



## Positive effects of afforestation efforts on the health of urban soils



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### ARTICLE INFO

#### Article history:

Received 18 November 2013

Accepted 20 November 2013

Available online 12 December 2013

#### Keywords:

Afforestation

Ecosystem services

Restoration

Urban forestry

### ABSTRACT

Large-scale tree planting projects in cities are increasingly implemented as a strategy to improve the urban environment. Trees provide multiple benefits in cities, including reduction of urban temperatures, improved air quality, mitigation of storm-water run-off, and provision of wildlife habitat. How urban afforestation affects the properties and functions of urban soils, however, is largely unknown. As healthy soils are critical for vigorous tree growth, our study explores the impact of soil restoration as part of a large-scale urban afforestation project. We collected data on multiple soil variables over the first three years of the New York City Afforestation Project (NY-CAP). The study consists of 56 plots of 225 m<sup>2</sup> arrayed across an urban parkland in Queens, NYC. Each plot contains 56 trees made up of two (low richness) versus six (high richness) native species. The richness treatment was crossed with stand complexity (with shrubs and herbs versus without), and soil amendment (with compost versus without). We sampled soils in 2009 prior to project establishment, in 2010 following site preparation but just prior to planting, and again in 2011 one year after the 3–5 year old saplings were planted and plot treatments were put in place. We present results for the effects of site preparation on soil properties over time from baseline conditions in 2009 through the first year of afforestation in 2011. We also explore the impact of plot treatments (listed above and implemented right after our 2010 soil sampling) on soil parameters in 2011. Overall, site preparation improves soil conditions for the native tree saplings across time, with reductions in bulk density from ~1.4 to 0.72 g cm<sup>-3</sup>, acidification of the soil from pH 7.36 to 7.03, a 4-fold increase in microbially-available carbon and a 1.3-times increase in microbial biomass. Furthermore, soil carbon concentrations increased by 1.33-times between 2009 and 2011. Exploring plot treatments in 2011, compost had the largest effect, with 1.23-times more microbial biomass in composted plots, more acidic pH values (6.66 versus 7.37 in non-composted plots) and increased water holding capacity (35% versus 31% in non-composted plots). The observed changes in soil physical, chemical and biological properties suggest that site preparation and management improves traits of urban soils that are critical for infiltration, decomposition, mineralization and nutrient retention. The initial trajectories of change in these soil properties provide support for the expectation that urban afforestation – and specifically the preparation of urban soils for tree planting – will improve the health of urban soils and consequently the urban environment.

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### 1. Introduction

Afforestation has increased in pace and extent in recent years, as policies for greenhouse gas mitigation drive the conversion of other land uses into forests (Berthrong et al., 2009). Studies on

the impacts of afforestation have focused primarily on the ability of newly created forests to sequester carbon in tree biomass and soils (Guo and Gifford, 2002; Vesterdal et al., 2002; Berthrong et al., 2012). Though much of this work has focused on the establishment of plantation forests in natural areas, afforestation projects are also increasingly common in cities. There, as in natural lands, projects are intended to capture carbon as well as improve air quality, lower air temperatures, increase storm-water infiltration and create wildlife habitat (Oldfield et al., 2013). These benefits rely on healthy urban soils to facilitate vigorous tree growth and to improve the environment for soil microbes whose activities cycle nutrients through decomposition and store carbon through

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the production and aggregation of microbial-derived compounds, the primary constituents of stabilized soil organic matter (Schmidt et al., 2011). Urban afforestation efforts have traditionally relied on street-tree plantings, but more recently cities such as Auckland, London, Los Angeles and New York have implemented large-scale, tree-planting campaigns to establish contiguous stands of urban forest composed predominantly of native species (Oldfield et al., 2013).

Assessments of how urban forests benefit people living in cities unanimously conclude that tree cover improves the urban environment (Brack, 2002; Nowak et al., 2002; Davies et al., 2011). These assessments, however, are based on established mature trees primarily planted along streets. Assessments of how urban forest stands affect ecosystem properties related to the health of the environment have instead been made across urban-to-rural gradients. Gradient studies have focused primarily on remnant forest patches that compare in age and composition to rural forests (Pouyat et al., 2002; Golubiewski, 2006; Pouyat et al., 2009). Largely unanswered is what happens to the properties of urban soils as they undergo afforestation. Data assessing the efficacy of urban afforestation projects at improving soil health are, as for urban afforestation effects on ecosystem properties and processes in general, necessary but lacking (Pataki et al., 2011). Cities are then investing in urban afforestation projects without knowing whether these new forests will provide the expected benefits to the urban environment.

