ABSTRACT

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Elements of urban soil quality such as compaction and low organic matter are underexamined, important challenges to urban afforestation. In this Beltsville, Maryland field experiment, I examined biosolids and compost as amendments to improve soil quality and planted tree survival in a degraded urban proxy soil and identified correlations between soil properties and tree survival. Organic amendments increased organic matter content, decreased bulk density, and had no effect on tree survivorship compared to controls. Effects on soil were more profound and lasting with compost than with biosolids. Soil organic matter and bulk density were correlated with tree survival early in the study and microbial respiration was correlated with tree survival throughout the study. High tree mortality was driven by transplant shock, limiting insights from tree response data. This study highlights the importance of soil quality and good planting practices in future research.

BIOSOLIDS AND COMPOST FOR URBAN SOIL RESTORATION AND FORESTRY

by

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Dedication

I dedicate this thesis to my family, especially to my grandmother and reading partner, Gayle; you are deeply missed.

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Chapter 1: Introduction

1.1: Overview

Vegetated areas such as urban forests provide ecosystem services (or environmental benefits) in cities (Gomez-Baggethun et al., 2013). Because of the services provided by urban forests, the establishment and protection of urban forests is a crucial component of addressing urban problems related to climate change, environmental health, and stormwater management (Chen et al., 2014; Phillips et al., 2019; Davies et al., 2017). Urban land use is subject to complex environmental, social, and economic influences, resulting in highly dynamic and heterogeneous urban landscapes that pose unique challenges for efforts to establish forests and other greenspace in cities (Johnson et al., 2020; Herrmann et al., 2020). The growing presence of vacant land in many cities offers an opportunity for the establishment of urban forest (Barkasi et al., 2012; Herrmann et al., 2017; Kelleher et al., 2020; Beniston et al., 2015). However, a major challenge accompanies this opportunity: soil quality in urban vacant land (Pavao-Zuckerman and Pouyat, 2017; Craul, 1985).

There is promise for overcoming soil quality challenges using organic soil amendments (Larney and Angers, 2012; Chen et al., 2014; Scharenbroch and Watson, 2014; Layman et al., 2016; Scharenbroch et al., 2009; Basta et al., 2016; Scharenbroch et al., 2013; Oldfield et al., 2015; Olson et al., 2013; Lopez et al., 2008; Beniston et al., 2015). Here I define soil quality as the ability of soil to perform functions including water retention and infiltration, physical stability, nutrient cycling, and sustaining biological activity and diversity (Scharenbroch and Catania, 2012). In this thesis work, I explored the potential of soil management with organic amendments for improving urban soil quality and reducing tree mortality after planting. In the following sections, I will outline current scientific knowledge on soil quality challenges to urban forestry and the mechanisms through which organic matter additions have been observed to improve quality and function in poor-quality soils. I will then describe a field study in which I explore the effects of two organic soil amendments—biosolids and compost—on soil quality and transplanted tree survival at a Maryland site with a history of construction, demolition, and frequent heavy vehicle traffic.

1.2: Urban Forestry Goals

Definitions of urban forests have traditionally referred to all tree canopy in an urban area, including street trees and forested natural areas. Pregitzer et al. (2019[a]) define forested natural areas as urban forest spaces that "often occur in parkland, are managed at the stand level, and are maintained by natural processes such as regeneration (p. 1)"; these areas are spatially arranged in patches in the urban landscape. The scope of this thesis focuses on informing soil management for tree planting activities aimed towards establishing urban forested natural areas.

Urban forests provide a myriad of benefits to cities through ecosystem services such as carbon (C) capture, air pollution mitigation, and local climate benefits that allow for energy savings; conservative estimates suggest these benefits save cities upwards of $18 billion per year (Nowak and Greenfield, 2018[b]). Recently, urban tree cover has been declining in the U.S. (Nowak and Greenfield, 2018[a]), and cities are working to increase tree cover through urban afforestation and reforestation practices. Urban afforestation is the activity of planting of trees in the urban environment with the goal of establishing new urban forest while reforestation is the activity of reestablishing forest in previously forested areas (Moll and Ebenreck, 1989; Hallett, 2013). In many cities, urban reforestation and afforestation goals are motivated largely by climate change mitigation and adaptation benefits; urban trees help regulate local climate through local cooling effects and mitigate climate change by sequestering C (Smith et al., 2020). For example, the state of Maryland has a goal to plant 5 million new trees over the next decade to mitigate climate change and associated sea level rise and shore erosion in the state and is designating 500,000 of these tree plantings to urban areas (Thomas, 2021). In New York City (NYC) the cool neighborhoods initiative is a movement that is planting trees with the goal of improving local climatic conditions and local air quality, targeting areas for tree planting based on communities' degrees of heat vulnerability. This vulnerability is often quantified using social data such as healthcare access and race (Smith et al., 2020). Environmental equity in cities is another common motivation for urban tree planting initiatives. For example, tree planting initiatives in Baltimore, Maryland, seek to improve the city's tree equity score (treeequityscore.org) by targeting planting projects toward low-income and minority communities with lower canopy cover (Shwe, 2021).

The desired end product of many urban forest restoration and afforestation goals is a self-sustaining ecosystem composed predominantly of native vegetation (Smith et al., 2020), such as the regeneration of oak-dominated forest ecosystems in Baltimore, MD (Sonti et al., 2021). Another common afforestation objective is to successfully establish young, newly planted trees and achieve quick canopy closure in hopes to shade out competing and invasive plant species (Freshkills Park Alliance). Successful seedling establishment is important as cities are increasingly investing resources into urban forest preservation and restoration efforts (Sonti et al., 2021).

Forest establishment activities often occur on large land restoration scales, and soil restoration actions are often a necessary component of planning. For example, most trees plantings in the NYC Million Trees Initiative are focused on restoring degraded or destroyed ecosystems over large restoration sites (Simmons et al., 2016). One such large scale project is the ongoing restoration of the Freshkills landfill site in NYC, where soil restoration activities and mass tree planting are major components of converting the highly human-impacted 2,200+ acre site into what is projected to be NYC's second largest park (New York City Department of Parks and Recreation; Smith et al., 2020). As another example, Hantz woodlands—the United States' largest urban tree farm located in east Detroit, Michigan—was established over contiguous former vacant lots, where the organization has planted more than 36,000 trees over an area of 2,000 parcels (Hantz Farms Detroit). Vacant lots from abandoned building demolition are also likely to be used as tree planting sites to meet tree equity goals in West Baltimore, MD (Shwe, 2021). In these afforestation efforts and others, restoration of soil that has been heavily degraded through human activities such as construction and addition of fill materials is a major component of meeting objectives to establish and maintain forested natural areas in cities (Freshkills Park Alliance; Simmons et al., 2016).

1.3: The Soil Challenge for Urban Forestry

Despite interest in urban afforestation and reforestation practices (Davies et al., 2017; Oldfield et al., 2013), high tree mortality shortly after establishment remains a challenge in and around cities, leading to neutral or even negative ecosystem service returns on resource investments in forest establishment efforts (Nowak et al., 1990; Roman et al., 2016; Oldfield et al., 2013). Because urban landscapes are heavily dominated by human land development, urban environmental conditions pose challenges for efforts to increase forest cover in cities (Roman et al., 2016). Common urban conditions that inhibit successful tree establishment include lower access to light, poor tree species selection, lack of proper tree maintenance, poor water quality, and poor soil conditions for supporting plant life (Beatty and Heckman, 1981; Hilbert et al., 2019).

Though the increasing availability of vacant urban land from the demolition of vacant buildings in many cities (Popper and Popper, 2010) provides potential site opportunities for establishing urban forested natural areas, urban soil quality in many of these sites remains a major challenge that needs to be addressed. De Kimpe and Morel (2000) define urban soils as "soils that are under strong human influence in the urban and suburban landscape (p. 32)." Drastic soil disturbances from human influence often include topsoil removal, excavation of native soil, addition of fill material, building demolition, and heavy compaction from foot and vehicle traffic (Pouyat et al., 2010; De Kimpe and Morel, 2000). Construction and demolition on urban vacant lots frequently results in heavily compacted soils with a high presence of coarse material from demolition debris (Pavao-Zuckerman and Pouyat, 2017; Shuster et al., 2014). These activities also tend to result in low soil fertility and organic matter content (Pouyat and Trammell, 2019; Pouyat et al., 2010; Pouyat, 2001) compared to northeastern U.S. natural forest counterparts, where soil organic matter content in the upper 20 cm is often as high as 11.9% (Zukswert et al., 2021). Topsoil erosion from urban runoff—along with land management activities like grass clipping and leaf litter removal—also contributes to lower organic matter content and quality in urban soils.

The impaired physical, chemical, and biological components of soil quality common in urban vacant lots pose challenges for the successful establishment and survival of newly planted trees (Pregitzer et al., 2016; Day and Bassuk, 1994; Xiao and McPherson 2011; Grabosky and Bassuk 2016; Scharenbroch et al. 2017). Root growth can be limited by heavily compacted soil conditions in cities (Kozlowski, 1999; Day et al., 2010[b]). Soils with higher levels of compaction have lower total pore space and macro pore space, resulting in lower soil water infiltration and storage and lower soil aeration (Kozlowski, 1999). The decreased air and water availability for roots in soil is a direct driver of plant stress (Weil and Brady, 2017). Excessive levels of compaction also limit root exploration, leading to lower root-to-soil contact area which further exacerbates tree stress because of the reduced access to water, air, and nutrients in the soil (Weil and Brady, 2017; Day et al., 2010[a]; Kolzowski, 1999). High levels of soil compaction and low soil organic matter content are especially common challenges to tree survival in cities (Pouyat and Trammell, 2019; De Kimpe and Morel, 2000; Kozlowski, 1999; Pouyat et al., 2007).

1.4: Organic Amendments as a Solution for Urban Soil Quality and Afforestation

1.4.1: Soil Restoration for Urban Forestry

Soil quality management has become an urban forestry research priority because of the challenges urban soil conditions pose to afforestation efforts (Layman et al., 2016; Scharenbroch and Catania, 2012; Pouyat and Trammell, 2019). Current common practices in urban forestry for creating a healthier soil environment include mulching, tillage, and additions of organic matter (Layman et al., 2016), but urban tree survival remains limited by factors such as poor species selection, vandalism, inadequate maintenance, and site soil quality (Hilbert et al., 2019; Roman et al., 2016). Among the major site characteristics associated with high urban tree mortality, soil quality is an aspect that has remained underexamined (Hilbert et al., 2019). Understanding the role of soil quality in urban afforestation and exploring solutions for soil restoration are crucial for informing management and meeting urban afforestation goals.

Soil restoration is defined as reversing negative impacts of soil degradation on soil quality and function through site management (Lal, 2015). Goals of soil restoration include aspects of physical quality (minimizing erosion and improving soil stability and pore geometry), biological quality (increasing the activity and diversity of soil biota), and chemical quality (improving soil organic C budgets and soil fertility). This can be achieved through multiple site-specific approaches such as nutrient management, conservation agriculture, and erosion prevention (Lal, 2015). Organic soil amendments show promise as a solution for jumpstarting soil quality restoration and providing long-term improvements in primary productivity (Larney and Angers, 2012).

Though the reuse of organic matter for land application is not a new practice and has been researched in agricultural and non-urban forestry applications, there is still limited research on the use of organic matter additions for urban soil restoration, including research to inform recommendations for organic material selection and application rates (EPA OSRTI, 2007; Scharenbroch et al., 2013; Basta et al., 2016). The USEPA (2000) reports typical forestry application rates for biosolids range from 11 to 224 Mg/ha. An urban study by Scharenbroch et al. (2013) using organic amendments in degraded urban soils to improve conditions for tree seedlings used a rate of 25 Mg/ha for both biosolids and compost, following recommendations from the USEPA (2000), US Composting Council (2001) and USDA NRCS (2011).

Soil quality benefits observed from applications of organic amendments in degraded urban soils include long-term increases in plant available nitrogen (N) and phosphorus (P), microbial respiration, and soil structure (López et al., 2008), improved food web structure (Cheng and Grewal, 2009), improvements in soil available N, N mineralization, microbial respiration, and total organic C (Scharenbroch et al., 2013) decreased soil bulk density (Scharenbroch et al., 2014), improved temperature buffering, improved soil hydrological function and increased porosity (Scharenbroch, 2009), and improved soil pH and cation exchange capacity (Guerrini et al., 2017). Organic amendments also increase tree root and shoot growth (Scharenbroch and Watson, 2014; Guerrini et al., 2017; Scharenbroch, 2009), biomass production (Basta et al., 2016), tree canopy cover, and soil/plant system C sequestration rates (Layman et al., 2016).

Though these studies provide an early foundation for our knowledge of organic amendment use in urban soil restoration and afforestation, there are still gaps in our understanding due to conflicting results among studies and a lack of standardized amendment uses. For example, though most studies report improvements in soil quality and tree responses in amended urban soils, studies can report neutral or negative soil and tree responses as well (Scharenbroch, 2009). Additionally, the types of organic amendments available are very diverse in their composition and in the soil quality benefits they can offer (Larney and Angers, 2012). As the practice of organic matter applications becomes more commonplace in urban soil management and afforestation, empirical knowledge of different amendment types, their effects on soil, and their effects in urban forestry contexts is needed to inform soil management for forest establishment efforts.

1.4.2: The Role of Soil Organic Matter

Organic matter plays many important roles in soil, and low soil organic matter content is a common predictor of significant soil quality deficits and low plant-supporting function (Gregorich et al., 1994; Brenner et al., 1984). Soil organic matter supports chemical soil quality by increasing pH buffering capacity, by serving as a storehouse of nutrients to be mineralized, and by increasing N fixation and mineralization as well as organic C turnover via microbial activity (Weil and Brady, 2017; Blanco-Canqui et al., 2013). Physical soil quality is improved by the presence of coarser organic matter, which loosens soil and increases macropore space, increasing soil aeration and water infiltration (Weil and Brady, 2017). Soil biological quality is improved by finer, more decomposed organic material that provides substrate and habitat for soil organisms and increases soil biodiversity and microbial biomass and activity (Weil and Brady 2017). Increases in microbial biomass and activity from applications of organic amendments can largely be attributed to the supply of readily decomposable C in the added organic material (Werner and Dindal, 1989; Ros et al., 2003; Mabuhay et al., 2006; Belyaeva and Haynes, 2009). Microbial activity also produces organic byproducts that bind together soil mineral and organic matter particles, helping to form larger and more stable soil aggregates and improving physical soil structure and macropore space (Blanco-Canqui et al., 2013; Angers and Chenu, 1997; Chatigny et al., 1999; Sere et al., 2008).

Physical, chemical, and biological improvements to soil quality improve conditions for plant roots such as root aeration and water availability (Weil and Brady, 2017; Kozlowski, 1999). In addition to soil physical benefits from the formation of soil aggregates, the integration of organic material can reduce dry soil bulk density through lower material density of the organic amendment itself, decreasing the compaction that limits plant root exploration (Larney and Angers, 2012). Because soil organic matter is deeply integral to soil quality, improvements in soil physical, chemical, and biological quality from organic matter additions are all interrelated and can be seen even with small changes in total soil organic matter content (Blanco-Canqui et al., 2013; Weil and Brady, 2017). Therefore, organic matter additions may hold the key to a comprehensive management solution for soil-related urban forestry challenges (Pouyat et al., 2008; Larney and Angers, 2012).

1.4.3: Soil Restoration Goals and Amendment Selection

To strategically address soil quality challenges to urban forestry using organic amendments, we must first identify soil management goals to guide selection of organic materials. Heneghan et al. (2008) suggest that land restoration with more sophisticated goals—such as restoring an ecosystem to a specific historic ecological state—requires a more complex management approach that carefully considers interactions between soil physical, chemical, and biological conditions as well as interactions between plants and the soil environment. For simpler restoration goals aimed at reclaiming a specific soil process—such as supporting revegetation—a more simple, single-factor manipulation of a soil physical, chemical, or biological attribute can be very effective (Heneghan et al., 2008). For afforestation on degraded soil in vacant lots (the intended application for this thesis study), the soil restoration goal centers on the simpler objective of restoring a soil's plant-supporting function in order to increase planted tree survival. Therefore, a simpler management approach that simultaneously targets compaction and low organic matter—two soil conditions that are especially limiting to tree establishment and growth (Kozlowski et al., 1999; Six et al., 2004)—through addition of organic material and tillage may be appropriate.

Though the benefits of organic amendments have potential to improve urban tree survival by removing or reducing soil quality barriers, a remaining challenge is the strategic selection of organic amendments among the many types available. Here I will provide a description of biosolids and compost, the amendments selected for this study that have promise for increasing soil organic matter content and decreasing compaction to improve soil conditions for planted tree survival. The rationale for selection of these amendments will be further discussed in later sections (1.4.6 and 2.2.1).

1.4.4: Biosolids Overview

Biosolids are organic materials produced from the solid byproducts of wastewater treatment. The term "biosolids" is used to differentiate them from raw sewage sludge, as biosolids have undergone treatment to remove pathogens (Sullivan et al., 2015). Water treatment processes often include the removal of nutrients such as N and P to mitigate nutrient pollution in water bodies, so biosolids are characteristically high in these nutrients (Weil and Brady, 2017). Biosolids are also typically high in other plant macronutrients and micronutrients (Sullivan et al., 2015). Much of the macronutrient supply in biosolids is tied up in organic material and continues to be released over long periods of time, making biosolids a promising amendment for improving soil fertility in degraded urban soils long-term (Lu et al., 2012). Biosolids application also has the potential to decrease soil bulk density and penetration resistance (Larney and Angers, 2012).

