calculated as % sand/(% clay + % silt).

Discussion

As investments in urban tree planting initiatives grow, our understanding of best practices for the implementation and management of these projects should also increase. Urban soils, particularly those of anthropogenic origin, are an understudied but integral aspect of afforestation that affect the capacity of a site to support a range of management goals (Pavao-Zuckerman 2008, Oldfield et al. 2014). Globally, soil health initiatives are touted as a means to mitigate climate change through below-ground C sequestration while also enhancing ecological processes, such as microbial activity and nutrient cycling (Bradford et al. 2019, Vermeulen et al. 2019). Although soil health is context dependent, certain properties and processes are known to support common management objectives and can therefore be used as system-specific indicators of restoration success (Heneghan et al. 2008, Pavao-Zuckerman 2008, Bünemann et al. 2018). Soil pH governs chemical and biological processes that affect plant nutrient uptake; water holding capacity and bulk density reflect soil porosity, aeration, and moisture availability required for root growth, water infiltration, and drought resistance; C and N concentrations are indicative of the amount and composition of organic matter and thus potential nutrient availability; and microbial biomass controls the cycling and transformation of nutrients, thereby making them available for plant uptake through decomposition. Each of these variables showed significant improvements 6 yr postrestoration relative to baseline conditions, and the response was largest in the compost-amended plots. These soil health improvements also had positive impacts on urban forest management objectives, including increasing variables indicative of stormwater mitigation, belowground C storage, and native tree growth.

Urban soils often have elevated pH levels owing to chemical inputs from runoff and weathering of impervious surfaces, such as concrete (Craul 1985, Jim 1998*a*). Within forested natural areas in New York City, for instance, pH increases across gradients of soil and canopy disturbance (Pregitzer et al. 2016), and these changes are often associated with higher invasive plant cover (Ward et al. 2020). Nonurban temperate deciduous forests, on the other hand, commonly have acidic soils (Osman 2013), and the optimal pH values for the planted tree species at the experimental site range from slightly acidic (4.5–6.8 for *Q. alba* and 5.0–6.6 for *C. laciniosa*) to moderately alkaline (4.8–7.3 for *C. glabra*, 4.3–7.3 for *Q. rubra*, 4.0–7.5 for *P. serotina*, 4.5–7.5 for *T. americana*, and 6.0–7.8 for *C. occidentalis*; Crow 1990, Krajicek and Williams 1990, Marquis 1990, Rogers 1990, Sander 1990, Schlesinger 1990, Smalley 1990, Osman 2013, U.S. Department of Agriculture [USDA] 2020). Therefore, the reduction in mean soil pH in the compost-amended plots (7.0) relative to baseline conditions (7.3) indicates that pH may be approaching an optimal value range for all planted tree species. However, the soils in these plots were still substantially less acidic than mature forests within New York City (mean pH of 5.21; Ward et al. 2020), and the rise in pH from 2011 to 2016 suggests that the effects of the compost amendments on pH may be diminishing. Yet the planted trees are young (≤11 yr old), and over time we expect

Table 3. Standardized coefficients (±SE) for the linear mixed models (LMMs) used to analyze the association between soil conditions and planted tree growth (basal area in m^2/ha) for *Tilia americana*, *Quercus rubra*, and for both species combined (Q3).

	Standardized model coefficients (±SE)		
	Tilia americana	*Quercus rubra*	*T. americana* + *Q. rubra*
Compost			
Intercept	**0.0025 ± 0.0002**	0.0009 ± 0.0001	**0.0019 ± 0.0001**
$[H^+]$	−0.0002 ± 0.0002	−0.0001 ± 0.0001	−0.0001 ± 0.0001
Microbial biomass + (μg C-CO_2·g $soil^{-1}$·h^{-1})	−0.0002 ± 0.0021	*−0.0013 ± 0.0009*	−0.0004 ± 0.0018
Microbial biomass − (μg C-CO_2·g $soil^{-1}$·h^{-1})	0.0029 ± 0.0019	0.0001 ± 0.0008	0.0030 ± 0.0017
% N	−0.00003 ± 0.0001	−0.0002 ± 0.0001	*−0.0001 ± 0.0001*
Initial DRC (cm)	**0.0006 ± 0.0001**	0.0001 ± 0.0001	**0.0009 ± 0.0001**
No compost			
Intercept	0.0018 ± 0.0001	0.0010 ± 0.0003	0.0015 ± 0.0003
$[H^+]$	−0.00004 ± 0.0001	−0.00003 ± 0.0001	−0.00002 ± 0.0001
Microbial biomass + (μg C·g $soil^{-1}$·h^{-1})	0.0009 ± 0.0016	*0.0015 ± 0.0008*	0.0014 ± 0.0014
Microbial biomass − (μg C-CO_2·g $soil^{-1}$·h^{-1})	*−0.0030 ± 0.0015*	**−0.0022 ± 0.0008**	**−0.0040 ± 0.0012**
% N	**0.0003 ± 0.0001**	**0.0004 ± 0.0001**	**0.0006 ± 0.0001**
Initial DRC (cm)	**0.0005 ± 0.0001**	**0.0002 ± 0.0001**	**0.0006 ± 0.0001**