Of critical concern to urban afforestation is whether or not urban soils can effectively support forest vegetation. To support a growing forest, soils need to provide physical, chemical and biological conditions that provide adequate physical support, oxygen concentrations, and nutrient and water availability. To this end, many urban soils require remediation and/or improvement as urban afforestation projects are often implemented on filled wetlands or land converted from urban or industrial land uses. Urban soils are typically anthropogenically altered or created, and commonly are compacted with high percentages of human-made artifacts (>10%), including concrete, asphalt, brick and coal slag (NRCS, 2010). It is then an open question as to whether such soils can be remediated sufficiently to facilitate the establishment and growth of stands of healthy trees.

Given that successful forest growth relies on creating healthy soils, soil ecological knowledge can increase our understanding of how ecosystems respond to restoration (Ruiz-Jaen and Aide, 2005; Heneghan et al., 2008; Pavao-Zuckerman, 2008). Yet soils receive little attention in restoration projects compared to vegetation performance metrics such as growth, survival and diversity (Callahan et al., 2008; Heneghan et al., 2008). Our project helps redress this imbalance by assessing the effects of site preparation and different land managements (e.g., compost amendment and tree species diversity) on soil health at an afforestation site located in New York City (Fig. 1). Our project is a research component of the City's MillionTreesNYC Initiative. We assess how soil restoration and site managements affect key physical (e.g., bulk density), chemical (e.g., carbon concentrations) and biological (e.g., microbial biomass) properties of soils necessary for vigorous tree performance because of their influence on soil nutrient supply, aeration, moisture retention and hence root growth.

## 2. Methods

### 2.1. Site description and experimental design

Our experiment is dubbed the New York City Afforestation Project (NY-CAP). It is situated in Kissena Corridor Park (40°44'N, 73°49'W; 114 cm MAP, 13 °C MAT), a 40-ha urban park in eastern

Queens, New York that includes recreational fields and facilities, a community garden and parkland. Situated in the interior of the park are 56 afforestation research plots (Fig. 1). Urban afforestation at our site, as in much of the MillionTreesNYC Initiative, focuses on restoring public parkland and so our plots were located in areas densely overgrown with and dominated by a small number of largely invasive, herbaceous species, such as mugwort (*Artemisia vulgaris*) and phragmites (*Phragmites australis*) as well as native weedy species like goldenrod (*Solidago canadensis*). The Natural Resources Conservation Service (NRCS) classified soils across Kissena as Laguardia-Ebbets complex, meaning the soils are well drained, loamy-skeletal to coarse-loamy fill soils with more than 10% human-derived artifacts. Our research plots fall in the Ebbets series, characterized by <35% coarse fragments (NRCS, 2009).

Research plots were separated into eight different treatments, consisting of a crossed arrangement of tree species richness (six species versus two species), stand complexity (with shrubs and herbs versus without), and soil amendment (with compost versus without). We refer to these treatments as diversity, shrub and compost. Replication is uneven and is organized as follows: high diversity/shrubs/compost,  $n = 9$ ; high diversity/no shrubs/compost,  $n = 9$ ; high diversity/shrubs/no compost,  $n = 5$ ; high diversity/no shrubs/no compost,  $n = 5$ ; low diversity/shrubs/compost,  $n = 5$ ; low diversity/no shrubs/compost,  $n = 5$ ; low diversity/shrubs/no compost,  $n = 9$ ; low diversity/no shrubs/no compost,  $n = 9$ . Each plot is  $15 \times 15$  m ( $225 \text{ m}^2$ ) and includes 56 trees planted 2.1 m from the center of each other's trunks. The tree species in low diversity plots are 28 *Tilia americana* and 28 *Quercus rubra*. The high diversity plots comprise eight individuals of *T. americana* and *Q. rubra*, plus 10 individuals of *Quercus alba*, *Celtis occidentalis*, *Carya* spp. and *Prunus serotina* (Fig. 2a and b).

Planted trees were 3–5 year old saplings measuring approximately 0.6–1.2 m in height, with root masses contained in either 1 gallon or 2 gallon ( $\sim 3.79$  or  $7.58$  L, respectively) containers. Trees were planted with a hand-held mechanized post-hole digger in holes of appropriate size to house the tree roots ( $\sim 25$  cm diameter and  $\sim 25$  cm deep). Half of the 56 plots received compost (see details below), and half were planted with shrubs (5 species, 41 plants per plot) and herbaceous plantings (7 species, 672 plants per plot), in a crossed design with the compost amendments (see paragraph above). The most represented shrub species include *Sambucus canadensis*, *Hamamelis virginiana*, and *Viburnum dentatum*; herbaceous species include *Apocynum cannabinum*, *Asclepias syriaca*, and *Panicum virgatum*. A full species list is included in Felton et al. (2013).

### 2.2. Site preparation

The areas for afforestation received extensive site preparation in advance of the tree, shrub and herbaceous plantings. Site preparation was performed by landscape contractors according to specifications outlined by the New York City Department of Parks & Recreation through a contractual agreement. The site preparation details outlined below were obtained from this contract.