Biosolids have been used in agricultural as well as many non-urban forestry contexts (Scharenbroch, 2009; Lu et al., 2012). Though these practices have met some criticism over concerns that biosolids can pose health hazards, heavy metals and other contamination have been drastically reduced in recent decades, and contamination only begins to pose risks at application rates exceeding 100 dry tons per hectare (Lu et al., 2012). The use of biosolids as a soil amendment in forestry still offers one of the most promising solutions for addressing soil quality challenges, with the added advantage of reducing pressure on landfills from water treatment byproducts (USEPA, 2000; Larney and Angers, 2012; Kumar et al., 2017; Chu et al., 2017). Studies have begun to explore the use of biosolids for urban soil remediation and tree planting purposes, but most are pot or greenhouse studies, and there is a need to explore their effects in field contexts as well (Basta et al., 2016; Scharenbroch et al., 2013; Chu et al., 2017; Scharenbroch and Watson, 2014)

1.4.5: Compost Overview

Compost is one of the more commonly used organic amendments in urban soil management for tree planting, though it can also be one of the more costly organic amendments available (EPA OSRTI, 2007; US Composting Council, 2001). Compost can be produced both commercially and at the household scale, and types of compost vary widely in their source materials and properties. Compost is produced via conversion of organic materials into stable soil conditioners by microbial organisms (US Composting Council, 2001). It is characteristically high in C and organic matter and offers soil quality benefits including improved soil physical structure, improved water infiltration and storage, and improved cation exchange capacity and pH stabilization (Amlinger et al., 2007).

Though compost can come from a variety of source materials such as food waste, yard waste, and animal byproducts, it is consistent across its varieties in its general benefits of improving soil structure as a soil conditioner and serving as a rich source of organic matter that can be added in higher quantities without much concern for excess nutrient runoff pollution; an advantage of compost as an amendment compared to biosolids (EPA OSRTI, 2007). These overall benefits are well-suited to the soil restoration goals in this study of reducing compaction and increasing organic matter content and soil biological activity.

1.4.6: Biosolids and Compost for Urban Soil Restoration and Forestry

Compost and biosolids have unique benefits to soil and their use in urban restoration may be a powerful solution to mitigating common soil-related causes of early tree mortality. Specifically, the nutrient release benefits provided by biosolids, the soil physical benefits provided by compost, and the organic matter benefits provided by both amendments may aid in mitigating soil compaction and low organic matter content, reducing the water and nutrient stress that newly planted tree mortality in cities is often attributed to (Nowak et al., 1990). Exploring the application of these organic amendments in urban soil restoration and forestry will improve our understanding of the benefits that these amendments provide to soil quality and tree survival and further our knowledge of the relationship between soil quality and tree transplant survival to inform urban land management.

1.5: Study Objectives

The goal of this project was to evaluate the use of compost and biosolids—both individually and in combination—for soil restoration and tree planting in urban field settings. The first objective of this work was to understand the effects of compost and biosolid additions on soil quality (organic matter content, compaction, and biological activity). The second objective of this work was to understand the effects of these soil amendments on planted tree survival. The third objective of this study was to identify any predictors of planted tree survival among measured physical, chemical, and biological soil quality characteristics. To accomplish this, I conducted a field study on a demolition-disturbed site where I applied soil treatments of biosolids and compost—both individually and combined—before planting seedlings of white oak (*Q. alba*), and then monitored soil and tree responses. Soil physical, chemical, and biological responses examined included bulk density, penetration resistance, organic matter content, microbial respiration rates, moisture content, temperature, total C and N, pH, and electrical conductivity. Tree responses observed included mortality, cambium color, leaf color, leaf drop, tree height, and post-harvest dry shoot and root biomass.

I posed the following research questions:

- Does the addition of compost and biosolids to soil—both individually and in combination—affect measured physical, chemical, and biological soil characteristics compared to untreated controls?
- Does the addition of compost and biosolids to soil—both individually and in combination—affect newly planted tree survival, physiological responses, and biomass compared to untreated controls?
- Which measured soil characteristics best serve as predictors of newly planted tree survival?

I hypothesized that soils amended with biosolids, compost, and both amendments together would have lower levels of soil compaction (indicated by bulk density and penetration resistance) as well as higher soil organic matter content, total C and N, moisture, and microbial respiration rates compared to unamended controls. This prediction is based on previous research suggesting that additions of organic material to soil often result in these soil quality changes (Lopez et al., 2008; Cheng and Grewal, 2009; Scharenbroch et al., 2013; Scharenbroch et al., 2014; Scharenbroch, 2009; Guerrini et al., 2017). While I expected both biosolids and compost to benefit soil microbial activity, I expected the biosolids treatment to have a stronger effect on nutrient availability than on compaction and water retention, and I expected the compost treatment to have a stronger effect on compaction and water retention than on nutrient availability.

I expected tree survival to be greater in plots receiving organic amendments than in unamended controls. This prediction is based on the changes I expected to see in the root environment (increased aeration, increased water availability, increased nutrient availability, and decreased compaction) following the application of organic amendments. I expected to see the highest tree survival rates in plots treated with compost and biosolids combined because I expected the combination of their strengths as a soil conditioner (compost) and a nutrient source (biosolids) to address a greater variety of soil-related plant stress sources than the other treatments in the study.

Finally, I hypothesized that soil physical compaction (bulk density and penetration resistance), organic matter content, and microbial respiration rates would be the strongest predictors of tree survival because of their direct influence on leading causes of root stress: soil compaction, aeration, organic matter content, and nutrient availability (Kozlowski et al., 1999; Gregorich et al., 1994; Brenner et al., 1984).

Chapter 2: Materials and Methods

2.1: Site Selection and Description

This study was conducted at the Central Maryland Research and Education Center - Beltsville Facility (39°0.8675’N, 76°49.5525’W). The study site is the former location of a dairy barn and surrounding pavement. According to Google Earth image records, the road and building were demolished between 1993 and 2002. I established a 12 m by 21 m (east to west by north to south) study plot on the site that overlaps the former building and contiguous parking area (Figure 2.1).

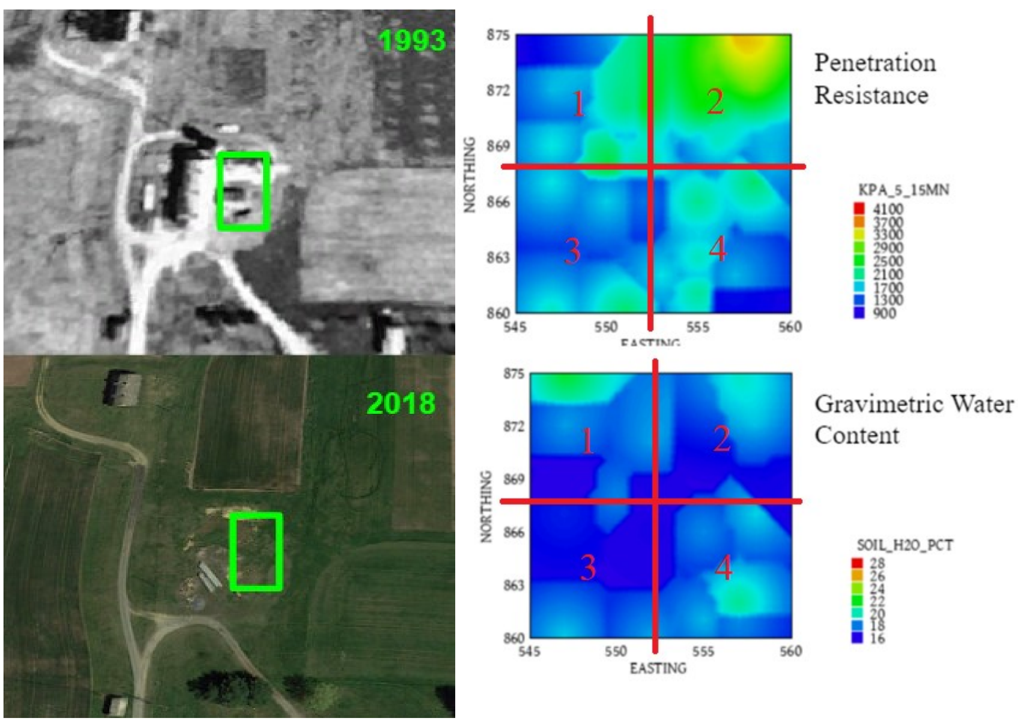

Figure 2.1: (left) Google Earth images of the study site at the in Beltsville, MD. The land use history and resulting soil conditions at this site allow it to serve as a proxy for disturbed urban soil conditions, such as those found in vacant urban lots; (right) Kriged images of penetration resistance (kPa) and gravimetric water content (g water/g dry soil) data from the study site with overlaying lines showing quadrants to be used as replication blocks in the Beltsville, MD, field experiment.

Historic construction and demolition and frequent heavy vehicle traffic have resulted in levels of compaction frequently above 2,000 kPa (sampled with cone penetrometer), coarse material content frequently upwards of 25% (separated with No.4 sieve), and organic matter content frequently below 6% (determined by loss on ignition) within the upper 20 cm of the soil profile. These values are similar to those from vacant lot soils in cities. Diaz-Sanz et al. (2020) found soil penetration resistance greater than 2,000 kPa in the upper 15 cm of most urban sites sampled throughout Marseilles, France. Pouyat et al. (2007) found an average soil organic matter content of 5.4% (standard error of 1.7%) throughout various urban sites sampled in Baltimore, MD. Shuster et al. (2014) found an average of 54% rock and debris fragments in urban soils sampled throughout Cleveland, OH and Nehls et al. (2012) found that brick coarse fragments alone represented 23% of the total soil mass in urban garden soil in Berlin, Germany. Because the levels of soil compaction, organic matter content, and coarse material found in the Beltsville site are similar to those found in cities, the site serves as a useful proxy for an urban vacant lot, creating an opportunity to understand the effects of organic amendment integration on soil quality and function in a heavily degraded and compacted soil. It is worth noting that other common elements of an urban vacant lot environment (i.e., chemical inputs from ambient air deposition and hydrological sources, urban heat island, invasive species) are not present at this site. However, this more simplified environment allows the study to focus on the soil conditions most relevant to my research questions (soil physical conditions and low organic matter content) with the added advantages of site security, facility staff, and land management resources that come with working at the Beltsville facility.

To understand the variability in compaction levels across the site, I collected penetration resistance data with a Spectrum FieldScout SC900 (Aurora, IL) cone penetrometer (Lowery and Morrison, 2002; Dayal and Allen, 1973). Sampling was conducted in a grid pattern across the study area at the intersections of 5 equidistant transects running north to south and 6 transects running east to west. Figure 2.1 reports averages of recordings taken at 2.5 cm depth intervals from 5 to 15 cm below the surface and shows the highest levels of compaction within the northeast quadrant of the study area, lower compaction levels in the southwest quadrant, and more moderate levels of compaction in the remaining northwest and southeast quadrants.

Preliminary soil fertility analysis (conducted by the Penn State Agricultural Analytical Services Laboratory, University Park, PA) of four composite soil samples collected from the northwest, northeast, southwest, and southeast sections of the study site showed pH ranging from 7.4 to 7.7 (1:1 soil:water pH), cation exchange capacity ranging from 8.0 to 10.2 (summation of cations), and exchangeable cations of ~0.2 meq/100 g for potassium (Mehlich 3 ICP), 0.8 – 1.3 meq/100 g for magnesium, 7.0 – 8.7 meq/100 g for calcium (Mehlich 3 ICP), and 44.8 – 91.84 kg/ha for P (potash and phosphate) (Mehlich 3 ICP). I used the Bouyoucos method for particle size analysis (Bouyoucos, 1964) with 17 hours for silt particle settling to assess site soil collected (to a depth of 20 cm) prior to the treatment application at 4 randomly selected locations per quadrant (northeast, northwest, southeast and southwest). Site soils are classified as sandy loams (with the exception of one sample classified as loamy sand), with sand content ranging from 62 to 83%, silt content ranging from 13 to 28%, and clay content ranging from 5 to 11%.

2.2: Treatments

2.2.1: Selection of Organic Amendments

In remediation contexts where soil quality is significantly impaired, biochemical stability of the organic material is important to consider as introducing unstable organic material can have negative effects on a soil's ability to support plant life, such as microbial immobilization of available soil N (Larney and Angers, 2012). In considering long-term plant N and other nutrient needs, it is also preferable to select for organic amendments that provide lasting slow-release nutrient benefits (Larney and Angers, 2012). Claassen and Carey (2007) suggest a combined use of both recalcitrant, lignin-rich amendments and bioavailable, nutrient-rich amendments for remediation goals where availability of N is necessary to support plant life, but N also needs to be controlled for reasons such as preventing weed growth or N runoff pollution. In this study, I decided to examine the benefits of a nutrient-rich and bioavailable organic amendment (biosolids) and a recalcitrant, lignin-rich organic amendment (compost) used both individually and in combination for restoring function and improving planted tree survival in a degraded soil heavily impacted by human activity. I chose these amendments because their characteristic soil quality benefits pair well with the main urban soil quality challenges to forestry: high compaction and low organic matter content.

Amendment selection is also influenced by the cost and availability of materials; even with more context-specific information to improve practice in the future, the organic amendment materials that would be considered ideal for management may not be cost-effective or available for use. For this study, I used amendments that have already been implemented in soil management for plantings in the greater Washington D.C. and Baltimore area, where the study occurred and where potential vacant lot tree planting sites are in abundance (especially in Baltimore). The specific brand of compost I used in this study, Leafgro, was chosen because of its popularity as an amendment for tree plantings in the greater Washington D.C. and Baltimore area. Bloom, the biosolid product chosen for this study, provides the nutrient and soil organic matter benefit I am interested in for the purpose of improving soil organic matter content, microbial activity, and fertility. Bloom is also a locally produced and used organic amendment that is more readily available than compost, which is produced in smaller volumes and can be less readily available (USEPA OSRTA, 2007; bloomsoil.com). Soil and tree response data are not typically quantified over time for urban plantings where amendments are implemented. Through this study, the effectiveness of these amendments for soil management and tree planting in an urban setting can be explored more empirically.

2.2.2: Organic Amendment Properties

Bloom is an anaerobically digested soil amendment produced and distributed by the Blue Plains Advanced Wastewater Treatment Plant in Washington, D.C. (bloomsoil.com). Leafgro is composed of windrow-composted leaf litter and lawn clippings from yard waste in Prince George's County and Montgomery County, Maryland (menv.com/leafgro). Fresh Bloom used in this study was picked up in plastic buckets from the production facility in Washington, D.C. Leafgro was purchased from Home Depot in College Park, MD, in plastic bags. The Fresh Bloom biosolids used in this study have a carbon to nitrogen ratio (C:N) of 7, contain 4.8% N dry weight, have a pH of 8.43, and have an organic matter content of 63.6% (Appendix B). The Leafgro compost has an organic matter content of 70% (Bloom, 2020), a C:N ratio ranging from 16.05 to 17.33 and contains 2% N and 25% C by dry weight, according to LECO analysis of 2 samples (using high temperature combustion 628 LECO C/N Analyzer, model 622-000-100, St. Joseph, MI; Yeomans and Bremner, 1991). Application rates for biosolids and compost were based on the suggested N need for the tree species provided with preliminary soil test results from Penn State Agricultural Analytical Services Laboratory (University Park, PA). The application rate was considerably lower for biosolids than for compost in order to reduce the risk of N runoff pollution from excess biosolids application; federal regulations require that rates for biosolids applied on agricultural lands must not exceed the N needs for the crop or vegetation being grown (40 C.F.R. § 503.11).

2.3: Study Design and Timeline

The five treatments for this experiment included biosolid application, compost application, and a combined application of both biosolids and compost (referred to as mixed) that were all tilled into the upper 10 cm of soil (application rates described below). I also used both tilled and untilled control groups that received no organic amendment applications. Due to the trends in soil compaction described above (Figure 2.1), the four quadrants of the study site were used as four replicate blocks for this study and within each block, all five treatments were assigned randomly to 5 or 6 plots. Plots sized 1.5 m by 1.5 m were laid out in a 4 by 7 grid design in each block to capture the variability in site soil compaction at a finer resolution and achieve better representation of all treatments across the variable degrees of compaction within each of the four replication blocks. Within each block, individual plots were assigned treatments from a pool of 5 to 6 replications of each treatment in the study, totaling 28 for each block (I randomly selected three treatments to have a sixth replication for each of the four blocks prior to this). Figure 2.2 shows the treatment assignments for all 112 plots.

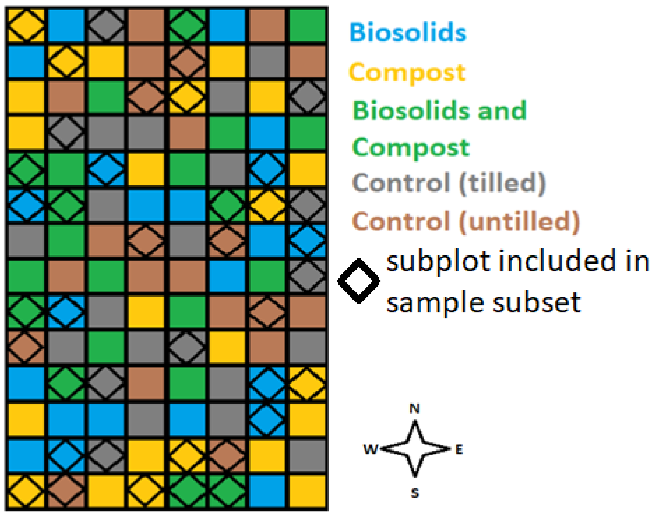

Figure 2.2: Spatial layout of experimental plots and treatment subplots at the Beltsville, MD study site. Each 1.5 m x 1.5 m subplot is colored according to its assigned treatment, and plots included in the sampling subset are marked with a diamond. Soil treatments were randomly assigned within each replication block.