Notes: Microbial biomass exhibited a non-linear, unimodal relationship with tree size, so coefficients for the positive and negative direction of microbial biomass are denoted by (+) and (–), respectively. We used hydrogen ion concentration $[H^+]$ in lieu of pH in the LMMs since it is on a linear rather than logarithmic scale. We analyzed changes in each variable in the subset of plots with (n = 26) and without (n = 28) compost based on strong soil responses to this particular treatment. For the full model (n = 54) results, see Appendix S1: Table S2. Significant (P < 0.05) and marginally significant (P < 0.1) coefficients are shown in bold and italic fonts, respectively. Initial diameter of root collar (DRC) was used to account for variation in sapling size on planting.

maturing trees to exert greater effects on soil acidity through increasing quantities of decomposing organic inputs into the soil.

Improving hydrology through natural solutions is a critical goal for urban land management since chemical-laden runoff is a major environmental pollutant (Booth and Jackson 1997, Kuehler et al. 2017, Phillips et al. 2019). Urban areas are often characterized by expansive impervious surfaces and compacted soils (Effland and Pouyat 1997), which limits their capacity to retain large volumes of water following storm events. We observed a significant decrease in bulk density—signifying reduced soil compaction—and an increase in water holding capacity, indicating that the physical soil environment is better able to support key ecosystem services through improved porosity, aeration, and moisture availability. Indeed, mean bulk density values in 2016 (0.93 g/cm^3) are within the range of values for uncultivated forests and grasslands (0.8–1.2 g/cm^3; Brady and Weil 2008) and similar to baseline conditions of forested reference sites within the park (0.84 g/cm^3; Oldfield et al. 2014). Moreover, water holding capacity was the only variable that continued to improve from 1 to 6 yr postplanting, and this is likely the result of increasing root growth of the planted trees, which can increase water infiltration and retention. Although peak water holding capacity values were observed in 2010 following site preparation and compost applications (Oldfield et al. 2014), improvements from 2011 to 2016 suggest that the benefits of afforestation—rather than site preparation—are beginning to accrue in amended and unamended plots alike.

We found that compost amendments were key to sustaining long-term increases in belowground C storage as a result of urban afforestation. In the absence of compost, the effects of site preparation and planting on C concentrations and stocks were neutral and negative, respectively, despite the surface application of mulch to all plots. These findings are consistent with previous research that has found that afforestation can initially decrease soil C (Berthrong et al. 2009, Laganière et al. 2010, Wellock et al. 2011, Berthrong et al. 2012, Li et al. 2012), particularly when planting is associated with high levels of disturbance during site preparation (Laganière et al. 2010) or when initial C concentrations are high (Hong et al. 2020). However, we found that C concentrations and stocks both increased with the addition of compost. In heavily used and/or degraded urban sites, soil compaction can limit tree growth, necessitating tilling practices to reduce bulk density (Jim 1993, Jim 1998*b*). However, tillage also depletes soil C by disrupting aggregate formation, thereby exposing organic matter to levels of decomposition that exceed plant C inputs (Six et al. 2000, Haddaway et al. 2017). Yet our results suggest that compost amendments can offset the negative impacts of tilling on soil C in urban afforestation projects over longer time frames. Other studies have similarly found that organic amendments are required to build C stocks in urban soil remediation projects, particularly when initial organic matter concentrations are low (Beesley 2012, Chen et al. 2013). Rebuilding soil C in intensively managed systems has a multitude of co-benefits beyond climate mitigation (Bradford et al. 2019, Vermeulen et al. 2019). For instance, trends in soil microbial biomass, C stocks, and C and N concentrations observed in this study signify a buildup of organic matter as well as the potential mineralization of nutrients through microbial activity, which improves soil fertility. Thus, our results suggest that, when coupled together, compost amendments and tilling can have long-term positive impacts on soil physical, chemical, and biological properties and processes required to support tree growth and to sequester and store belowground C.