Soils were weeded and rototilled to de-compact soil and loosen large debris to a depth of  $\sim 15$  cm. Debris included "objectionable material" such as trees up to 15 cm diameter, shrubby growth, brush, vines, ground covers, stumps of all sizes, roots, weeds, stones, wood, and human-derived debris (e.g., blocks of concrete and scrap metal). The compost treatment plots were then amended with compost at a rate of  $2.5 \text{ m}^3$  per  $100 \text{ m}^2$ , incorporated to 15 cm depth. The commercial compost consisted of a blend of nutrient rich bio-solids and clean, ground wood. The compost was analyzed prior to addition and had a pH of 6.3, a bulk density of  $457 \text{ kg m}^{-3}$ , 60% C, 3.2% N, 3.7% P and 0.44% K (dry weight basis). In the following year (2010), all research plots received a surficial layer



**Fig. 1.** The location and orientation of the NY-CAP research plots ( $n = 56$ ) located within Kissena Corridor Park in Queens, NY. Plot treatments include high versus low tree species diversity; with the addition of shrub and herbaceous plantings versus without; and with the addition of compost and without.

(5 cm) of mulch to minimize drought-stress on the planted saplings. The mulch was shredded hardwood procured from tree material as opposed to wood-waste or wood by-products. We do not have information on the nutrient content of the mulch, but do know that the pH was between 5.8 and 7.

### 2.3. Soil sampling

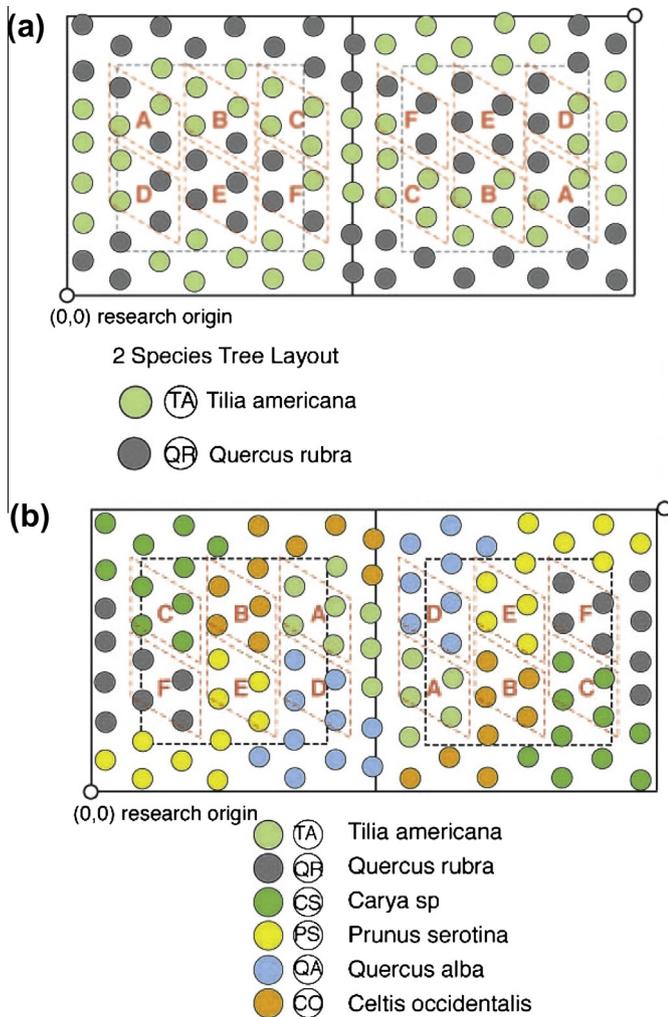
We sampled soils on three successive occasions to capture baseline conditions (year 1), site preparation impacts (year 2), and conditions after the first year of afforestation (year 3), where the latter sampling includes effects of both site preparation and treatment effects. Soil sampling took place in October 2009 before site preparation to capture the initial soil conditions of the plots. Sampling was repeated in October 2010 following site preparation but before tree, shrub and herbaceous plantings were planted later that same month. We then sampled in October 2011, approximately one year after the trees were planted and plot treatments (diversity, shrub and compost) were put into place. Our sampling scheme then captures how soil properties changed with site preparation (effects over all plots from 2009 to 2011) and how plot treatments influenced soil parameters (effects per treatment from 2010 to 2011).

We also collected soil samples from five reference zones across the parkland, including two *Robinia pseudoacacia* and two *Rhus typhina* stands, as well as under a *Populus deltoides* stand. We took these reference soils as a way to compare changes in soil parameters over time in research plots versus areas under continuous urban forest cover at the same location and so under the same soil and climate conditions.