Roundup was sprayed on all plots on September 22, 2018, to aid in preliminary sampling, plot delineation, and weed control prior to the start of the experiment in spring of 2019. The recommended soil pH range for growing *Q. alba* is 4.5 – 6.8 (Slattery et al., 2003). Preliminary sampling showed site soil pH ranged from 7.4 – 7.7. To lower pH, ferrous sulfate was added at 0.75 kg/sq meter to a depth of 20 cm in a 40 cm diameter hole at the center of each plot (where tree seedlings would later be planted) on June 4, 2019, prior to the addition of organic amendment treatments. Because of the higher natural N-release rate of biosolids compared to compost, supplemental N fertilization for non-biosolid treatments was used to supply N in treatments that did not include biosolids. N is one of the most common plant-limiting nutrients (Weil and Brady, 2017) and insufficient plant available N may have strongly limited tree survival and confounded the effects of other soil conditions on tree survival if it were not supplemented. Urea-N was applied to the soil surface beneath the woodchip layer on June 20, 2019, prior to planting, at a rate of 9 g/sq meter for treatments where no biosolids were applied, to best match the 1st year N-release rate of 20% for anaerobically digested biosolids (Weil and Brady, 2017). Amendments were integrated into soil 3 weeks prior to tree planting to allow some time for any immediate soil physical, chemical, and biological responses to treatments (such as physical resettling after tillage, spikes in microbial activity, and changes in the soil chemical environment) to become more stable before tree roots were introduced to the soil environment. This is different from the practice in many urban tree plantings, in which amendments are often integrated into soil at the time of planting. Compost was purchased and top-applied on June 6, 2019, and biosolids were picked up and top-applied to plots on June 7, 2019 (Figure 2.3, left). Compost was applied to plots at a rate of 2.15 kg/sq meter (21.5 Mg/ha) dry weight, biosolids were applied at 0.5 kg/sq meter (5 Mg/ha) dry weight, and for the biosolids + compost (mixed) treatment, the rates of 2.15 kg/sq meter (21.5 Mg/ha) for compost and 0.5 kg/sq meter (5 Mg/ha) for biosolids were used as well. Amendments were tilled into soil to a depth of 10 cm on June 11, 2019, using a gasoline-powered machine roto tiller approximately 56 cm in width (Figure 2.3, right). All treatments and controls were covered in a two-inch layer of aged wood chips (Figure 2.4). The wood chip layer served to prevent weed growth, preserve moisture, and to practice consistency with common urban tree planting methods.

Figure 2.3: (left) Soil amendments top-applied to plots prior to tillage. Untilled control plots are marked with pink tape; (top right) Plots were rototilled following amendment application to a depth of 10cm with a hand-pushed machine rototiller; (bottom right) Treatment plots immediately after tillage at the Beltsville, MD study site.

Figure 2.4: (left) Final experimental setup at a single plot following tillage of amendments, application of wood chips, tree planting, and installation of irrigation tape and tree cages; (right) Overview of the study site after the initial experimental setup was completed in Beltsville, MD.

Q. alba, commonly referred to as white oak, is a native species in Maryland that is suited to sandy loam soils, as are present at the study site. A moist-to-dry soil moisture regime and no-shade to partial-shade light regime are recommended for this species (Slattery et al., 2003). This aligns well with conditions at the Beltsville study site where soils are classified as moderately-well to well drained (though historic compaction at the site may affect this) (USDA-NRCS, 2021), and there is little to no shade. *Q. alba* is a common priority chosen for urban plantings, including in Baltimore, MD (Barron, 2018; City of Seattle, 2013; Lautar and Avins, 2017). The species has been shown to be tolerant to conditions exacerbated by climate change such as drought (Abrams, 2003; Goldblum, 2010) and is known to be resilient to nutrient deficient conditions (Boerner, 1984; Norby et al., 1986). This suggests that *Q. alba* may be well suited for adapting to urban environmental conditions such as degraded soils and elevated local temperatures. Because it is a common species in urban woodlands in the eastern United States (Nowak et al., 2016; Pregitzer et al., 2019[b]) and recommended priority for native species selection in local plantings (Schuster et al., 2008), using *Q. alba* is ideal for this study's goal to inform urban forestry soil management. Trees at the seedling stage were chosen for this study because planting trees at this size and stage allows for a large planting quantity in a shorter period of time; a helpful advantage for larger scale urban forest plantings as well as for the design of this study.

Seedlings of *Q. alba* were obtained from Hort Inc. (Chestertown, MD) on June 28, 2019. To understand the influences of soil treatments used in this study, trees had to be removed from container potting mix to be planted bare root. Trees were extracted from pots on the day of planting. Potting mix was loosened by gently tapping the side of the pot against the ground, taking care not to damage the tree stems or tap root. Once soil was loosened, the taproot and lateral root structure were removed from the container with attached soil. The soil and root ball were massaged by hand to remove excess soil and clumps of fibrous roots that were not directly attached to the taproot and lateral root structure, removing all potting soil as well as majority of fibrous roots that were not directly attached to the tap or lateral roots. Tree roots were then immediately transferred into water after brief photo documentation of tree height and root depth and structure (Figure 2.5). Remaining potting soil was removed by soaking roots in water. Roots were soaked in water for 1 hour prior to planting. Seedling height at the time of planting ranged from 12 to 36 cm. Seedling planting was completed June 30, 2019, and trees were kept in their potting mix on-site overnight while awaiting planting. Tree cages were installed for protection from deer browsing, and drip irrigation tape was laid along the base of all planted trees (Figure 2.4). Drip irrigation of 15 liters per square meter was applied over a course of 12 hours to the study site through irrigation tape running along the base of every tree after any period of more than one week without precipitation. Trees were kept on the study site for two growing seasons and living tree stems and roots, as well as the stems of remaining dead trees (8 were lost to decay and wind) were extracted in December 2020.

Figure 2.5: Example of a photo of a tree seedling taken just before planting to document tree height, root depth, and root structure at the time of planting in June 2019 at the Beltsville, MD study site.

2.4: Soil Sampling

2.4.1: Sampling Design

Soil physical, chemical, and biological characteristics were measured prior to the experiment so site heterogeneity and changes in post-treatment soil characteristics could be understood. I measured soil penetration resistance, microbial activity, moisture, and temperature for every individual plot using the methods described in the following sections. For more time-costly soil analyses where it was not feasible to take measurements for all 112 subplots, I randomly selected two plots per treatment x replication block combination, and these were used as a representative sample subset (Figure 2.2). These sub-sampled analyses included bulk density, percent coarse material, pH, electrical conductivity, total C and N, and organic matter content. Methods used to quantify these variables are described below and summarized in Table 2.1.

Analysis	**Reference**	**Description**	**Equipment**	**Sampling Times**	**Sampling quantity**
Soil Bulk Density	Grossman and Reinsch, 2002	Excavation method	Hand trowel, plastic wrap, water, graduated cylinder, soil drying oven	t0 t2	Subset
Coarse Mineral Fraction	U.S. Bureau of Reclamation, 1991	Separation of coarse material from bulk density samples, Volume via liquid displacement	No. 4 sieve, graduated cylinder, analytical balance	t0 t2	Subset
Texture	Gee and Bauder, 1986	Sieve, Hydrometer Method	Hydrometer	t0	Subset
Penetration Resistance	Lowery and Morrison, 2002; Dayal and Allen, 1973	Cone Index	Field Scout SC900 soil compaction meter	t0 t2	All plots
Moisture	Gardner, 1986	-Sieved samples, weighed, dried at 105 °C for 24 hours, and reweighed -Time Domain Reflectometry	Hydrosense II	Seasonally	All plots
Total C & N	Yeomans and Bremner, 1991	Combustion and gas analysis by infrared detection and thermal conductivity	628 LECO C/N Analyzer model 622-000-100	t0 t1 t2	Subset
pH and Electrical Conductivity	Thomas, 1996	1:1 soil water slurry	pH and Electrical Conductivity electrodes	t0 t1 t2	Subset
Temperature	McInnes, 2002	Dielectric Impedance	Hydrosense II	Seasonally	All plots
Total Organic Matter	Robertson, 2011	Loss of mass after 24 hours at 450 °C	Lindberg/Blue M BF51828C-1 Box Furnace	t0 t1 t2	Subset
Microbial Respiration Rate	Ryan and Law, 2005	Field Infrared Gas Analysis	EGM-5 Portable Carbon Dioxide Gas Analyzer	Seasonally	All plots

Table 2.1: Soil parameters measured in this study, methods, equipment used, sampling times, and sampling quantity (every individual plot or a subset of plots) at the Beltsville, MD site. t0 = pre-treatment (March 2019 for soil texture, pH, electrical conductivity, total

C & N, and total organic matter; April 2019 for bulk density and coarse mineral fraction; April and May 2019 for penetration resistance). t1 = end of the first growing season (February 2020). t2 = end of the second growing season (January 2021 for total C & N and total organic matter; April 2021 for pH and electrical conductivity; April and May 2021 for penetration resistance; June 2021 for bulk density and coarse mineral fraction). "Seasonally" = sampling events occurred in May 2019, July 2019, September 2019, November 2019, July 2020, and September 2020.

2.4.2: Pre-treatment Soil Measurement Methods: All Plots

A Spectrum FieldScout SC900 (Aurora, IL) soil compaction meter was used to quantify soil strength in each plot prior to the experiment (Lowery and Morrison, 2002; Dayal and Allen, 1973). Four readings were taken in each plot: one reading at the center of each of the northwest, northeast, southwest and southeast quadrants of each subplot, resulting in 448 total readings per sampling event. The cone penetrometer recorded the soil penetration resistance in kPa at every 2.5 cm of depth reached. I took each reading to the furthest depth up to 30 cm that was possible without obstruction of the instrument from coarse material or excessive soil compaction and recorded the depth reached for each measurement taken. Changes in soil moisture cause soil strength to vary, so soil moisture was quantified in each plot at the time of cone penetrometer measurements to account for effects of moisture on soil strength. Soil moisture was measured to a depth of 5 cm at two locations per plot: one reading at the center of the northern and of the southern half of each plot. Soil moisture was measured at the time of the penetration resistance reading using a Stevens Water Monitoring Systems HydraprobeII dielectric permittivity sensor (Portland, OR). This soil moisture instrumentation is a component of the set of equipment used to measure soil carbon dioxide (CO_2) efflux rates described below; the measurement process simultaneously monitors soil temperature and moisture in-situ which allows for higher sampling volume and lower sampling time.

As an indicator of soil microbial respiration, I measured surface soil CO_2 efflux rates (Ryan and Law, 2005). Prior to the experiment, in May of 2019, I collected two CO_2 efflux rate readings per plot: one at the center of each of the northern and southern halves of each plot. I used an EGM-5 Portable CO_2 Gas Analyzer from PP Systems (Amesbury, MA) with a soil respiration chamber attachment to take measurements of the CO_2 efflux rate, as well as the HydraprobeII attachment to simultaneously monitor soil and air temperature and soil moisture for each measurement. Mulch was cleared from the sampling location to allow access to surface soil and cleared sampling spaces were not sampled until a minimum of 45 minutes after the soil surface had been disturbed to avoid capturing excess CO_2 release from the physical soil disturbance. The soil moisture and temperature monitoring probe was inserted into soil as close as possible to the respiration chamber's location. Connecting tubes cycled gas to and from the EGM-5 device where infrared gas analysis was used to quantify the change in CO_2 content in the chamber over time until either a reading time of 60 seconds or a CO_2 concentration of 50 ppm in the soil respiration chamber was reached.

2.4.3: Pre-treatment Soil Measurement Methods: Sample Subset

Soil bulk density was one measure used to quantify the level of soil compaction at the study site; this is a helpful metric for understanding the effects of compaction on tree seedlings. Multiple studies have demonstrated benefits to seedling growth with lower soil bulk density values in contexts where soil physical compaction levels posed challenges for root exploration (Shishiachi and Adachi et al., 1982; Corns, 1988; Minore and

Weatherly, 1990; Kube, 1993; Farrish et al., 1995). Pre-treatment bulk density samples were collected at a randomly selected location at the midpoints between the center and edge of each plot in the sampling subset. I used the same samples to determine the soil coarse material content. Bulk density was determined via the excavation method (Grossman and Reinch, 2002), wherein soil is collected and the volume of the extracted soil space is quantified by lining the excavated area with plastic and filling it with a known volume of water. In this study, a 10 cm diameter hole was excavated to a depth of 10 cm, the extracted soil was collected, volume was determined, and the hole was then extended to a depth of 20 cm and the volume measured again. This method produced bulk density measurements for both the 0 – 10 cm and 10 – 20 cm depths. After collected soils were oven dried for at least 48 hours at 105 °C and their mass was quantified, a No. 4 sieve (4.75 mm opening size) was used to separate coarse mineral content. I determined the volume of the extracted coarse material by volumetric displacement of water in a graduated cylinder and weighed the coarse material to determine the coarse material mass. I also used this information to calculate a bulk density value that excludes the mass and volume of the coarse material fraction; I refer to this value as the adjusted bulk density.

Soil collected for analysis of pre-treatment pH, electrical conductivity, total C and N, and organic matter content were taken from randomly chosen locations at the midpoint between the tree-planting spot and the edge of each plot in the sampling subset. At these sampling locations, composite soil samples were collected by mixing samples from three holes within a 150 cm^2 area that were 10 cm in diameter and 20 cm deep. Samples were separated into 0 – 10 cm and 10 – 20 cm sampling depths. Soil samples were placed in labeled ziploc bags and kept in a cooler in the field and when being transported to the lab. Samples were stored in the refrigerator (< 1 week) while they were sieved to 2mm. Field-moist subsamples were collected for microbial biomass analysis, and remaining sample was then air dried prior to further analysis. Samples were kept separate by depth for organic matter content analyses, and depths were mixed (equal parts by mass) for pH, electrical conductivity, and total C and N analyses. Soil for pre-treatment analyses was collected March 30, 2019.

Soil organic matter content was analyzed by loss on ignition using a Lindberg/Blue M BF51828C-1 Box Furnace (Asheville, NC) (Robertson, 2011). I dried 10 – 15 g samples of soil at 105 °C for 48 hours and then weighed them to 0.0001 g and placed them in the box furnace for a 24-hour cycle with 2 hours at 550 °C. Percent organic material was calculated based on the difference in dry soil mass before and after the furnace. Soil pH and electrical conductivity were quantified on soil and deionized water slurries using an APERA Instruments (Columbus, OH) AI311 Premium Series PH60 tester kit (Thomas, 1996). Because the final pH and electrical conductivity measurements of the experiment were quantified with field moist soil on a 1:1 moist soil to water ratio, the soil:water ratios for the pre-treatment measurements were adjusted based on the moisture content of the field-moist samples to keep the slurry soil:water ratios for pH and electrical conductivity consistent for all sampling times. Total C and N were determined by combustion and gas analysis by infrared detection and thermal conductivity using a 628 LECO C/N Analyzer (St. Joseph, MI) model 622-000-100 (Yeomans and Bremner, 1991). Air-dried samples were pulverized to pass through 0.5 mm sieve using a Spex SamplePrep 8000M mixer mill (Metuchen, NJ) with hard steel grinding chambers, dried at 105 °C for one hour, and weighed into 0.245 – 0.255 g samples for analysis.

2.4.4: Post-treatment Soil Measurement Methods: All Plots

I resampled for soil CO_2 efflux rates throughout the study along with soil and air temperature and soil moisture using the same methods described for pre-treatment analyses above. Post-treatment sampling events occurred in middle summer, late summer, and late fall of 2019, and middle summer and late summer of 2020. Each sampling event consisted of 2 to 3 field days completed over periods of time ranging from 2 consecutive days to a 14-day period, as sampling times were heavily determined by weather and soil moisture conditions; exact months for sample collection are listed in Table 2.1. Soil and air temperature and soil moisture data from the July 2019 sampling event were lost due to instrument recording error in the field for plots 1 – 28, so data from this sampling event were excluded from analysis. I collected soil penetration resistance data following the end of the experiment in the spring of 2021 using the same methods as in the pre-treatment sampling event described previously.

2.4.5: Post-treatment Soil Measurement Methods: Sample Subset

I collected samples for post-treatment bulk density measurements in the summer of 2021 and determined bulk density, coarse material content, and adjusted bulk density using the same methods described for pre-treatment analysis above. Bulk density samples were collected as closely as possible to the location of the pre-treatment bulk density sampling locations without overlapping with previous sample collection space and without overlapping areas where physical properties would have been altered by tree planting and extraction.