Although we observed long-lasting impacts of site preparation and compost on soil conditions, the linear trends in soil pH (analyzed as [H^+]), bulk density, and microbial biomass observed during the first 3 yr of this study (Oldfield et al. 2014) tapered or declined from 2011 to 2016. Extrapolation of initial trends can therefore lead to poor projections compared to observed values, emphasizing the importance of longer-term studies for quantifying restoration outcomes. Moreover, it remains unclear whether the positive effects of restoration continue to be driven by initial site preparation and compost amendments (Oldfield et al. 2014), or whether they are increasingly the result of tree growth, canopy closure, and understory plant recruitment. Over time, we would expect the growth of planted trees to increase in their effects on soil conditions. For example, planted trees can reduce bulk density and increase water holding capacity through root expansion (Vidal-Beaudet et al. 2018) as well as increase soil C stocks as leaf litter and fine root inputs are incorporated into the soil (Six et al. 2002). Although we were unable to test these relationships directly with an unplanted (i.e., no trees) experimental treatment since the project was a collaboration with city agencies in a high visibility park with defined public–private tree-planting goals, we advocate for future work that specifically examines the role that planted trees play in improving soil conditions relative to other site treatments.

Because the primary goal of urban afforestation projects is to establish closed-canopy, native-dominated forests (Gobster 2010), we also evaluated the effects of improved soil conditions on planted tree growth. Although we did not detect any changes in tree growth in response to soils in plots that were amended with compost, tree growth increased significantly with increases in %N in unamended plots. This finding suggests that compost amendments shifted soil conditions so that N was no longer limiting tree growth. Indeed, this one-time compost application appeared sufficient to transition the soils at our study site to a threshold that could support tree growth (in 2016 %N ranged from 0.17% to 0.45% in unamended plots versus 0.26% to 0.66% in amended plots). Compost applications in

urban settings can range in intensity from low-cost surface spreading to more time- and cost-intensive methods that incorporate compost into the soil to a depth of up to 60 cm (Kranz et al. 2020). In our study, we incorporated compost to 15 cm and found that increases in tree growth diminished as soil N approached ~0.4%, suggesting that more time- and cost-intensive practices may not yield the same incremental benefits. Furthermore, responses to soil conditions were species-dependent. *Tilia americana* demonstrated more vigorous growth as a result of improved soil conditions whereas *Q. rubra* did not. These species-specific responses are worth considering given that compost applications can be expensive (Environmental Protection Agency 2020). Moreover, Doroski et al. (2018) found that regeneration of nonnative woody plants was significantly higher in compost-amended plots at the experimental site, which could jeopardize goals to establish native-dominated forests in urban areas (Pregitzer et al., 2021). As such, we suggest that land managers consider potential trade-offs between compost applications, planted tree composition, and recruitment from nonnative and invasive species.

Afforestation of degraded urban landscapes can involve obstacles different from those found in nonurban settings, such as remediating disturbed soil conditions of anthropogenic origin (Effland and Pouyat 1997, Pavao-Zuckerman 2008). Yet as the proportion of the global population residing in cities continues to grow, investments in restoration projects that support and improve ecosystem services in urban greenspaces will become increasingly important (Pataki et al. 2011, Elmqvist et al. 2015). We found marked improvements in soil health conditions 6 yr postrestoration relative to baseline conditions. Although indicators of soil quality are highly context-dependent (Bünemann et al. 2018), our results support the future implementation of soil-focused site treatments in our study region and can help inform expectations for the duration and effects of different restoration practices. In particular, the addition of compost had substantial, long-lasting effects and increased the capacity of the site to support other management objectives, such as belowground C storage and the growth of planted trees. These planted trees will, in turn, hopefully support a multitude of additional ecosystem services, such as aboveground C storage. Thus, our results suggest that compost applications along with targeted plant selection may be critical to realizing the potential of urban afforestation projects as nature-based solutions in moist, temperate systems.

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Supporting Information

Additional supporting information may be found online at: http://onlinelibrary.wiley.com/doi/10.1002/eap.2336/full

Data Availability

Data (Doroski and Ward 2021) are available in the Dryad Digital Repository. https://doi.org/10.5061/dryad.qnk98sffm