Within each plot and reference zone, we collected 5 soil cores (8 cm diameter) at two depths (~0 to 8 cm and 8–15 cm) for a total of 610 soil cores in 2009 and again in 2010. We collected soils at two depths to quantify changes in soil carbon stocks following site preparation. We sampled surface soils only (~0 to 8 cm) in 2011 for a total of 305 cores. Samples were collected in the four plot corners, approximately 2–3 meters from the plot edge, and in the center of each plot. We combined data from 2009 and 2010 surface soils with data from 2011 to assess site preparation effects on soil properties such as pH and microbial biomass across time. In 2011, our soil-sampling scheme avoided the area immediately around saplings (i.e., we sampled at least 50 cm away from woody trunks) and so did not include any imported nursery soil that came in with planted tree or shrub roots. We had intended to sample to 30 cm depth in 2009 and 2010, but severe subsurface soil compaction prevented us from hand-coring below ~15 cm. For each soil sample we recorded the depth from the surface and the mass. Soil cores were pooled by plot and depth to minimize the influence of fine-scale spatial variation in our measurements, giving per year 56 surface and 56 subsurface (in 2009 and 2010 only) research plot samples, and an additional 5 reference soils. Samples were sieved to 4 mm, homogenized and stored at 5 °C or air-dried prior to analyses.

### 2.4. Tree health and survival

In August 2011, ~10 months after trees were planted, we conducted a field survey of tree health and survival for the 24 trees centrally located in each research plot for a total assessment of 1296 trees. Tree health was based on a visual assessment of tree



**Fig. 2.** Layout and species of trees within the 56 research plots: (a) low and (b) high diversity plots.

vigor, which rated trees on a scale of 1–5. A rating of 1 signified less than 10% branch or twig mortality, defoliation or discoloration; a 2 signified between 10% and 25%, a 3 between 26% and 50%, a 4 more than 50%, and trees that were dead received a rating of 5. This vigor metric was developed and used by the US Forest Service for the North American sugar maple decline project (Cooke et al., 1996) and has been used in other assessments of forest stand health in the US (Horsley et al., 2000).

### 2.5. Lab analyses

To measure soil pH, water and field-moist soil were mixed in a 1:1 volumetric ratio, allowed to stand for 10 min, and pH was then estimated in the supernatant using a bench-top pH meter. Gravimetric moisture was determined by oven drying to constant mass at 105 °C and water-holding capacity by wetting the soil to beyond field-capacity, allowing it to drip drain over filter paper for 2 h, before being weighed and then weighed again following oven drying. Non-sieved soil cores were air-dried and used to determine bulk density based on core volume (depth × area) and oven-dry mass. Values were corrected for root and stone volume and mass retained on a 2 mm sieve.

Microbial biomass was determined using a modified substrate-induced respiration technique (Fierer and Schimel, 2003) and labile

carbon availability using a lab incubation technique (Fierer et al., 2005; Bradford et al., 2008a). Substrate-induced respiration provides an index of microbial biomass by measuring rates of CO<sub>2</sub> efflux over a given incubation time. Soils (4 g dry weight equivalent) were incubated overnight at 20 °C, slurried with 4-mL autolyzed yeast solution by shaking for 1 h, and then capped with an airtight-lid modified for gas analysis (Bradford et al., 2008a). Samples were flushed with CO<sub>2</sub>-free air, and after 4 h of incubation at 20 °C, headspace CO<sub>2</sub> concentrations were measured using an Infra-Red Gas Analyzer (Li-COR model Li-7000, Lincoln, NE, USA). Carbon mineralization assays estimate the amount of labile carbon, also using an incubation technique. Soils (6 g dry weight equivalent) were adjusted to 65% water-holding capacity following Bradford et al. (2008b), and then analyzed for headspace CO<sub>2</sub> following 24 h at 20 °C. Cumulative carbon mineralized was determined by integrating at least five flux values across a 30-day period. Total carbon concentrations were measured after ball milling samples to a fine powder and then analyzing them using a Costech ESC 4010 Elemental Analyzer (Costech Analytical Technologies Inc., Valencia, CA).

To quantify how site preparation affected soil carbon stocks, we calculated change in carbon stocks from 2009 to 2010. We analyzed carbon stocks by cumulative mass coordinates (i.e., a mass-dependent approach). We chose this mass approach because we anticipated a reduction in bulk density with site preparation, and so sampling a single core to a fixed depth (i.e., a depth-dependent approach) would have been inadvisable because in 2009 a greater dry mass of soil would have been sampled than in 2010 (Gifford and Roderick, 2003; Wendt and Hauser, 2013). The cumulative mass approach accounts for the effects of changing bulk density on carbon stocks by estimating stocks to a standard dry soil mass rather than standard depth. Using cumulative mass coordinates requires the measurement of soil carbon in at least two soil cores: a surficial core and one immediately below (Gifford and Roderick, 2003). Linear interpolation is then used across values for the two cores to express mass of soil C per unit ground area per a standard soil mass (kg m<sup>-2</sup> of C to 0.1 Mg soil depth) (Gifford and Roderick, 2003).