Soil sampling events for soil organic matter, pH, electrical conductivity, and total C and N analyses occurred twice after treatments were applied: once after the end of the 2019 growing season (February 1, 2020), and once after the end of the 2020 growing season (samples used for organic matter content and total C and N were collected January 6, 2021, and field moist soil for pH and electrical conductivity analysis was collected April 9, 2021). Samples were collected from the same sampling subset used in the pre-treatment sampling event for these soil analyses. In these sampling events, separate samples were collected to represent the area inside the planting hole and the area surrounding. Composite samples were collected using a screw auger (1.2 cm diameter) at five equidistant points within the planting hole area and five equidistant points outside of the planting hole area. Samples were also separated into the 0 – 10 cm and 10 – 20 cm depths and sieved to 2 mm. Samples were analyzed separately both by depth and by inside-vs-outside planting hole status for soil organic matter content. For soil pH, electrical conductivity, and total C and N, the depths were mixed (equal parts by mass), but samples remained separated by within versus outside-planting hole status. The sample transportation, storage, preparation, and analysis methods used for these two sampling events are the same as those of the pre-treatment sampling event described above. Sampling times and methods for soil variables are also summarized in Table 2.1.

2.5: Tree Data Collection

Tree height and root depth were determined from photos of trees in front of measuring tape taken the day of planting (Figure 2.5). Throughout the experiment, I visually monitored and recorded aboveground tree responses including leaf color, leaf drop, the presence of new leaves and shoots, and leader stem cambium color. Leader stem cambium color was determined via scratch test (Kinzler, 2016), and any of the following were considered aboveground signs of tree survival and qualified a tree as living during a sampling event: green cambium on leader stem, green leaves, newly emerged stems with living buds or leaves. I conducted tree survival and response observations in July 2019, September 2019, May 2020, and December 2020, when I extracted all the still-living trees with their roots, as well as the stems of dead trees. Eight dead tree stems were lost to decay or wind during the second growing season. I collected a second round of photos to document tree height and root depth (where roots could be extracted) after harvesting trees in December 2020 and noted any broken stems to account for any possible inaccuracies in quantifying the true tree height. I oven-dried living trees at 80 °C for 48 hours and weighed them to determine dry root and shoot biomass (SERAS, 1994). Sampling times and methods for tree response variables are summarized in Table 2.2.

Analysis	Reference	Description	Equipment	Sampling Times	Sampling quantity
Tree height	American Horticulture Industry Association, 2014	Measured In Situ	Measuring tape	June 2019 Dec. 2020	All trees
Survival/Mortality (aboveground)		Observed In Situ	Visual observation	July 2019 Sept. 2019 May 2020 Dec. 2020	All trees
Leader stem Survival		Scratch test	Pocket knife	July 2019 Sept. 2019 May 2020 Dec. 2020	All trees
Leaf/stem observations		-Leaf drop -Living leaves on leader stem -New shoot from root crown	Visual observation	July 2019 Sept. 2019 May 2020	All trees
Living tree dry root and shoot biomass		-Oven dry and weigh 48 hours at 80 °C	Drying oven, analytical balance	Dec. 2020	Living trees

Table 2.2: Tree parameters measured in this study, methods, equipment used, sampling times, and sampling quantity (for all trees or only for surviving trees) at the Beltsville, MD site.

2.6: Statistical Analyses

The study was a randomized complete block design with 5 soil treatments and 4 blocks. Soil treatment was the independent variable, and measured soil and tree responses were the dependent variables. Response variables quantified on all plots had a sample size of 112 per sampling event, and response variables measured on the sample subset had a sample size of 40 per sampling event. Sampling subset data collection times included pre-treatment (t0) and after the first (t1) and second (t2) growing seasons, as mentioned above, except for soil bulk density which was not quantified for the t1 sampling time. I sampled soil microbial respiration rates six times throughout the experiment (summarized above and in table 2.1) and sampled soil penetration resistance once prior to the experiment and once following the experiment.

I used two-way ANOVA for each variable at each sampling time to assess significant treatment x block effect interactions. Statistically significant differences in response variables between treatment groups and replication blocks were determined with one-way ANOVA for each variable x sampling time combination. Differences between blocks and between treatment groups were assessed for pre-treatment dependent variables to identify differences between blocks and to ensure there were no significant differences in starting conditions between treatment groups. Pairwise t-tests were used as post-hoc analysis for statistically significant ANOVA results. Datasets were assessed for normality and homogeneity, and transformations were performed to achieve normality for the analysis of some. Kruskal-Wallis test by ranks was used instead of one-way ANOVA for datasets that could not be normalized through transformations, and Wilcoxon signed rank tests were used as post-hoc analyses for significant Kruskal-Wallis tests. All statistical tests were performed using the aov(), pairwise.t.test(), kruskal.test(), and wilcox.test() functions in the R 'stats' package. The $\alpha = 0.05$ level was used as the threshold for significance (R version 4.0.2).

Chi-square analyses were used to assess differences between treatments and between blocks in categorical response variables including tree survival and other observed tree responses (leaf drop, cambium color, presence of green leaves, and presence of new shoots) for each tree observation time. Chi-square analyses were performed using the chisq.test() function in the R 'stats' package. Unpaired t-tests were used to compare soil characteristics between groups of surviving vs dead trees for each tree observation event. Unpaired t-tests were performed using the t.test() function in the R ‘mosaic’ package.

Chapter 3: Results

3.1: Pre-treatment Soil Conditions

Prior to treatment application, site soil had an average pH of 7.17 in the upper 20 cm and average soil organic matter content of 5.65 and 3.46% at the 0 – 10 cm and 10 – 20 cm depths, respectively (Table 3.1). Average pre-treatment linear soil respiration rate was 1.0639 g m^2 hr^{-1}. Average volumetric soil moisture content in the upper 5 cm of soil was 30% on average for the pre-treatment sampling event. Average soil C:N ratio in the upper 20 cm of soil at the site was 16.05 (Table 3.1). Average soil bulk density was 1.4 and 1.8 g/mL for the 0 – 10 and 10 – 20 cm sampling depths, respectively, and average penetration resistance for the 5 – 15 cm depth was 2,906 kPa (Table 3.1).

3.1.1: Block Comparisons of Pre-treatment Soil Conditions

I compared soil properties prior to treatment application between blocks to understand pre-existing variability in soil conditions in addition to the observed differences in soil compaction levels from preliminary sampling, and calculated site averages for these soil characteristics (Table 3.1). The southeast replication block (block D) had lower pre-treatment soil penetration resistance than all other blocks (northwest = block A, northeast = block B, southwest = block C), and had lower soil moisture content at the time of penetration resistance measurement than all other blocks ($p < 0.05$) (Table 3.1). Block A had lower pre-treatment C:N ratios for the 0 – 20 cm depth ($p < 0.01$, Kruskal-Wallis comparison), soil C for the 0 – 20 cm depth ($p < 0.01$), and soil organic matter content for the 0 – 10 cm ($p < 0.01$) and 10 – 20 cm ($p < 0.05$, Kruskal-Wallis comparison) depths than the other blocks (Table 3.1).

Variable	Site Avg.	Block Effect	Block A Avg.	Block B Avg	Block C Avg.	Block D Avg.	Block Differences
Bulk Density Adjusted (g/mL) 0 – 10cm 10 – 20cm	1.4 1.8	p = 0.115 p = 0.942	1.5 1.8	1.4 1.8	1.4 1.8	1.4 1.8	None None
Penetration Resistance (kPa) (5 – 15 cm reading average)	2906	p = 1.88e-06**	2940	3036	3088	2558	D<A, D<B, D<C
Soil Moisture for Penetration Resistance Reading (%, Volumetric)	30.02	p < 2e-16**	29.93	28.79	24.46	36.90	A<D, B<D, C<D, C<A, C<B
% Coarse Material by Volume: 0 – 10cm 10 – 20cm	7.7 20.4	p = 0.65 (KW) p < 2e-16**	5.5 17.4	8.1 16.4	9.1 20.9	8.5 26.9	None A<D, B<D
% Soil Organic Matter 0 – 10cm 10 – 20cm	5.65 3.46	p = 0.009** p = 0.017 (KW)*	4.83 2.73	5.76 3.83	5.89 3.47	6.11 3.79	A<B, A<C, A<D A<B, A<C, A<D
pH	7.17	p = 9.47e-06**	7.06	7.02	7.38	7.21	A<C, A<D, B<C, B<D, D<C
Electrical Conductivity (µS)	230.3	p = 0.049*	195.1	248.4	224.3	253.3	A<B, A<D
% Soil C	2.355	p = 0.0001**	1.612	2.566	2.406	2.837	A<B, A<C, A<D, C<D
% Soil N	0.1598	p = 0.1162(KW)	0.1913	0.1502	0.1419	0.1589	None
C:N Ratio	16.05	p = 0.0004 (KW)**	11.97	17.01	16.94	17.86	A<B, A<C, A<D

Table 3.1: Table shows the site averages of pre-treatment soil characteristics at the Beltsville, MD study site, the significance ($p < 0.05$*, $p < 0.01$**) of ANOVA or Kruskal-Wallis (indicated by "KW" after the p-value) tests for differences between blocks in soil conditions, and significant post-hoc test differences between blocks ($p < 0.05$). $n = 40$ for adjusted bulk density, coarse material, organic matter, pH, electrical conductivity, and soil C. $n = 39$ for soil N and C:N. $n = 112$ for penetration resistance and volumetric moisture content.

3.1.2: Treatment Group Comparisons of Pre-treatment Soil Conditions

I tested for differences in pre-treatment soil conditions between treatment groups and found no significant differences in starting soil conditions between treatment groups for any measured response variables in the study ($p > 0.05$). I also found no significant differences in tree stem height or root depth between blocks or between treatment groups at the time of planting ($p > 0.05$).

3.2: Soil Treatment Responses

Soil amendments did significantly alter metrics of soil quality. Though there were block differences in pre-treatment soil conditions, I found no significant treatment x block effect interactions in any soil or tree response datasets from two-way ANOVA. I therefore used one-way ANOVA going forward to examine treatment effects on response variables without concern for block effect interactions. Interactions between block and treatment effects were not examined for root depth and dry biomass response data because within-group sample size was too small when grouping by block (these variables were only quantified for trees that were still living in December 2020). Datasets that could not be transformed to a normal distribution were % soil N in the planting hole for t1 and t2, soil C:N ratio in the planting hole for t1 and t2 and outside the planting hole for t2, and tree stem height in December 2020.

3.2.1: Soil Physical Responses

Soil physical responses were observed following tree extraction at the end of the experiment. Measurements were taken at least 15 cm away from the planting hole area to avoid sampling where extraction may have altered these properties. Soil treatments did not have a significant effect on soil penetration resistance (average of 5 – 15cm readings) or on soil bulk density at the 10 – 20 cm sampling depth ($p > 0.05$). Soil bulk density at the 0 – 10 cm sampling depth was around ~15% lower in compost and mixed treatments than in the untilled control group ($p < 0.05$) (Table 3.2).

Treatment	Avg. Bulk Density (g/mL)	S.D.	S.E.	n	Significant Differences	p-value
Biosolids	1.2	0.053	0.019	8	None	
Compost	1.1	0.15	0.053	8	<Untilled Control	0.012
Mixed	1.1	.014	0.05	8	<Untilled Control	0.012
Tilled Control	1.2	0.13	0.046	8	None	
Untilled Control	1.2	0.15	0.056	8	>Compost >Mixed	0.012 0.012

Table 3.2: Table shows the average, standard deviation, standard error, and significant difference p-value between treatments (unpaired t-test) for coarse material-adjusted bulk density in the upper 10 cm of soil measured after the application of organic amendments at the Beltsville, MD study site (spring 2021). Only significant differences ($p < 0.05$) between treatment groups are shown.

Soil treatments influenced volumetric soil moisture content (0 – 5cm) recorded with soil respiration rate sampling in September 2019, November 2019, June 2020, and November 2020. Soil moisture was consistently higher in the untilled control group than in all other treatment groups for all four sampling events ($p < 0.05$) (Table 3.3). Additionally, soil moisture was higher in the tilled control group than in treatment groups that received organic amendment treatments for the September 2019 sampling event and was higher for the biosolids-treated group than for the compost-treated group in the November 2019 sampling event ($p < 0.05$) (Table 3.3).

Sampling Time	Treatment	Avg. % Soil Moisture	S.D.	S.E.	n	Significant Differences	p-value
September 2019	Biosolids	19.20	3.35	0.71	22	<Tilled <Untilled	0.004 3.9e-14
	Compost	19.36	2.97	0.62	23	<Tilled <Untilled	0.005 5.4e-14
	Mixed	19.31	3.22	0.70	21	<Tilled <Untilled	0.005 1.1e-13
	Tilled Control	22.09	3.33	0.68	24	>Biosolids >Compost >Mixed <Untilled	0.004 0.005 0.005 3.9e-08
	Untilled Control	27.83	3.51	0.75	22	>Biosolids >Compost >Mixed >Tilled	3.9e-14 5.4e-14 1.1e-13 3.9e-08
November 2019	Biosolids	28.01	3.53	0.75	22	>Compost <Untilled	0.038 4.0e-10
	Compost	26.04	3.02	0.63	23	<Biosolids <Untilled	0.038 6.7e-15
	Mixed	26.55	3.04	0.66	21	<Untilled	3.0e-13
	Tilled Control	26.90	3.29	0.67	24	<Untilled	4.9e-13
	Untilled Control	34.58	2.88	0.61	22	>Biosolids >Compost >Mixed >Tilled	4.0e-10 6.7e-15 3.0e-13 4.9e-13
June 2020	Biosolids	16.58	4.05	0.86	22	<Untilled	5.6e-05
	Compost	16.93	3.89	0.81	23	<Untilled	0.0001
	Mixed	18.15	4.35	0.95	21	<Untilled	0.004
	Tilled Control	17.66	4.12	0.84	24	<Untilled	0.0009

	Untilled Control	21.97	4.84	1.03	22	>Biosolids >Compost >Mixed >Tilled	5.6e-05 0.00013 0.00404 0.00085
November 2020	Biosolids	30.77	4.38	0.93	22	<Untilled	0.008
	Compost	30.66	5.22	1.09	23	<Untilled	0.006
	Mixed	31.57	5.17	1.13	21	<Untilled	0.04
	Tilled Control	30.41	3.05	0.62	24	<Untilled	0.003
	Untilled Control	34.33	3.66	0.78	22	>Biosolids >Compost >Mixed >Tilled	0.008 0.006 0.04 0.003

Table 3.3: Table shows the average, standard deviation, standard error, and significant difference p-value between treatments (unpaired t-test) for volumetric soil moisture content in the upper 5 cm of soil measured at the time of seasonal soil microbial respiration readings over the course of the experiment in September 2019, November 2019, June 2020, and November 2020, following application of organic amendments at the Beltsville, MD study site. Only significant differences ($p < 0.05$) between treatment groups are shown.

3.2.2 Soil Chemical Responses

There were no significant mean differences between treatment groups at the t1 or t2 sampling times for measured soil chemical response variables (those that could meet normality assumptions for ANOVA) including exponential-transformed pH, electrical conductivity, and % soil C. I found through analysis of remaining data sets (those that could not meet normality assumptions) that average % soil N in the planting hole area was higher in the mixed treatment group than in the biosolids and tilled control groups for the t1 and t2 sampling events, and was higher in the compost treatment group than in the tilled control group for the t2 sampling event ($p < 0.05$) (Tables 3.4 and 3.5). Average

C:N ratio inside the planting hole area was also higher in the tilled control group than in the compost and mixed treatment groups for the t2 sampling event ($p < 0.05$) (Table 3.6).

Average soil pH remained above 7 for post-treatment sampling times after both growing seasons. Average pH for the 0 – 20 cm depth was 7.16 prior to treatment application, 7.00 and 7.23 after the first growing season inside and outside the planting hole area, respectively, and 7.53 and 7.54 after the second growing season inside and outside the planting hole area, respectively.

Treatment	Avg. % Soil N	S.D.	S.E.	n	Significant Differences (Wilcoxon)	p-value
Biosolids	0.1100	0.0474	0.0168	8	<Compost <Mixed	0.083 0.010
Compost	0.1588	0.0677	0.0239	8	>Biosolids	0.083
Mixed	0.3384	.3374	0.1192	8	>Biosolids >Tilled	0.010 0.028
Tilled Control	0.1232	0.0396	0.0140	8	<Mixed	0.028
Untilled Control	0.1333	0.0623	0.0220	8	None	

Table 3.4: Table shows the average, standard deviation, standard error, and significant difference p-value between treatments (Wilcoxon signed rank) for % total soil N in the planting hole in the upper 20 cm of soil measured after the first growing season (February 2020) at the Beltsville, MD study site. Only significant differences ($p < 0.05$) between treatment groups are shown.

Treatment	Avg. % Soil N	S.D.	S.E.	n	Significant Differences (Wilcoxon)	p-value
Biosolids	0.0609	0.0351	0.0124	8	<Mixed	0.021
Compost	0.0985	0.0375	0.0133	8	>Tilled	0.021
Mixed	0.1870	0.2540	0.0898	8	>Biosolids >Tilled	0.021 0.010
Tilled Control	0.0524	0.0288	0.0102	8	<Compost <Mixed	0.021 0.010
Untilled Control	0.1078	0.1169	0.0413	8	None	

Table 3.5: Table shows the average, standard deviation, standard error, and significant difference p-value between treatments (Wilcoxon signed rank) for % total soil N in the planting hole in the upper 20 cm of soil measured after the second growing season (January 2021) at the Beltsville, MD study site. Only significant differences ($p < 0.05$) between treatment groups are shown.