### 2.6. Statistical analyses

Our analysis is divided into two components: (1) the effects of site preparation on soil parameters from 2009 to 2011; and (2) the impact of plot treatments (implemented following our 2010 soil sampling) on soil parameters in 2011. We used a linear mixed model (LMM) approach for both sets of analyses because the spatial layout of our research plots necessitated models that accounted for non-independent spatial and temporal associations in our data. Specifying plot as a random effect then accounted for the likelihood that plots clustered by location are more similar to each other than to other plots; an approach equivalent to accounting for nesting and repeated measures in ANOVA. For our first set of analyses about soil preparation, ‘year’ was identified as a fixed effect to assess how soils responded over time to site preparation.

For the second component of our analysis (the impacts of plot treatments on soil parameters in 2011) our statistical models also had to account for the unbalanced treatment design (see Section 2.1), and the LMM approach (in contrast to ANOVA) can accommodate both for this and the spatial non-independence of the layout. Plot was identified as a random effect with the treatments identified as fixed effects, allowing us to test interactions between treatments. Plot treatments included the addition of compost to the plot (amended or non-amended), the species diversity of the plot (high versus low) and the presence of a planted understory (with planted herbs and shrubs and without). To select the best models for soil response variables, we used the model with

the lowest Akaike information criterion (AIC) score (Burnham et al., 2011). Variance inflation factors <5 indicated that collinearity was sufficiently low among predictor variables. The LMMs were all fit assuming a Gaussian error distribution. Since the *F*-statistic is not considered to be accurate for the 'lme4' package (used for our LMMs), we used a Markov Chain Monte Carlo (MCMC) approach in the 'language R' package to estimate the mean and standard deviation of a distribution of values for each coefficient and the associated *P* values (Baayen, 2007; Baayen et al., 2008). These coefficients and *P* values retain the same interpretation as the classical frequentist statistics.

### 3. Results

#### 3.1. Site preparation impacts, 2009–2011

Across the 56 research plots, all of the surface soil properties that we measured changed significantly ( $P < 0.001$ ) from 2009 to 2011. Linear changes were observed for microbial available carbon, microbial biomass, pH and bulk density, indicating changes resulting from site preparation and compost amendment (Table 1). Microbial available carbon showed the greatest change of all variables, being about 4-times greater in 2011 than 2009. Microbial biomass also increased, but to a lesser degree (1.3-times). Soil pH decreased linearly (analyzed as  $[H^+]$ ) from 2009 to 2011, becoming more acidified following site preparation. Soil bulk density fell by about half from 2009 to 2011, revealing a marked reduction in soil compaction. In contrast to the linear changes in the other surface soil properties, soil carbon concentrations and water holding capacity showed peak values in 2010 despite increasing significantly overall from 2009 to 2011.

We analyzed reference plots to see how changes in soil properties within adjacent forested areas compared to our research plots (Table 1), enabling us to determine whether temporal changes resulted from site preparation or just natural variation. Five of the six soil properties were unaffected by time in the reference plots, but microbial available carbon did show an increase (MCMC coeff. = 71.87,  $P < 0.05$ ). This increase, however, was smaller than the response we observed in our research plots, being limited to a 1.6-times increase across 2009–2011 compared to the 4-times increase in our research plots. These different temporal changes meant that although the research and reference plots had similar

values in 2009 ( $\sim 250 \mu\text{g C g soil}^{-1}$ ), in 2011 the afforestation plot values were almost  $1000 \mu\text{g C g soil}^{-1}$  compared to  $\sim 400 \mu\text{g C g soil}^{-1}$  in the references.

#### 3.2. Soil carbon stocks, 2009–2010

Total carbon stocks ( $\text{kg m}^{-2}$  of C) increased from 2009 to 2010. The cumulative mass coordinates approach had stock values (mean  $\pm$  SE) of  $4.76 \pm 0.17$  ( $\text{kg m}^{-2}$  of C to 0.1 Mg soil depth) in 2009 and  $7.64 \pm 0.31$  ( $\text{kg m}^{-2}$  of C to 0.1 Mg soil depth) in 2010, demonstrating a significant increase in carbon stocks from 2009 to 2010 ( $F_{1,81} = 131$ ,  $P < 0.001$ ). We did not estimate carbon stocks for the reference plots, nor did we measure carbon stock values for 2011.

#### 3.3. Plot treatment effects, 2011

The full plot treatments were implemented in early November 2010 when the trees were planted, meaning that by the October 2011 soil sampling the plot treatments had  $\sim 1$  year to influence the soils. Compost was the only treatment to have a significant effect on soil properties, although diversity and shrubs were retained in some of the best-fit statistical models (Table 2), suggesting that they helped explain variance in the data despite not having significant effects. Compost did not significantly affect microbially available carbon, percentage soil carbon, nor bulk density; however, microbial biomass and water holding capacity were consistently higher with compost amendment (i.e.,  $P < 0.05$ ). In addition plots treated with compost had significantly lower pH ( $P < 0.001$ ).

#### 3.4. Tree health and survival

While compost did drive changes in certain soil properties, we did not see any significant impact of plot treatments on the survival or overall vigor of planted trees in summer 2011 (following planting in 2010). The mean vigor ( $\pm$ SE) across all plots and treatments was  $2.5 \pm 0.10$ , indicating that, in general, planted trees experienced between 10% and 50% branch mortality, twig die-back and/or foliage discoloration. Of the 1296 trees we assessed, only 28 trees were scored as dead (a vigor of 5) or missing.

**Table 1**  
Soil biological, physical and chemical properties of research and reference plots across time, 2009–2011.

Variable	Value (mean $\pm$ SE)			Coeff. (MCMC)	<i>P</i> value (MCMC)
	2009 Baseline conditions	2010 Post-site preparation	2011 Post-planting		
<i>(A) Research plots</i>					
Microbial available carbon ( $\mu\text{g C g soil}^{-1}$ )	261.28 $\pm$ 9.08 <sup>a</sup>	745.22 $\pm$ 49.16 <sup>b</sup>	981.04 $\pm$ 62.02 <sup>c</sup>	360.9	<0.001
Microbial biomass ( $\mu\text{g C g soil}^{-1} \text{ h}^{-1}$ )	1.28 $\pm$ 0.04 <sup>a</sup>	1.58 $\pm$ 0.06 <sup>b</sup>	1.69 $\pm$ 0.05 <sup>b</sup>	0.208	<0.001
pH	7.37 $\pm$ 0.06 <sup>a</sup>	7.13 $\pm$ 0.07 <sup>b</sup>	7.03 $\pm$ 0.08 <sup>b</sup>	−0.165	<0.001
Bulk density ( $\text{g cm}^{-3}$ )	1.42 $\pm$ 0.07 <sup>a</sup>	0.86 $\pm$ 0.04 <sup>b</sup>	0.72 $\pm$ 0.02 <sup>c</sup>	−0.350	<0.001
%C	5.18 $\pm$ 0.17 <sup>a</sup>	8.90 $\pm$ 0.39 <sup>b</sup>	6.89 $\pm$ 0.31 <sup>c</sup>	0.852	<0.001
Water holding capacity	29.82 $\pm$ 0.63 <sup>a</sup>	40.51 $\pm$ 0.70 <sup>b</sup>	33.00 $\pm$ 4.76 <sup>c</sup>	1.601	<0.01
<i>(B) Reference plots</i>					
	2009		2011		
Microbial available carbon ( $\mu\text{g C g soil}^{-1}$ )	253.22 $\pm$ 163.67 <sup>a</sup>		396.96 $\pm$ 14.05 <sup>b</sup>	71.87	<0.05
Microbial biomass ( $\mu\text{g C g soil}^{-1} \text{ h}^{-1}$ )	1.48 $\pm$ 0.44		1.56 $\pm$ 0.14	0.043	0.555
pH	7.22 $\pm$ 0.74		6.88 $\pm$ 0.17	−0.1712	0.167
Bulk density ( $\text{g cm}^{-3}$ )	0.84 $\pm$ 0.69		0.80 $\pm$ 0.08	−0.019	0.82
%C	4.79 $\pm$ 0.38		6.03 $\pm$ 0.98	0.626	0.233
Water holding capacity	28.52 $\pm$ 6.72		26.87 $\pm$ 1.56	−0.827	0.399

(A) Mean values  $\pm$  SE of surface (0–8 cm) soil parameters across research plots capturing baseline, post-site preparation, and the first year of afforestation, 2009–2011 ( $n = 56$ ); (B) mean values of soil parameters for reference plots, measured in 2009 and 2011 are also presented. Markov Chain Monte Carlo (MCMC) coefficients and *P* values are presented for each parameter, and are analogous to linear mixed effect regression coefficients and *P* values. Values with different superscripts are significantly different from each other ( $P < 0.05$ ).

**Table 2**  
Measured soil parameters under plot treatments in 2011.

Variable	2011 Treatment effects				Shrubs & herbs (–) (+)	Coeff. (MCMC)	P value (MCMC)		
	Compost		Diversity						
	(–)	(+)	Low	High					
Microbial available C ( $\mu\text{g C g soil}^{-1}$ )	1014.96 ± 113.75	948.32 ± 42.46	1068.59 ± 111.53	886.76 ± 42.23	951.30 ± 46.08	1010.79 ± 116.14	Diversity: –181.8 Compost: 0.38 Diversity: –0.13	0.15 <0.001 0.15	
Microbial biomass ( $\mu\text{g C g soil}^{-1} \text{ h}^{-1}$ )	1.53 ± 0.07	1.88 ± 0.05	1.71 ± 0.07	1.68 ± 0.07	1.66 ± 0.06	1.73 ± 0.08			
pH	7.37 ± 0.09	6.66 ± 0.09	7.02 ± 0.12	7.04 ± 0.11	7.06 ± 0.11	7.00 ± 0.12		Compost: –0.77 Diversity: 0.23	<0.001 0.10
Bulk density ( $\text{g cm}^{-3}$ )	0.76 ± 0.04	0.69 ± 0.02	0.72 ± 0.04	0.73 ± 0.03	0.72 ± 0.03	0.72 ± 0.04		Compost: –0.07	0.16
%C	7.00 ± 0.46	6.78 ± 0.41	6.85 ± 0.35	6.94 ± 0.52	7.06 ± 0.43	7.00 ± 0.44		Compost: 0.43 Shrub: 0.04	0.62 0.96
Water holding capacity	31.17 ± 0.84	34.97 ± 0.85	33.11 ± 0.97	32.88 ± 0.87	33.45 ± 0.92	32.54 ± 0.92		Compost: 3.80	<0.05