Treatment	Avg. C:N	S.D.	S.E.	n	Significant Differences (Wilcoxon)	p-value
Biosolids	75.22	115.3	40.77	8	None	
Compost	25.40	4.460	1.576	8	<Tilled	0.015
Mixed	22.69	9.526	3.368	8	<Tilled	0.021
Tilled Control	107.7	205.2	72.55	8	>Compost >Mixed	0.015 0.021
Untilled Control	28.19	11.18	3.953	8	None	

Table 3.6: Table shows the average, standard deviation, standard error, and significant difference p-value between treatments (Wilcoxon signed rank) for soil C:N ratios in the planting hole in the upper 20 cm of soil measured after the second growing season (January 2021) at the Beltsville, MD study site. Only significant differences ($p < 0.05$) between treatment groups are shown.

3.2.3: Soil Biological Responses

Average soil organic matter content in the upper 10 cm of the planting hole area was ~20% higher in the compost and mixed treatments than in all other treatments for both the t1 and t2 sampling times ($p < 0.05$) (Tables 3.7 and 3.8). There were no significant differences between treatment groups in soil organic matter content in the planting hole at the 10 – 20 cm depth, or outside the planting hole area.

Treatment	Avg. % S.O.M.	S.D.	S.E.	n	Significant Differences	p-value
Biosolids	5.06	1.34	0.475	8	<Compost <Mixed	0.007 0.005
Compost	6.67	0.999	0.353	8	>Biosolids >Tilled >Untilled	0.007 0.029 0.002
Mixed	6.73	1.38	0.489	8	>Biosolids >Tilled >Untilled	0.005 0.023 0.001
Tilled Control	5.39	0.934	0.330	8	<Compost <Mixed	0.029 0.023
Untilled Control	4.77	0.872	0.308	8	<Compost <Mixed	0.002 0.001

Table 3.7: Table shows the average, standard deviation, standard error, and significant difference p-value between treatments (unpaired t-test) for soil organic matter content sampled at the Beltsville, MD study site in the planting hole area at the 0 – 10 cm depth for after the first growing season (February 2020). Only significant differences ($p < 0.05$) between treatment groups are shown.

Treatment	Avg. % S.O.M.	S.D.	S.E.	n	Significant Differences	p-value
Biosolids	4.89	1.12	0.396	8	<Compost <Mixed	0.012 0.007
Compost	6.32	1.19	0.419	8	>Biosolids >Tilled >Untilled	0.012 0.022 0.017
Mixed	6.45	1.22	0.433	8	>Biosolids >Tilled >Untilled	0.007 0.013 0.009
Tilled Control	5.03	0.914	0.323	8	<Compost <Mixed	0.022 0.013
Untilled Control	4.96	0.899	0.318	8	<Compost <Mixed	0.017 0.009

Table 3.8: Table shows the average, standard deviation, standard error, and significant difference p-value between treatments (unpaired t-test) for soil organic matter content sampled at the Beltsville, MD study site in the planting hole area at the 0 – 10 cm depth for after the second growing season (January 2021). Only significant differences ($p < 0.05$) between treatment groups are shown.

Mean soil linear respiration rates were lowest in untilled control plots and were higher in compost and mixed treatments than the tilled control for the September 2019 sampling event ($p < 0.05$) (Table 3.9). Mean log soil linear respiration rates from the November 2019 sampling event were higher in the compost treatment than in the biosolids treatment and the untilled control ($p < 0.05$). Mean soil linear respiration rates were higher in compost and mixed treatments than in all other treatments for the June 2020 sampling event ($p < 0.05$) (Table 3.9). I found no significant differences in log soil linear respiration rates between treatment groups for the November 2020 sampling event ($p > 0.05$).

Sampling Time	Treatment	Avg. CO_2 rate (g m^2 hr^{-1})	S.D.	S.E.	n	Significant Differences	p-value
September 2019	Biosolids	0.8458	0.2481	0.0529	22	>Untilled	3.5e-05
	Compost	0.9252	0.2320	0.0484	23	>Tilled >Untilled	0.004 1.8e-07
	Mixed	0.9343	0.2405	0.0525	21	>Tilled >Untilled	0.004 1.7e-07
	Tilled Control	0.7428	0.1873	0.0382	24	>Untilled	0.007
	Untilled Control	0.5681	0.1520	0.0324	22	<Biosolids <Compost <Mixed <Tilled	3.5e-05 1.8e-07 1.7e-07 0.007
November 2019 (log)	Biosolids	0.3139	0.0939	0.0200	22	<Compost	0.003
	Compost	0.4214	0.1689	0.0352	23	>Biosolids >Untilled	0.003 0.006
	Mixed	0.3640	0.1056	0.0231	21	None	
	Tilled Control	0.3131	0.0960	0.0196	24	None	
	Untilled Control	0.3162	0.0796	0.0170	22	<Compost	0.006
June 2020	Biosolids	0.9245	0.1618	0.0345	22	<Compost <Mixed	0.005 0.012
	Compost	1.0700	0.1447	0.0302	23	>Biosolids >Tilled >Untilled	0.005 0.003 0.0002
	Mixed	1.0548	0.1606	0.0350	21	>Biosolids >Tilled >Untilled	0.012 0.009 0.0006
	Tilled Control	0.9206	0.1531	0.0313	24	<Compost <Mixed	0.003 0.009

	Untilled Control	0.8733	0.2122	0.0452	22	<Compost <Mixed	0.0002 0.0006

Table 3.9: Table shows the average, standard deviation, standard error, and significant difference p-value between treatments (unpaired t-test) for soil linear respiration rates sampled in September 2019, November 2019, and June 2020, following application of organic amendments at the Beltsville, MD study site. Only significant differences ($p < 0.05$) between treatment groups are shown. November 2019 respiration rate data were log-transformed before mean comparisons.

3.3: Tree Responses

Tree mortality was highest within the first month of the study and decreased throughout the remainder of the study (Figure 3.1). Of the 112 trees originally planted, 69 survived through the first month after planting (July 2019), 48 survived through the first three months after planting (September 2019), 23 survived to the beginning of the second growing season (May 2020), and 19 survived through the second growing season to the end of the study (December 2020).

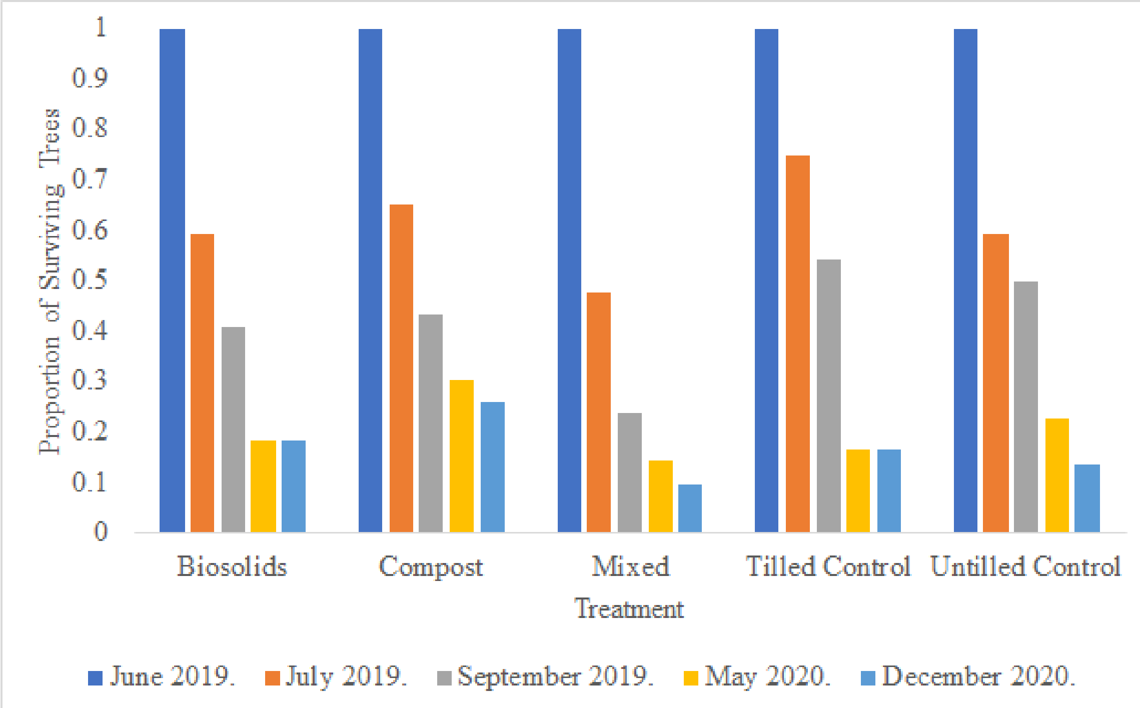

Figure 3.1: Bar graphs show the proportion of surviving trees out of trees originally planted throughout the study following amendment application and tree planting for each soil treatment group at the Beltsville, MD study site. Sampling events displayed occurred July 2019, September 2019, May 2020, and December 2020, and there are bars representing all trees initially planted in June 2019. Chi-square analyses did not indicate any significant treatment effects on survivorship rates ($p > 0.05$). The number of trees originally planted was 22 for biosolids, 23 for compost, 21 for combined biosolids and compost, 24 for tilled control plots, and 22 for untilled control plots.

Premature leaf drop was observed for many trees in the first season of this study and occurred in different proportions between treatment groups. Of the 112 trees originally planted, full or partial premature leaf drop had occurred in 18 (16.07%) within the first month after planting (by July 2019), and in 23 (20.54%) by the third month after planting (September 2019). In all these instances of premature leaf drop, leaves wilted and turned brown before dropping (Figure 3.2). Other trees had leaves wilt and turn brown after planting, but leaves remained attached to stems instead of dropping (Figure 3.3). By July 2019, one month after planting, the percentage of trees that had dropped leaves in each treatment group was 0% for biosolids, 13% for compost, 14% for the mixed treatment, 33% for the tilled control group, and 18% for the untilled control group. By September 2019, 3 weeks after planting, the percentage of trees that had dropped leaves in each treatment group was 5% for biosolids, 13% for compost, 24% for the mixed treatment, 33% for the tilled control group, and 27% for the untilled control group.

Figure 3.2: (left) Seedling showing premature leaf drop in the first growing season after planting, with all remaining leaves wilted and brown; (right) Seedling showing wilted and brown leaves that did not drop in the first growing season after planting at the Beltsville, MD study site.

Figure 3.3: Seedlings that had experienced leader stem death and had produced new living shoots and leaves originating from below the base of the leader stem at the Beltsville, MD study site.

For trees that prematurely dropped some or all leaves, a greater proportion survived for longer in the study than for trees that did not prematurely drop leaves. Of the 18 trees that dropped leaves within the first month of planting, 13 (72%) survived to September 2019, 5 (28%) survived to May 2020, and 4 (22%) survived to December 2020. In contrast, of the 94 trees that had not dropped leaves within the first month after planting, only 34 (36%) survived to September 2019, 18 (19%) survived to May 2020, and 15 (16%) survived to December 2020. Of the 23 trees that had dropped leaves in the first three months after planting, 18 (78%) survived to September 2019, 9 (39%) survived to May 2020, and 5 (22%) survived to December 2020. In contrast, of the 89 trees that did not drop leaves within the first three months after planting, 29 (33%) survived to September 2019, 14 (16%) survived to May 2020, and 14 (16%) survived to December 2020.

Survival of leader stems and leaves shortly after transplant was a common observation among trees that survived for the entire duration of the study. Among the 19 seedlings that survived through the entire duration of the study, all 19 had still-surviving leader stems and at least some still-living leaves one month after planting. By September 2019, 16 of these 19 trees still had living leader stems, and the remaining 3 with dead leader stems had produced new living shoots from near the base of the leader stem (Figure 3.3). In contrast, of the 93 seedlings that had died by the end of the study (December 2020), 44 had dead leader stems one month after planting, and only 19 had at least some living leaves one month after planting.

3.3.1: Block Differences in Tree Responses

Block A had a higher proportion of trees showing leaf drop than block C, and a lower proportion of trees with green leaves than blocks C and D in July 2019 ($p < 0.05$). Block A had a lower proportion of tree survival and a lower proportion of trees with green leaves than blocks C and D in September 2019, and block B had a lower proportion of trees with green leaves than block C at this time ($p < 0.05$). The proportion of trees with living leader stem leaves was higher in block D than in all other blocks in May 2020 ($p < 0.05$). The proportion of surviving trees was higher in block D than in block A in December 2020 ($p < 0.05$).

3.3.2: Treatment Differences in Tree Responses

Metrics of tree survival did not differ between soil amendments and controls. There were no significant treatment effects on log-transformed dry stem biomass, dry root biomass, or root depth ($p > 0.05$) from living trees harvested at the end of the experiment. There were also no differences in stem height between treatment groups at the end of the experiment ($p > 0.05$).

There were no significant treatment effects on tree survival for the July 2019, September 2019, May 2020, and December 2020 observation times ($p > 0.05$), and tree survival decreased until May 2020 for all treatment groups, and through December 2020 for compost, mixed, and untilled control treatment groups (Figure 3.1). There were also no significant differences between treatments in the proportion of trees that had leaf drop, green leaves, green leader stem cambium, or new stems. The only exception to this was the proportion of trees that had dropped leaves in July 2019, which was higher in tilled control plots than in biosolids-treated plots; leaf drop had not occurred in July 2019 for any biosolids-treated plots ($p < 0.05$).

3.4: Relating Soil Properties and Tree Survival

Averages of soil characteristics from all sampling times were compared between tree survivorship groups (plots with living trees versus plots with dead trees) from each tree observation event in the study (July 2019, September 2019, May 2020, and December 2020). Some soil physical, chemical, and biological characteristics were correlated with tree survival status.

Plots with trees that survived to July 2019 had lower mean pre-treatment bulk density (0 – 10 cm) ($p < 0.05$), higher soil moisture content (from the November 2020 sampling event), and higher post-treatment soil penetration resistance compared to the non-surviving tree group ($p < 0.05$) (Table 3.10). The July 2019 living tree group also had significantly higher pre-treatment percent organic matter, higher percent organic matter (sampled in the planting hole area) at the t1 sampling time, and higher percent organic matter (sampled outside the planting hole area) at the t2 sampling time than plots with dead trees (Table 3.10). Additionally, the July 2019 living tree group had higher pre-treatment electrical conductivity and % C, higher % C and % N (sampled outside the planting hole area) at the t1 sampling time ($p < 0.05$), and higher pH (sampled in the planting hole area) at the t2 sampling time (Table 3.10).

Post-treatment adjusted bulk density for the 10 – 20cm depth was lower in the living tree than in the dead tree groups from both September 2019 and May 2020 ($p < 0.05$) (Tables 3.11 and 3.12). November 2019 log-transformed soil microbial respiration rates were lower in living tree groups than in dead tree groups for all four tree survival observation times ($p < 0.05$) (Tables 3.10, 3.11, 3.12, and 3.13). November 2020 log-transformed microbial respiration rates were also lower in living tree groups than in dead tree groups for all tree survival observation times except July 2019 ($p < 0.05$) (Tables 3.11, 3.12, and 3.13). September 2019 microbial respiration rates were lower for the September 2019 living tree group than the dead tree group ($p < 0.01$) (Table 3.11).

Soil Variable	Living Trees Avg. Subsample n=26 All plots n=69	Dead Trees Avg. Subsample n=14 All plots n=43	p-value
Adjusted Bulk Density (g/mL) t0: 0 – 10 cm	1.4	1.5	0.046*
% Soil Moisture (at time of respiration reading) Nov. 2020	32.5	29.9	0.002**
Penetration Resistance (kPa) t2 2.5 – 15 cm (kPa)	1734	1528	0.024*
% Soil Organic Matter t0: 0 – 20 cm 0 – 10 cm t1, In planting hole: 0 – 10 cm 10 – 20 cm t2, Outside planting hole: 0 – 10 cm 10 – 20 cm	 4.81 5.97 5.86 4.09 5.37 3.25	 4.08 5.06 5.10 3.22 4.54 2.54	 0.002** 0.0003** 0.033* 0.011* 0.041* 0.024*
Soil Respiration Rate (g*m^2/hr) Nov. 2019 (log)	-1.1906749 (log) e^-1.1906749 = 0.3040	-0.9913607 (log) e^-0.9913607 = 0.3711	0.001**
pH t2, In planting hole	2073.934 (exp) log(2073.9) =7.64	1668.6 (exp) log(1668.6) = 7.42	0.009**
Electrical Conductivity (μS) t0	243	206	0.032*
% Soil C t0 t1, Outside planting hole	 2.579 2.532	 1.941 2.009	 0.005** 0.047*
% Soil N t1, Outside planting hole	0.1479	0.1224	0.048*

Table 3.10: Comparison of soil properties for plots with living trees and dead trees in July 2019 at the Beltsville, MD study site. Table reports the average value for each soil response variable in each group, and t-test significance value for the mean difference ($p < 0.05$*, $p < 0.01$**). Only variables with significant differences ($p < 0.05$) are reported. t0 = pre-treatment (March 2019 for % organic matter, electrical conductivity, and % C; April 2019 for bulk density). t1 = end of first growing season (February 2020). t2 = after second growing season (January 2021 for % organic matter; April 2021 for pH; April and May 2021 for Penetration Resistance). Soil respiration rate data were log transformed for mean comparison.