Results of soil analyses from 2011, one year after plot treatments went into effect. Table presents mean ( $\pm$  SE) values of surface (0–8 cm) soil properties per treatment. MCMC coefficients and P values are presented for the fixed effects retained in the best-fit statistical models. Compost treatment had the most marked effects on soil properties, as illustrated by responses of microbial biomass carbon, pH and water holding capacity. Fixed effects, MCMC coefficients and P values for the best-fit statistical models for bulk density, microbial available carbon and carbon concentrations are also shown, though none of the treatments were statistically significant.

#### 4. Discussion

The starting conditions of urban soils and how they respond to restoration will affect the likelihood that afforested sites support healthy stands of trees and hence provide the intended benefits of afforestation initiatives. Our project assessed the effects of site preparation (measured as changes in soil parameters from 2009 to 2011) and management strategies (measured as the impacts of plot treatments on soil parameters in 2011) on soil health. Soil health is a catch-all term and we measured specific physical, biological, and chemical properties of soils that are essential for vigorous tree growth and resilience to disturbances such as drought. We measured bulk density and water holding capacity as indicators of soils' ability to reduce storm water surges and increase soil water infiltration and aeration, necessary for both microbial and plant health; pH and soil carbon as indicators of suitable conditions for plant growth and nutrient availability; and microbial biomass and microbial available carbon because microbes perform soil processes important for plant growth such as organic matter decomposition and nutrient transformation.

Our results indicate a strong response of soils to site preparation. In managed cropland systems, heavy tillage and ripping can have destructive effects on soil structure and soil biology (Lal, 2011). At our study site, however, tilling for de-compaction coupled with the addition of organic matter (i.e., compost and mulch) had beneficial effects on soil physical, chemical and biological properties. For instance, bulk density values of the research plots now more closely resemble those of the forested reference areas ( $\sim 0.8 \text{ g cm}^{-3}$ ), and are well below the threshold of  $<1.60 \text{ g cm}^{-3}$  where plant growth is restricted in sandy loam soils (Hazelton and Murphy, 2007). Microbial biomass has a short turn-over time and is very sensitive to soil environmental conditions and disturbances, making it a useful indicator for diagnosing changes in soil nutrient dynamics (Kallenbach and Grandy, 2011). We observed increases in microbial biomass and microbial-available carbon following site preparation, indicating that soils are likely improving in functions such as nutrient cycling and transformations that will aid tree growth into the future.

Optimal pH ranges vary for the six tree species planted at our site, with *Quercus* species and *T. americana* preferring more acidic soils (pH range of 4.5–5.5) and *P. serotina*, *Carya* species and *C. occidentalis* preferring more neutral soils (range 6–7) (Burns et al., 1990). Site preparation did acidify the soils, with mean pH values decreasing from 7.37 in 2009 to 7.03 in 2011, suggesting that pH conditions are approaching those considered optimal for at least some of the plant species. In contrast to the pH, microbial and bulk density values, site preparation effects on water-holding capacities and carbon concentrations peaked in 2010. With its water retention properties and high carbon concentrations, the 2010 mulch addition likely explained the 2010 peak values. Indeed, the mulch application was intended to retain moisture and hence reduce water stress on the new plantings and was composed of hardwood material, which presumably was  $\sim 50\%$  carbon by mass. Although values decreased from 2010 to 2011, there was still an overall net increase in carbon concentrations and water holding capacities from 2009 to 2011. Future samplings are required to assess whether the net positive (for tree growth) temporal trajectories in all of the soil properties continue beyond 2011. Nevertheless, our observations suggest that soil preparation for afforestation has only positive initial effects on the health of urban soils.