Soil Variable	Living Trees Avg. Subsample n=16 All plots n=47	Dead Trees Avg. Subsample n=24 All plots n=65	p-value
Bulk Density Adjusted (g/mL) t2 10 – 20 cm	1.4	1.6	0.023*
Soil Respiration Rate (g*m^2/hr) Sep. 2019 Nov. 2019 (log) Nov. 2020 (log)	 0.7183 -1.210788 (log) e^-1.210788 = 0.2980 -1.2028 (log) e^-1.2028 = 0.3004	 0.8627 -1.044278 (log) e^-1.044278 = 0.3519 -1.0592 (log) e^-1.0592 = 0.3467	 0.002** 0.006** 0.012*

Table 3.11: Comparison of soil properties for plots with living trees and dead trees in September 2019 at the Beltsville, MD study site. Table reports the average value for each soil response variable in each group, and t-test significance value for the mean difference **(**p < 0.05*, p < 0.01**). Only variables with significant differences (p < 0.05) are reported. t2 = after second growing season (June 2021 for bulk density). Soil respiration rate data from November 2019 and November 2020 were log transformed for mean comparison.

Soil Variable	Living Trees Avg. Subsample n=9 All plots n=23	Dead Trees Avg. Subsample n=31 All plots n=89	p-value
Bulk Density adjusted (g/mL) t2 10 – 20 cm	1.4	1.6	0.041*
Soil Respiration Rate (g*m^2/hr) Nov. 2019 (log) Nov. 2020 (log)	 -1.272565 (log) e^ -1.272565 = 0.2801 -1.257562 (log) e^-1.257562 = 0.2843	 -1.073214 (log) e^-1.073214 = 0.3419 -1.083770 (log) e^-1.083770 = 0.3383	 0.007** 0.013*

Table 3.12: Comparison of soil properties for plots with living trees and dead trees in May 2020 at the Beltsville, MD study site. Table reports the average value for each soil response variable in each group, and t-test significance value for the mean difference **(**p < 0.05*, p < 0.01**). Only variables with significant differences (p < 0.05) are reported. t2 = after second growing season (June 2021 for bulk density). Soil respiration rate data were log transformed for mean comparison

Soil Variable	Living Trees Avg. Subsample n=7 All plots n=19	Dead Trees Avg. Subsample n=33 All plots n=93	p-value
Soil Respiration Rate (g*m^2/hr) Nov. 2019 (log) Nov. 2020 (log)	 -1.324526 (log) e^-1.324526 = 0.2659 -1.271296 (log) e^-1.271296 = 0.2805	 -1.071173 (log) e^-1.071173 = 0.3426 -1.088439 (log) e^-1.088439 = 0.3367	 0.0001** 0.015*

Table 3.13: Comparison of soil properties for plots with living trees and dead trees in December 2020 at the Beltsville, MD study site. Table reports the average value for each soil response variable in each group, and t-test significance value for the mean difference

($p < 0.05$*, $p < 0.01$**). Only variables with significant differences ($p < 0.05$) are reported. Soil respiration rate data were log transformed for mean comparison

Chapter 4: Discussion and Conclusion

4.1: Overview

In this thesis I evaluated how the addition of compost and biosolids to soil—both individually and in combination—affected soil physical, chemical, and biological response variables and newly planted tree survival, and I identified soil characteristics correlated with tree survival. The results of my thesis demonstrated that the addition of organic amendments did have impacts on soils at an urban vacant lot proxy site in Beltsville, MD. Among these impacts were significant ($p < 0.05$) decreases in bulk density with the use of compost (~15%) (Table 3.2), increases in organic matter content with the use of compost (~20%) (Tables 3.7 and 3.8), and increases in microbial respiration rates with the use of either amendment (~15 to 20%) (Table 3.9) compared to unamended controls. Effects on microbial respiration were more lasting and significant with the use of compost than with the use of biosolids (Tables 3.9). Tree mortality overall was high (83%), and tree mortality rates did not differ significantly ($p > 0.05$) between treatments. I was able to identify soil characteristics that may serve as meaningful predictors of tree survival by comparing soil characteristics between surviving and dead tree groups throughout the study. I found lower soil compaction and higher organic matter content in living tree groups compared to dead tree groups observed early in the study (Tables 3.10, 3.11, and 3.12) and lower microbial activity in living tree groups throughout the entire study (Tables 3.10, 3.11, 3.12 and 3.13). Lastly, I identified soil quality and planting practice factors that were likely important to tree survival for this study.

4.2: Trends in Soil Quality Variables

4.2.1: Pre-existing Site Conditions and Natural Forest Soil Conditions

Pre-existing site soil pH was an average of 7.17 in the upper 20 cm and average organic matter content was 5.65 and 3.46% at the 0 – 10 cm and 10 – 20 cm depths, respectively (Table 3.1). This is quite different compared to the more "ideal" growing conditions for *Q. alba* found in northeastern forest soils where pH is found in ranges as low as 2.28 to 5, and upper soil organic matter content averages 11.9% (Zukswert et al., 2021). Average volumetric soil moisture content in the upper 5 cm of soil was 30% for the pre-treatment sampling event. The soil moisture levels measured in site soils are not concerning as *Q. alba* seedling survival as high as 87% has been observed in soil conditions where available soil moisture was as low as 3% (Minckler, 1965). Average soil C:N ratio in the upper 20 cm was 16.05 (Table 3.1), which is on the lower end of a range of 11.6 – 45.3 that is typical for northeastern forest soils (Ross et al., 2011). Average soil bulk density was 1.4 and 1.8 g/mL for the 0 – 10 cm and 10 – 20 cm sampling depths, respectively, and average penetration resistance for the 5 – 15 cm depth was 2,906 kPa (Table 3.1). Soil strength values above 2,000 kPa are restrictive to the growth of most roots (Gilker et al., 2002).

4.2.2: Treatment Effects on Soil Variables

The addition of compost and biosolids to soils at this proxy site for urban land use affected some key soil physical, chemical, and biological response variables. Soil bulk density decreased with the addition of compost in comparison to controls ($p < 0.05$) as hypothesized, but biosolids application alone had no significant impact ($p > 0.05$) on bulk density (Table 3.2). The decrease in soil compaction with the addition of compost is consistent with existing knowledge that has demonstrated the strength of compost as a physical soil conditioner (Amlinger et al., 2007). Though tillage decreases soil bulk density temporarily, in my study, only the addition of compost led to lasting decreases in soil bulk density compared to controls ($p < 0.05$) that were observable two years after treatment application. Similarly, organic matter content increased with the addition of compost in both the compost and mixed treatments compared to controls ($p < 0.05$), but biosolids application alone had no significant impact ($p > 0.05$) on organic matter content (Tables 3.7 and 3.8).

The absence of a significant effect of biosolids on organic matter content and bulk density is inconsistent with results from similar studies that found increases in organic matter content and decreases in bulk density with biosolids additions (Scharenbroch et al., 2013; García-Orenes et al., 2005). However, it is important to consider that because of concerns over nutrient pollution, the biosolids application rate used in this study was low compared to the application rates of studies that saw increases in organic matter with biosolids (Scharenbroch et al., 2013; García-Orenes et al., 2005) and compared to the application rate of compost treatments in this study. The influence of organic matter dilution effects (the decrease in soil bulk density resulting directly from the integration of large amounts of lower density organic material) on bulk density was likely greater with the higher application rate of compost in this study and of biosolids in other studies (García-Orenes et al., 2005; Layman et al., 2016). In addition to this, the effects of biosolids application on soil compaction and organic matter content can quickly become undetectable at lower application rates (22 Mg/ha or less) due to quick decomposition of the added organic matter (Larney and Angers, 2012; Bendfeldt et al., 2001).

Microbial respiration (CO_2 efflux) rates measured throughout this study represent the degree to which treatments stimulated soil microbial activity. Soil microbial activity impacts many crucial plant-supporting functions including nutrient cycling, inorganic nutrient transformations, and N fixation (Weil and Brady, 2017). Compost treatments led to increases in soil microbial respiration rates compared to both tilled and untilled controls throughout the entire study ($p < 0.05$) (Table 3.9). Microbial respiration rates were higher in both biosolids treatment plots and tilled control plots than in untilled controls ($p < 0.05$) in the first growing season, but these differences did not persist into the second growing season (Table 3.9). An increase in microbial respiration following the addition of compost and biosolids is consistent with observations from other studies, including urban studies (Scharenbroch and Watson, 2014; Baballari, 2019; Scharenbroch et al., 2013; Treonis et al., 2010).

The longer-lasting increase in microbial respiration rates observed with compost treatments compared to biosolids may be attributed to the amendments' differing C:N ratios. Increases in microbial activity from the addition of organic materials with lower C:N ratios (such as biosolids) tend to subside more rapidly after a brief initial spike while increases in microbial activity from the addition of amendments with a higher C:N ratio (such as compost) tend to subside very gradually over time (Weil and Brady, 2017). Additionally, there were no differences in soil microbial respiration rates between the biosolids treatment and the tilled control at any time in the study (Table 3.9), so a considerable portion of the increased microbial activity observed in the biosolids treatment can likely be attributed to the effects of tillage alone (Treonis et al., 2010). The lower application rate of biosolids than compost discussed above is again important to consider here. The effect of biosolids on soil respiration rates would likely have been more profound given a higher application rate, as has been observed in other studies where biosolids were applied at higher rates than in this study (Scharenbroch et al., 2013; Scharenbroch and Watson, 2014).

Soil moisture content was consistently higher in untilled control plots than in all other treatment groups ($p < 0.05$) throughout the study (Table 3.3). Soil moisture was measured simultaneously with soil respiration sampling events, which were always conducted at least 24 hours after any rainfall or irrigation event, and moisture was measured to a depth of 5 cm. Compacted soils often hold onto soil moisture for longer periods of time (Badalíková, 2010). It is possible that tillage reduced compaction in this study (though this effect was not detected in the bulk density and penetration resistance data), increasing the rate at which water drains beyond the 5 cm depth reached by the soil moisture probe in tilled plots (Somerville et al., 2018). Additionally, soil moisture was quantified by an in-situ soil moisture probe based on volume, and lower soil bulk density can result in lower dielectric permittivity, resulting in lower soil moisture content readings (Gong et al., 2003). Soil moisture data determined by mass-based methods may have shown a different treatment effect.

4.3: Trends in Tree Survivorship

The addition of compost and biosolids to soils at this urban proxy site did not have any significant effect ($p > 0.05$) on tree survivorship rates (Figure 3.1). These observations are like those of Somerville et al. (2018), who found no differences in tree growth between compost-treated plots compared to controls. The high number of trees that either died or prematurely shed leaves in the first growing season helps explain why differences were not observable in height and biomass metrics for tree stems and roots at the end of the experiment, as many trees did not have sufficient time to demonstrate any growth responses. Tree mortality in this study was around 83% by the end of the second growing season (Figure 3.1). Mortality rates are often high for small trees in urban settings, especially in the years immediately after establishment; urban tree mortality in the first five years after establishment typically ranges from 5 to 10% (Hilbert et al., 2019), but mortality rates as high as 68% have been reported (Yang and McBride, 2003).

4.3.1: Planting Practices and Transplant Shock

Transplant shock is defined as "seedling mortality or impaired growth soon after planting (p. 112)" (Close et al., 2005), and it is the most common culprit of death in recently planted trees (Gauthier et al., 2014). Symptoms of transplant shock include premature leaf drop, leaf scorching or browning, and branch dieback (Gauthier et al., 2014; Reitveld, 1989). Tree survival observations including premature leaf drop, leaf wilting, and leader stem death (Section 3.3) (Figure 3.2) throughout this study suggest that initial transplant shock was one of the strongest drivers of tree mortality. Most of the trees that survived through the first growing season and winter (23 out of 112 trees) also survived through the second growing season (19 out of 112 trees) to the end of the study (Figure 3.1). Furthermore, all 19 seedlings that survived to the end of the study had still-living leaves through the entire first growing season, and only 3 of those 19 had leader stem dieback occur during the study (Section 3.3).

It is important to consider the unique circumstances of tree planting in this study and how this also likely contributed to the high mortality rate observed. The timing of planting was a major factor. Trees were planted in June, when leaves were already out. The recommended practice is to extract and transplant trees while they are still dormant in fall or spring for the best chance of survival from transplant shock (Department of Environmental Protection and Sustainability, 2013). Logistical barriers prevented the bare root trees originally ordered for this study to be planted when planned in early spring and new trees had to be acquired, prolonging the planting time to late spring. Furthermore, the trees that were acquired for planting were delivered in pots rather than bare root. Obtaining trees that were extracted bare root and planting during dormancy would have been the preferred practice in an ideal version of this study.

Though tree roots were removed from pots with care in this study, there is always root loss and some risk of root damage when separating roots from potting mix. The significant loss of root mass during the extraction of trees for bare root planting may have contributed to the high frequency of leader stem death in this study (Section 3.3) (Hamilton, 1989). A sudden decrease in root volume is not desirable for re-establishment following transplant as excessively high shoot:root ratios can make successful establishment infeasible (Larsen et al., 1988). In transplant scenarios with shoot:root ratios that are too high, there may not be sufficient root volume for water and nutrient acquisition to support the higher proportion of shoots, leading to drought stress, nutrient stress, or both (Ledig, 1983; Reitveld, 1989; Close, 2001). Drought stress is one of the most common causes of transplant shock in seedlings (Jarvis and Jarvis, 1963; Burdett et al., 1983; Grossnickle, 1988), and though the combination of rainfall, irrigation, and mulching minimized dry soil conditions in this study, insufficient root volume for water acquisition may have led to drought stress. Irrigation is also not frequently implemented or available for many urban sites, and considering water needs for planted trees is important; low-cost options do exist for water and nutrient containment devices such as the Tree-T-Pee and tree watering bags and rings for irrigation.

4.3.2: Possible Roles of Soil in Tree Mortality

Though the design of this study does not allow me to definitively distinguish the role of soil conditions from the role of planting technique in the high tree mortality observed, elements of the soil environment may have also contributed to tree stress and mortality. High soil compaction—which leads to in reduced root growth and access to water and air—is one common contributor to transplant stress. After treatment applications, average soil penetration resistance for the 2.5 – 15 cm depth was below the 2,000 kPa root-inhibition threshold (Table 3.10) (Weil and Brady, 2017). However, soil strength values may have been higher if measured at a time when soil moisture content was lower (Gilker et al., 2002). Thus, soil compaction may still have contributed to root stress in this experiment, especially in unamended and untilled control plots where there was no intervention to help reduce compaction, apart from digging of the planting hole.

The high pH of soils at the Beltsville site also likely contributed to the high tree mortality observed. High soil pH can lead to base-cation nutrient imbalances that can threaten tree health (Zukswert et al., 2021). In northeastern forests—considered the native or natural environment for *Q.alba*—soil pH levels are typically below 5 (Zukswert et al., 2021). Urban soils, in contrast, are frequently alkaline due to factors such as urban fill materials containing limestone and excess deposition of pollutants and nutrients in the urban environment (Sonti et al., 2021). At the Beltsville study site, despite the application of ferrous sulfate to help bring pH down to the optimal range of 4.5 – 6.8 for the tree species (Slattery et al., 2003), soil pH remained above 7 throughout the study for all treatment groups (Section 3.2.2). The organic amendments used in this study both had pH values greater than 7 and could have influenced pH, though this effect was not detected as there were no differences ($p > 0.05$) in pH between amended soils and unamended controls (Section 3.2.2).

It is more likely that pH at the study site remained high because a larger application rate of ferrous sulfate was needed than what was applied to sufficiently reduce soil pH at the Beltsville site. Ferrous sulfate application rates used in this study were determined based on recommendations provided with preliminary soil test results from the Penn State Agricultural Analytical Services Laboratory (University Park, PA). Soil testing labs such as these tend to base recommendations on agricultural soil data and nutrient management goals. While it is common for these recommendations to be used for urban soil management as well (Scharenbroch et al., 2014), future work to inform soil pH management recommendations based on data from urban forestry contexts could prove more effective for positive urban forestry outcomes.

Overapplication of fertilizer at the time of planting can also result in extreme soil chemical conditions for roots, referred to as "root burn" (Gauthier et al., 2014). However, N fertilizer was only applied in some treatment plots in this study, so this was not likely a strong contributor to the transplant shock observed. Macronutrient deficiencies are another common reason for transplant shock (Close et al., 2005), though macronutrient deficiencies could not be determined by the dataset collected in this study; future study designs should consider including plant available nutrient data.

4.4: Relationships Between Soil and Plant Variables

Multiple soil characteristics were different ($p < 0.05$) between tree survivorship groups (living versus dead trees) at different tree observation times throughout the study, though these relationships did not translate directly to soil treatment effects on tree survival. There were significant differences ($p < 0.05$) in microbial respiration rates between living and dead tree groups for all tree survivorship observation events in the study (Tables 3.10, 3.11, 3.12, and 3.13). This suggests the importance of soil chemical and biological interactions and their effects on the soil root environment for future research.

Bulk density was different ($p < 0.05$) between tree survivorship groups for all tree observation events in the study except December 2020, at the end of the study (Tables 3.10, 3.11, and 3.12) and organic matter content was different ($p < 0.05$) between tree survivorship groups observed one month after planting (Table 3.10). It is possible that insufficient sample sizes for these variables did not allow for differences in bulk density and organic matter between survivorship groups later in the study to be detected. Still, their correlations with tree survival observations early in the study are consistent with the notion that bulk density and organic matter are important considerations for future research identifying soil quality priorities for forestry (Layman et al., 2016; Scharenbroch and Catania, 2012; Jim, 1998).

Post-treatment bulk density data revealed more about the ability of organic amendments to reduce compaction and about compaction's correlation with tree survival than penetration resistance data in this study. Analysis of bulk density data revealed treatment effects on soil compaction (Table 3.2) as well as lower average levels of compaction in living versus dead tree groups from all but the last tree survivorship observation time in the study (Tables 3.10, 3.11 and 3.12). In comparison, penetration resistance data only demonstrated differences in pre-treatment soil compaction between blocks (Table 3.1) and between living and dead tree groups one month after planting (Table 3.10).

When looking at differences between soil bulk density and soil strength (measured as penetration resistance) as indicators of soil compaction, it is important to consider that soil strength decreases significantly with increases in soil moisture content, which varies considerably (Weil and Brady, 2017). Soil strength is also largely determined by soil bulk density which—compared to soil strength—is relatively unchanged across time and moisture content (Gilker et al., 2002). Thus, while penetration resistance data provide a better measure of root inhibition from compaction in a single moment, soil bulk density data provide a better sense of how frequently and how long soil strength rises to root-inhibiting levels with changes in a site's soil moisture regime.

There are also important caveats to consider in interpreting penetration resistance data trends in this study. The pre-treatment penetration resistance readings were sampled on different days for each block, so differences in moisture content may have caused variability in the data that would not have been present if it were possible to take all site penetration resistance readings on the same day under more uniform soil moisture conditions. Additionally, the cone penetrometer is prone to showing more zero-value readings at levels of compaction that are below a certain detection threshold. Since compost-treated soils had lower levels of bulk density in the upper 10 cm following treatment application (Table 3.2), compaction readings at the 0 – 10 cm depth for the compost and mixed treatment groups may have been underrepresented in the post-treatment penetration resistance dataset.

Though there were no observable connections between soil treatments and tree survival in this study, trees in the biosolids treatment group had no tree leaf drop occur in the first month after planting; this was not true for any other treatment groups in the study. Stress from reduced nutrient uptake is a common component of transplant shock (Struve, 2009). Because biosolids are a richer source of immediately plant-available nutrients compared to compost, there may have been an important soil chemical variable—such as nutrient availability—influencing leaf drop that either was not measured in this study or that had insufficient sample size for an effect to be observed.

The observation that soil microbial respiration rates were consistently higher in dead tree groups than in living tree groups (Tables 3.10, 3.11, 3.12, and 3.13) further supports the notion of an important soil chemical driver influencing tree mortality that was either under-sampled or not included in this dataset. For example, available inorganic N pools may have been immobilized by the highly active soil microbial community, leading to N deficiency stress for trees (Weil and Brady, 2017). This phenomenon was observed by Somerville et al. (2018), who found that large compost additions led to temporary N immobilization, partly explaining the lack of improvements to tree survivorship in compost-treated soils compared to controls in their study.

Pre-treatment soil conditions were also correlated with tree survival, as pre-treatment soil organic matter content and electrical conductivity were higher for plots with living trees than those with dead trees in July 2019 (Table 3.10). Block differences in pre-treatment soil organic matter content also aligned with block differences in survival; tree mortality throughout the study was often lower in block A than in other blocks (Section 3.3.1), and block A had the lowest average pre-treatment soil organic matter content (Table 3.1). Pre-treatment penetration resistance was also lowest in block D (Table 3.1), where tree survival rates were highest for the entirety of the study (Section 3.3.1), though it is important to consider that soil moisture at the time of penetration resistance sampling was significantly higher for block D (Table 3.1), as penetration resistance decreases with increases in soil moisture.

4.5: Importance of Site-specific Conditions

Because of the amount of resource investment and public visibility attached to many urban afforestation projects, there is tremendous pressure placed on the initial planting success as well as the long-term sustainability of new urban forested natural areas (Hallett, 2013). The correlation between pre-treatment conditions and planted tree survival in this study suggests the importance of considering existing site conditions in urban restoration and afforestation. This is consistent with the suggestions of many other authors who have noted the importance of developing urban soil quality indices as site evaluation tools for site selection, goal setting, and management in future soil restoration and forestry work (Beniston et al., 2015; Scharenbroch and Catania, 2012; Scharenbroch et al., 2017; Layman et al., 2016; Jim, 1998; Hilbert et al., 2019).

Heterogeneous land use histories and environmental inputs across the urban landscape make the spatial variability of urban soil characteristics both dynamic and complex (Zukswert et al., 2021; Scharenbroch et al., 2005; Groffman et al., 2006; Pouyat et al., 2007). Urban afforestation initiatives, such as million trees initiatives in multiple cities (whatcommilliontrees.org; milliontreesnyc.org; miamidade.gov) and the five million trees program in Maryland (Thomas, 2021) will require plantings in areas with diverse land use histories and related soil conditions to meet their goals. For example, urban plantings from the five million trees program in Maryland will be offered to cities, counties, schools, nonprofit organizations, farmers, civic and faith-based groups, and private property owners (Thomas, 2021). Tree plantings aimed at improving environmental equity in west Baltimore are likely to utilize urban vacant lots—where soil conditions are diverse and heterogeneous at between-parcel as well as within-parcel scales—as planting sites (Shwe, 2021; Zukswert et al., 2021; Pouyat et al., 2010).

Urban soil heterogeneity and its consequences for native tree health is still largely understudied (Pregitzer et al., 2016) and will be an important priority going forward for research informing soil management for establishment and preservation of urban forested natural areas. Components of this new area of study should consider the role of the constructed environment as parent material shaping novel urban pedogenesis (Effland and Pouyat 1997; Huot et al., 2015), especially as recent work has demonstrated associations between novel anthropogenic soil types and native tree seedling growth and health (Pregitzer et al., 2016). Considering site soil pH is also of major importance. Soil pH management in urban forestry can often be reactive rather than proactive (Zukswert et al., 2021), which can contribute to poor tree survival outcomes when the planted species and site soil pH are poorly matched, as seen in this study. Quantifying pre-existing soil pH conditions at prospective tree planting sites is an important priority for site selection and soil management planning. Determining soil buffering capacity could also help to inform more effective soil pH management plans whenever interventions are needed; quantifying pre-treatment buffering capacity at the Beltsville study site could have allowed for a better-informed pH management approach in this study.

Just as there is a need for more research to inform the soil quality component of urban forestry planning, there is a need for research to inform species selection and links between urban growing conditions and individual tree species' responses (Calfapietra et al., 2015). Work on genotype x environment interactions has shown that such information can be effective for achieving faster canopy closure—a common desired outcome for urban forest establishment activities (Zalesny et al., 2014). Future work on species and planting site matching can help identify preferable species to act as pioneers and species to act as later successional species for urban planting efforts intending to establish mature forest (Zalesny et al., 2012). Similarly, Pregitzer et al. (2016) suggest that matching soil types with appropriate species can be valuable for optimizing survival and growth in urban forestry activities.

4.6. Need for Urban Soil Ecological Knowledge

There is still little soil management research from urban contexts available to inform practice compared with the research available from agricultural, non-urban forestry, and non-urban soil restoration contexts (Cogger, 2005; Jim, 1998). Common conditions of urban soils differ greatly from those common to agricultural, non-urban forestry and non-urban soil restoration contexts; greater fine-scale heterogeneity, higher compaction, anthropogenic materials, confined root space, and altered chemical soil environments common to urban soils present novel challenges common to urban soils that have not been addressed in research from non-urban contexts (Craul, 1985). Urban land management and soil restoration goals—which tend to focus on ecosystem services, ecosystem regulation, and local environmental quality—also differ greatly from objectives common in agriculture and non-urban forestry such as larger-scale food production, timber production and pollution remediation (Kumar and Hundal, 2016; Kim, 2016). While knowledge from these fields is valuable, more knowledge on the use organic amendments drawn from urban field studies is needed to create more context-specific recommendations for urban soil restoration and forestry by building on growing urban soil ecological knowledge.

Soil ecological knowledge is the combined knowledge of "physical, chemical and biological processes and properties to better understand and manage ecosystems, communities, and species' function and interactions" (Callaham et al., 2008, p. 1). This approach, which focuses on the role of soil within the context of ecological systems, is valuable for informing urban forestry soil management. Drastic anthropogenic changes imposed on urban soils can require drastic remediation actions that should be informed by an interdisciplinary and shared understanding of soil's role in the context of management goals and ecosystem—an understanding that soil ecological knowledge can provide (Smith et al., 2020).

There is still sparse literature exploring relationships between urban environmental factors—such as soil—and urban forest restoration and afforestation efforts; multiple authors have noted the lack of and need for more studies addressing this dynamic (Sonti et al., 2021; Doroski et al., 2018; Oldfield et al., 2014). Additionally, studies exploring relationships between soil quality and urban trees tend to focus on tree growth and nutrient status—rather than survival and mortality—following establishment (Scharenbroch and Watson, 2014; Guerrini et al., 2017; Scharenbroch, 2009; Layman et al., 2016) and there is a need for more studies examining associations between soil properties and planted tree survivorship (Hilbert et al., 2019). Soil ecological knowledge provides a powerful framework for understanding soil-plant dynamics in these future research areas.

4.7: Study Limitations and Implications

There are limitations to inferences from the results of this study for urban vacant lot soil management and urban forestry. For example, the study was conducted on a single site with uniform soil texture and parent material, only one tree species was planted, and the compost and biosolids used only represent one type of compost and one type of biosolid product, and only at specific application rates. Additionally, much of the tree mortality in this study occurred in the first growing season, resulting in a tree survivorship sample size that may have been too small for a treatment effect to be detected.

An ideal version of this experiment would have taken place with replications in multiple urban vacant lots with diverse soil properties including texture, pH, and parent materials, allowing for more patterns to be seen in the context of urban soil heterogeneity. Including more than one tree species in the study would also increase the power of the study for drawing inferences to inform urban forestry practice, providing knowledge on how different tree species respond to site soil conditions and organic amendments.

Exploring a variety of application rates that are the same for each amendment used would have also improved this study, allowing me to differentiate between the effects of amendment type and application rate; this was not possible with the design of this study.

Planting trees bare root and while still dormant would have been the preferred method for this study and is ideal for similar future studies. Planting practices that ensure greater tree survivorship in these experiments would allow studies to continue for more time after planting, collecting response data from a larger percentage of surviving trees over a period of multiple years. A study duration of at least 3 to 5 years after planting is ideal since it can take this long for trees to recover from transplant stress and become fully established in new environments (Gauthier et al., 2014). A multi-year study with higher overall tree survival would be more likely to reveal treatment effects on tree survival and other tree responses in its later years. Effects of treatments on soil quality, tree stress, and tree survival may become more detectable after the initial effects of transplant shock have subsided. There is also a general need for more data on the long-term success of urban afforestation efforts (Oldfield et al., 2013).

Despite its limitations, there is much to be learned from this study to inform future research on soil management for urban forestry as we continue to investigate organic amendments as a management tool. The correlation between tree survival and bulk density as well as organic matter early in the study (Tables 3.10, 3.11 and 3.12) supports the notion that soil compaction and organic matter content are still important considerations for management and research (Scharenbroch et al., 2013; García-Orenes et al., 2005; Larney and Angers, 2012).

This study also suggests that positive urban soil and tree responses to organic amendments reported in pot and greenhouse experiments (Scharenbroch et al., 2013; Chu et al., 2017; Ghosh et al., 2015; Pregitzer et al., 2016) might be different when the same questions are investigated in an urban field setting (Somerville et al., 2018; Scharenbroch, 2009)—though it is important to consider that the limited sample sizes and tree survival in this study may not have allowed for treatment effects to be detected. Containerized studies inherently place restrictions on rooting volumes and are limited in the extent to which they can replicate urban field conditions such as urban hydrology and soil heterogeneity. As we continue to pursue knowledge around the use of organic amendments for real-world urban forestry applications, it is crucial to investigate whether the effects we observe in more controlled experimental settings hold up when implemented in the field. In addition to extending these questions to more field settings, future work should apply these questions to more tree species, ages, and planting times following organic amendment applications, and should begin to report more tree survivorship and mortality responses to organic amendment treatments (Hilbert et al., 2019). The differences in the effects of compost and biosolids treatments on soil physical, chemical, and biological properties in this study highlight the importance of continuing to investigate amendment type, application rate, and application method in future studies.

The importance of amendment application rates, especially, can be seen from the results of this study, as we observed lesser impacts on soil physical and biological characteristics from biosolids, which were applied at a lower rate (Tables 3.2, 3.7, and 3.8). The compost application rate in this study (21.5 Mg/ha) was comparable to the 25 Mg/ha rate used for compost and biosolids applications in a similar urban-focused study by Scharenbroch et al., (2013) and the typical biosolids application rate range of 11 – 224 Mg/ha for forestry applications described by USEPA (2000). In contrast, the biosolids application rate used in this study (5 Mg/ha) was well below these rates (the biosolids application rate in this study was limited by concerns over excess nutrient runoff pollution, as discussed previously). Studies examining effects of amendments on plant growth have suggested that growth improvements can begin to be observed when a minimum level of 20 – 30% organic matter by volume is reached (Cox et al., 2001; De Lucia et al., 2013). In this study, the amendment application rates were not high enough to bring organic matter content to this level (Tables 3.7 and 3.8), and this suggested organic matter goal may be excessive for the main purpose of tree establishment (not growth) in this study. The importance of organic amendment application rate for achieving soil quality outcomes highlights the need for future studies to build a larger knowledge basis to inform application rate recommendations for urban soil restoration.

Lastly, this study suggests the importance of considering soil chemical variables in addition to compaction and organic matter content in urban soil management practice and research for forestry (Table 3.10), as is also suggested by Somerville et al. (2018). In this study, the soil chemical environment appeared to have an important impact on tree responses. Additions of soil organic matter can bring improvements to soil compaction and organic matter content, but they are still a major disturbance to the soil system (Lynch et al., 2006; Larney and Angers, 2012) and future research and practice should consider the temporary states—especially chemical states—of soil systems as they restabilize over time and what these changes mean for the tree root environment (Somerville et al., 2018; Weil and Brady, 2017). For example, organic amendments are often mixed into soil at the time of tree planting in current urban forestry practice, which may lead to temporary soil chemical challenges in the root environment such as limited nutrient availability from the sudden increase in microbial activity (Somerville et al., 2018). Providing supplemental plant nutrients when adding amendments at the time of planting is one suggestion for ensuring adequate available N for newly planted trees (Madrid et al., 2000). Alternatively, it may be helpful to increase the amount of time between amendment application and tree planting when possible, allowing the soil environment to stabilize and plant N availability to increase before planting (Somerville et al., 2018; Larney and Angers, 2012). Considering physical, chemical, and biological aspects of pre-existing site soil conditions is also a recommended practice moving forward as managers match tree species to sites and develop soil management goals and strategies (Scharenbroch et al., 2017; Layman et al., 2016; Hilbert et al., 2019).

Drawing on soil ecological knowledge will be critical for understanding changes in and interactions between physical, chemical, and biological aspects of the soil environment, especially when considering sudden soil fertility changes that impact trees (Heneghan et al., 2008; Scharenbroch and Catania, 2012; Chen et al., 2013). The consistent correlation between soil microbial respiration rates and tree survival in this study (Tables 3.10, 3.11, 3.12, and 3.13) suggests that soil microbial activity—most likely via its direct relationships with soil organic matter and nutrients—could be an important mechanism to explore in order to deepen our understanding of changes in the soil system over time following organic matter additions. This is supported by Scharenbroch et al., (2013) who found N availability and microbial respiration to be the strongest predictors of tree growth in a similar study. Knoepp et al., (2000) also found microbial respiration, along with microbial biomass and N mineralization, to be powerful in estimating soil quality and nutrient availability.

4.8: Conclusion

My objective in this thesis was to understand how two organic amendments, compost and biosolids, impacted newly planted tree survival, as well as physical, chemical, and biological aspects of soil quality. Additionally, I sought to identify which soil characteristics served as the strongest predictors of newly planted tree survival. To achieve this, I conducted an experiment at an urban proxy field site in which I applied compost and biosolids to soils, then planted trees and observed tree responses along with soil physical, chemical, and biological responses. I found that organic amendment applications affected some key soil response variables, including organic matter content, bulk density, and microbial respiration, and these effects were greater and longer lasting with compost applications than with biosolids applications. Tree mortality in this study was high largely due to transplant shock. Factors that may have contributed to this included tree extraction and planting practices, soil compaction, and soil chemical variables. The strong influence of transplant shock on tree survival in this study highlights the importance of considering both planting practices and soil quality in future urban afforestation field experiments. Despite low seedling survivorship, differences between living and dead tree groups throughout the study demonstrated that lower microbial respiration rates were characteristic of soils in surviving tree groups, and lower bulk density and higher organic matter content were also characteristic of living tree groups early in the study. Though soil organic amendments had no significant effect on tree survivorship and tree mortality in this study was high, the ability of compost treatments to decrease bulk density and increase organic matter content does suggest that organic amendments still have great potential as a solution to soil quality obstacles in urban forestry, and their use should continue to be explored with greater attention paid to chemical impacts on the soil environment in addition to physical and biological effects.

Future work should build and expand on our understanding of organic amendments as a solution for soil management in urban forestry by exploring these questions in urban field settings, by planting different tree species, by exploring more organic amendment types, application rates and application methods, and by including observations of tree mortality responses to organic amendments in study designs. Results of this study are consistent with the importance of managing against soil physical compaction and low soil organic matter content suggested in previous literature, as well as the utility of soil organic matter additions for doing so. Results from this study also suggest that it is important to consider soil chemical changes following organic matter additions and their effects on the root environment in future urban soil restoration and forestry research. Applying soil ecological knowledge to our understanding of soil system responses to organic amendment applications is also important as we continue to explore their use in urban soil restoration and forestry.