The idea of a restoration endpoint is challenging in many urban systems because urban soils commonly have no natural analogues in soil type or vegetative cover (as in our study), and so restoring to a natural reference is not possible (Pavao-Zuckerman, 2008). This

lack of an endpoint makes it difficult to disentangle the effects of restoration efforts on ecosystem properties from effects that arise through other factors, such as climate variability. To overcome this limitation we sampled soils from reference-forested areas adjacent to our research plots. Five of our six soil parameters did not change significantly between 2009 and 2011, in contrast to the same parameters in our afforestation plots (Table 1). Only microbial-available carbon significantly changed from 2009 to 2011 in the reference areas, but the measured gain in this variable was much less marked (1.3- compared to 4-times increase) than in our research plots. Overall, then, the responses of our six measured parameters seemed to result primarily from the site preparation activities. Our use of reference plots to make these inferences demonstrates the importance of establishing baseline soil characteristics and associated reference areas if one is to reliably infer the temporal effects of urban restoration on soil health.

Urban soils and vegetation are important stores of carbon, and may help offset local and regional CO<sub>2</sub> emissions (Poudyal et al., 2011; Raciti et al., 2012). Indeed, cities are investing large sums into urban afforestation as a way to address climate change (Oldfield et al., 2013). Carbon accumulation is then a key response variable from both an ecological and policy perspective, and so we measured soil carbon stocks following site preparation to establish a baseline for future stock estimations. We used a mass-dependent approach to assess soil carbon stocks in 2009 and 2010 given the expectation that marked changes in bulk density would make the more traditional depth-dependent assessment methods unsuitable (Gifford and Roderick, 2003). For example, the reductions in bulk density that we observed means that less mass of soil was sampled from the 0 to 8 cm surface sample in 2010 than 2009, which could lead to the conclusion (using depth approaches) that site preparation caused reductions in carbon stocks. In contrast, the mass-dependent approach revealed that site preparation increased carbon stocks by 1.6-times from 2009 to 2010. Future work is required to assess whether these stock increases persist. Notably, between 2010 and 2011 surface soil carbon concentrations decreased (Table 1), suggesting that afforestation was associated with decomposition rates exceeding plant carbon inputs. If this inference were accurate then we would expect declines in carbon stocks from a potential high in 2010 (likely caused by mulching), matching observations of initial declines in carbon stocks following afforestation in non-urban systems (Wellock et al., 2011; Berthrong et al., 2012).

Compost amendments (i.e., the incorporation of organic material into soils) are widely used in restoration projects to aid plant growth; but compost effects on physical, chemical and biological soil properties are often not assessed (Callaham et al., 2008; Heneghan et al., 2008). Addition of organic matter is expected to improve soil health (Cogger, 2005) and, indeed, we observed significant compost effects on three of our six soil parameters in our 2011 sampling of the plot treatments (Table 2). Specifically, compost increased microbial biomass by ~1.2-times, acidified the soil to bring the pH within the optimal range for four of our six planted species (pH 7.37 versus 6.66 in amended plots), and raised water-holding capacities from ~31% to 35% (Table 2). These soil changes would be expected to improve conditions for tree growth by enhancing nutrient and water availabilities.

We expected to see only minimal effects of the planting treatments (i.e., diversity and shrubs) on the soil parameters given that there was only a year from when trees and shrubs were planted to when we sampled the soils. Diversity was, however, retained in the best-fit LMMs for three of the soil parameters (microbial available carbon, microbial biomass and pH) and shrub was retained in the best-fit model for carbon concentrations (Table 2). Although neither of the treatments had statistically significant effects in these best-fit models, their inclusion does suggest they are having some

influence on soil properties, and so they might eventually be expected to affect tree performance and afforestation success through modulation of soil properties.

In line with our findings that the plot treatments were, by in large, having relatively minor effects on soil properties after a single year of application, we did not find any plot treatment effects on tree vigor or survival. Nevertheless, the very low mortality rate of planted trees (~2%) suggests that the pronounced temporal responses of our six soil parameters in response to site preparation were effective at improving soil conditions for tree establishment.

Our findings inform expectations for how human-created soils respond to site preparation for urban afforestation. It will be many years before the trees start providing the environmental benefits associated with mature individuals, but it is likely that the starting conditions of urban soils are essential determinants of whether healthy stands of mature trees establish and hence whether the projects provide the intended ecosystem services. Our data demonstrate that site preparation through tilling, debris removal, compost amendment, mulching and initial planting improves soil structure and functioning. The very low mortality rate among planted trees indicates that site preparation – and the associated changes in soil physical, chemical and biological parameters – likely played an important role in creating suitable soil conditions for tree establishment. Our data on the responses of soils to site preparation can help inform the likely success of urban forestry initiatives to support the ecosystem services they are planted for. The long-term benefits of afforestation in New York and other cities remain to be seen, but our initial findings suggest that urban soil properties respond to restoration efforts for afforestation in a manner that will support future tree growth and so potentially the health of urban populations.

## Acknowledgements

We thank the New York City Department of Parks and Recreation for implementing the project and facilitating the research; the Soil Science class of 2010 at the Yale School of Forestry and Environmental Studies and Yoshiki Harada for soil sampling and processing; and Robert Warren for guidance with statistical models.

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