Future studies should consider soil physical, chemical, and biological variables both when designing treatments and when selecting measured response variables. Because of its strong connection to soil organic matter and nutrient cycling, soil microbial activity may be especially helpful for understanding the mechanisms behind soil environment changes from organic matter additions. Site soil conditions are crucial to consider for identifying soil management decisions, developing strategic soil management actions, and selecting tree species for planting that are best suited to site conditions. As we continue to explore solutions for soil quality obstacles to urban forestry, future studies should aim to illuminate soil ecological knowledge that can inform management recommendations tailored to local and site-specific urban challenges and identify key soil quality management priorities for urban forestry.

Appendix: Vendor-supplied chemical analyses of Leafgro compost and Bloom biosolids used in this study.

Leafgro

LABORATORY ANALYSIS

Comparison to Maryland Department of Agriculture's and EPA's acceptable ranges

Analyte	MDA Limits	Leafgro®
pH	6.0 –8.0	7.65
Metals	mg/kg dry wt.	mg/kg dry wt.
Arsenic	41	5.66
Cadmium	39	1.17
Chromium	1200	16.4
Copper	1500	47.9
Lead	300	25.4
Mercury	17	ND
Molybdenum	18	2.34
Nickel	420	10.1
Selenium	36	ND
Zinc	2800	177
Total Nitrogen		1.29 %
Phosphorous		0.25 %
Potassium		0.95 %

Western Branch
Prince George's County

Appendix A: Chemical composition of Leafgro compost (provided by product representative, June 2017).

ANALYTICAL CHEMISTS
and
BACTERIOLOGISTS
Approved by State of California

SOIL CONTROL LAB

42 HANGAR WAY
WATSONVILLE
CALIFORNIA
95076
USA

TEL: 831-724-5422
FAX: 831-724-3188
www.compostlab.com

Account #: 9060695-1/3-8679
Group: Jun19C #46
Reporting Date: July 8, 2019

DC Water & Sewer Authority
5000 Overlook Avenue, SW
Washington, DC 20032
Attn: James Fotouhi

Date Received: 19 Jun. 19
Sample Identification: BLM-Fresh
Sample ID #: 9060695 - 1/3

Nutrients	Dry wt.	As Rcvd.	units
Total Nitrogen:	4.8	1.4	%
Ammonia (NH_4-N):	4100	1200	mg/kg
Nitrate (NO_3-N):	< 1.0	< 0.3	mg/kg
Org. Nitrogen (Org.-N):	4.4	1.3	%
Phosphorus (as P_2O_5):	6.8	2.0	%
Phosphorus (P):	30000	8600	mg/kg
Potassium (as K_2O):	0.11	0.033	%
Potassium (K):	950	270	mg/kg
Calcium (Ca):	2.4	0.68	%
Magnesium (Mg):	0.32	0.092	%
Sulfate (SO_4-S):	460	130	mg/kg
Boron (Total B):	5.0	1.4	mg/kg
Moisture:	0	71.3	%
Sodium (Na):	0.042	0.012	%
Chloride (Cl):	0.048	0.014	%
pH Value:	NA	8.43	unit
Bulk Density :	14	50	lb/cu ft
Carbonates ($CaCO_3$):	28	8.0	lb/ton
Conductivity (EC5):	4.7	NA	mmhos/cm
Organic Matter:	63.6	18.3	%
Organic Carbon:	33.0	9.6	%
Ash:	36.4	10.4	%
C/N Ratio	7.0	7.0	ratio
AgIndex	> 10	> 10	ratio

Metals	Dry wt.	EPA Limit	units
Aluminum (Al):	4000	-	mg/kg
Arsenic (As):	4.4	41	mg/kg
Cadmium (Cd):	1.2	39	mg/kg
Chromium (Cr):	64	-	mg/kg
Cobalt (Co)	7.6	-	mg/kg
Copper (Cu):	440	1500	mg/kg
Iron (Fe):	87000	-	mg/kg
Lead (Pb):	33	300	mg/kg
Manganese (Mn):	530	-	mg/kg
Mercury (Hg):	< 1.0	17	mg/kg
Molybdenum (Mo):	13	75	mg/kg
Nickel (Ni):	26	420	mg/kg
Selenium (Se):	4.1	100	mg/kg
Zinc (Zn):	780	2800	mg/kg

*Sample was received and handled in accordance with TMECC procedures.

Stability Indicator:	
CO2 Evolution	Respirometery
mg CO_2-C/g OM/day	6.5
mg CO_2-C/g TS/day	4.1
Stability Rating	*moderately unstable*

Maturity Indicator: Cucumber Bioassay	
Compost:Vermiculite (v:v)	1:2
Emergence (%)	93
Seedling Vigor (%)	68
Description of Plants	*stunted*

Pathogens	Results	Units	Rating
Fecal Coliform	< 7.5	MPN/g	*pass*
Salmonella	< 3	MPN/4g	*pass*

Date Tested: 19 Jun. 19

Physical Contaminants**	% by weight
Total Plastic	< 0.1
Film Plastic	< 0.1
Glass	< 0.1
Metal	< 0.1
Sharps	ND
Total	< 0.5

Size Distribution	
MM	% by weight
> 50	0.0
25 to 50	0.0
16 to 25	8.4
9.5 to 16	35.3
6.3 to 9.5	17.5
4.0 to 6.3	12.8
2.0 to 4.0	14.0
< 2.0	12.0

**Greater than 4mm in size (Sharps greater than 2mm)

Analyst: Assaf Sadeh

Appendix B: Chemical analysis results for fresh Bloom biosolids published by D.C. Water in June 2019 (Bloomsoil.com).

Bibliography

Abrams, M. D. (2003). Where has all the white oak gone? *Bioscience, 53,* 927–939.

Amlinger, F., Peyr, S., Geszti, J., Dreher, P., Karlheinz, W., & Nortcliff, S. (2007). Beneficial Effects of Compost Application on Fertility and Productivity of Soils. Literature Study, Federal Ministry for Agriculture and Forestry, Environment and Water Management, Austria.

American Horticulture Industry Association. (2014). *American Standard for Nursery Stock*. www.AmericanHort.org

Angers, D. A., & Chenu, C. (1997). Dynamics of soil aggregation and C sequestration. In R. Lal, J. M. Kimble, R. F. Follett & B. A. Stewart (Eds.), *Soil Processes and the Carbon Cycle* (pp. 199–206). CRC Press.

Baballari, E. (2019). Bloom Biosolids: What is their microbial community and how do they affect soil and plant health? [Master's Thesis]. University of Maryland College Park.

Badalíková, B. (2010). Chapter 2: Influence of Soil Tillage on Soil Compaction. In A. Dedousis & T. Bartzanas (Eds.), *Soil Engineering* (Vol. 20, pp. 19–30). Springer, Berlin, Heidelberg.

Barkasi, A. M., Dadio, S. D., Losco, R. L., & Shuster, W. D. (2012). Urban soils and vacant land as a stormwater resource. World Environmental and Water Resources Congress 2012: Crossing Boundaries, Proceedings of the 2012 Congress, (May 2015), 569–579.

Barron, J. (2018, April 15). A Plan for New York City's Forests. Yes, Forests. *The New York Times*. https://www.nytimes.com/2018/04/15/nyregion/new-york-city-forests.html

Basta, N. T., Busalacchi, D. M., Hundal, L. S., Kumar, K., Dick, R. P., Lanno, R. P., & Granato, T. C. (2016). Restoring Ecosystem Function in Degraded Urban Soil Using Biosolids, Biosolids Blend, and Compost. *Journal of Environmental Quality, 45*(1), 74–83.

Beatty, R. A., & Heckman, C. T. (1981). Survey of urban tree programs in the United States. *Urban Ecology*, *5,* 81–102.

Belyaeva, O. N., & Haynes, R. J. (2009). Chemical, microbial and physical properties of manufactured soils produced by co-composting municipal green waste with coal fly ash. *Bioresour. Technol., 100*, 5203–5209.

Bendfeldt, E. S., Burger, J. A., & Daniels, W. L. (2001). Quality of Amended Mine Soils After Sixteen Years. *Soil Science Society of America Journal, 65*(6), 1736–1744.

Beniston, J. W., Lal, R., & Mercer, K. L. (2015). Assessing and Managing Soil Quality for Urban Agriculture in a Degraded Vacant Lot Soil. *Land Degrad. Develop.,* 1–11.

Blanco-Canqui, H., Shapiro, C. A., Wortmann, C. S., Drijber, R. A., Mamo, M., Shaver, T. M., & Ferguson, R. B. (2013). Soil organic carbon: The value to soil properties. *Journal of Soil and Water Conservation*, *68*(5), 129–134.

Bloom. (2020). *Bloom vs Leafgro Leaf Compost*. https://bloomsoil.com/bloom-vs-leafgro/

Boerner, R. E. (1984). Foliar nutrient dynamics and nutrient use efficiency of four deciduous tree species in relation to site fertility. *Journal of Applied Ecology*, *21,* 1029–1040.

Bouyoucos, G. J. (1962). Hydrometer method improved for making particle size analysis of soils. *Agronomy Journal*, *54*, 464–465.

Brenner, F. J., Werner, M., & Pike, J. (1984). Ecosystem development and natural succession in surface coal mine reclamation. *Minerals and the Environment*, *6*(1), 10–22.

Burdett, A. N., Simpson, D. G., & Thompson, C. F. (1983). Root development and plantation establishment success. *Plant and Soil, 71*, 103–110.

Calfapietra, C., Peñuelas, J., & Niinemets, Ü. (2015). Urban plant physiology: adaptation-mitigation strategies under permanent stress. *Trends Plant Sci*, *20*, 72–75.

Callaham, M. A., Rhoades, C. C., & Heneghan, L. (2008). A striking profile: Soil ecological knowledge in restoration management and science. *Restoration Ecology*, *16*(4), 604–607.

Chatigny, M. H., Angers, D. A., & Beauchamp, C. J. (1999). Aggregation and Organic Matter Decomposition in Soils Amended with De-Inking Paper Sludge. *Soil Science Society of America Journal*, *63*(5), 1214–1221.

Chen, Y., Day, S. D., Wick, A. F., & McGuire, K. J. (2014). Influence of urban land development and subsequent soil rehabilitation on soil aggregates, carbon, and hydraulic conductivity. *Science of the Total Environment*, 494–495.

Cheng, Z., & Grewal, P. S. (2009). Dynamics of the soil nematode food web and nutrient pools under tall fescue lawns established on soil matrices resulting from common urban development activities. *Applied Soil Ecology, 42*(2), 107–117.

Chu, S., Wu, D., Liang, L. L., Zhong, F., Hu, Y., Hu, X., & Zeng, S. (2017). Municipal sewage sludge compost promotes Mangifera persiciforma tree growth with no risk of heavy metal contamination of soil. *Scientific Reports*, *7*(1), 1–11.

City of Seattle (2013). Urban Forest Stewardship Plan. 79 p.

Claassen, V. P., & Carey, J. L. (2007). Comparison of slow-release nitrogen yield from organic soil amendments and chemical fertilizers and implications for regeneration of disturbed sites. *Land Degradation & Development, 18*(2), 119–132.

Close, D. C. (2001). Cold-induced photoinhibition, pigment chemistry, growth and nutrition of Eucalyptus nitens and E. globulus seedlings during establishment. [Doctoral dissertation]. University of Tasmania.

Close, D. C., Beadle, C. L., & Brown, P. H. (2005). The physiological basis of containerized tree seedling 'transplant shock': A review. *Australian Forestry*, *68*(2), 112–120.

Cogger, C. G. (2005). Potential compost benefits for restoration of soils disturbed by urban development. *Compost Science and Utilization, 13*(4), 243–251.

Corns, I. G. W. (1988). Compaction by forestry equipment and effects on coniferous seedling growth on four soils in the Alberta foothills. *Canadian Journal of Forest Research, 18*, 75–84.

Cox, D., Bezdicek, D., & Fauci, M. (2001). Effects of compost, coal ash, and straw amendments on restoring the quality of eroded Palouse soil. *Biology and Fertility of Soils, 33*(5), 365–372.

Craul, P. J. (1985). A description of urban soils and their desired characteristics. *Journal of Arboriculture, 11*, 330–339.

Davies, H., Doick, K., Handley, P., O'Brien, L., & Wilson, J. (2017). Delivery of ecosystem services by urban forests. Forestry Commission.

Day S. D., & Bassuk, N. L. (1994). A review of the effects of soil compaction and amelioration treatments on landscape trees. *Journal of Arboriculture, 20,* 9–17.

[a]Day, S. D., Wiseman, P. E., Dickinson, S. B., & Harris, J. R. (2010). Contemporary Concepts of Root System Architecture of Urban Trees. *Arboriculture and Urban Forestry*, *36*(4), 149–159.

[b]Day, S. D., Wiseman, P. E., Dickinson, S. B., & Harris, J. R. (2010). Tree Root Ecology in the Urban Environment and Implications for a Sustainable Rhizosphere. *Arboriculture & Urban Forestry, 36*(5), 193–205.

Dayal, U., & Allen, J. H. (1973). Instrumented Impact Cone Penetrometer. *Canadian Geotechnical Journal*, *10*(3), 397–409.

De Kimpe, C. R., & Morel, J .L. (2000). Urban Soil Management: A Growing Concern. *Soil Sci, 165*(1), 31–40.

De Lucia, B., Cristiano, G., Vecchietti, L., & Bruno, L. (2013). Effect of different rates of composted organic amendment on urban soil properties, growth and nutrient status of three Mediterranean native hedge species. *Urban Forestry and Urban Greening, 12*(4), 537–545.

Department of Environmental Protection and Sustainability (2013). Policy and Guidelines for Community Tree Planting Projects. Towson, MD.

Diaz-Sanz, J., Robert, S., & Keller, C. (2020). Parameters influencing run-off on vegetated urban soils: A case study in Marseilles, France. *Geoderma*, *376*, 0–2.

Doroski, D. A., Felson, A. J., Bradford, M. A., Ashton, M. P., Oldfield, E. E., Hallett, R. A., & Kuebbing, S. E. (2018). Factors driving natural regeneration beneath a planted urban forest. *Urban Forestry and Urban Greening, 29*, 238–247.

Effland, W. R., & Pouyat, R. V. (1997). The genesis, classification, and mapping of soils in urban areas. *Urban Ecosystems*, *1*, 217–228.

EPA Office of Superfund Remediation and Technology Innovation (OSRTI). (2007). The Use of Soil Amendments for Remediation, Revitalization, and Reuse. Enviromental Protection Agency, 1–27.

Farrish, K. W., Adams, J. C., & Vidrine, C. G. (1995). Survival and growth of planted loblolly pine seedlings on a severely rutted site. *Tree Planters Notes*, *46*, 28–31.

Freshkills Park Alliance. (n.d.) Afforestation Strategies. *Freshkills Park Alliance*. https://freshkillspark.org/scientific-research/afforestation-strategies

García-Orenes, F., Guerrero, C., Mataix-Solera, J., Navarro-Pedreño, J., Gómez, I., & Mataix-Beneyto, J. (2005). Factors controlling the aggregate stability and bulk density in two different degraded soils amended with biosolids. *Soil and Tillage Research, 82*(1), 65–76.

Gardner, W. H. (1986). Water Content. A. Klute (Ed.), *Methods of Soil Analysis Part 1: Physical and Mineralogical Methods*. Soil Science Society of America.

Gauthier, N. W., Kaiser, C., & Klahr, M. (2014). Transplant Shock: Disease or Cultural Problem? *University of Kentucky Cooperative Extension Service*. University of Kentucky College of Agriculture, Food and Environment.

Gee, G. W., & Bauder, J. W. (1986). Particle-size analysis in Methods of soil analysis. In A. Klute (Ed.), *Methods of Soil Analysis Part 1: Physical and Mineralogical Methods*. Soil Science Society of America.

Ghosh, S., Ow, L. F., & Wilson, B. (2015). Influence of biochar and compost on soil properties and tree growth in a tropical urban environment. *International Journal of Environmental Science and Technology, 12*(4), 1303–1310.

Gilker, R. E., Weil, R. R., Krizek, D. T., & Momen, B. (2002). Eastern gamagrass root penetration in adverse subsoil conditions. *Soil Science Society of America Journal, 66*(3), 931–938.

Gómez-Baggethun, E., Gren, A., Barton, D. N., Langemeyer, J., McPhearson, T., O'Farrel, P., and Kremer, P. (2013). Urban Ecosystem Services. In T. Elmvquist, M. Fragkias, J. Goodness, B. Günerlap, P. J. Marcotullio, R. I. McDonals, S. Parnell, M. Schewenius, M. Sendstad, K. C. Seto & C. Wilkinson (Eds.), *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment* (pp. 175–251). Springer.

Gong, Y., Cao, Q., & Sun, Z. (2003). The effects of soil bulk density, clay content and temperature on soil water content measurement using time-domain reflectometry. *Hydrological Processes, 17*(18), 3601–3614.

Grabosky, J., & Bassuk, N. (2016). Seventeen years' growth of street trees in structural soil compared with a tree lawn in New York City. *Urban Forestry and Urban Greening,* 16, 103–109.

Gregorich, E. G., Carter, M. R., Angers, D. A., Monreal, C. M., & Ellert, B. H. (1994). Towards a minimum data set to assess soil organic matter quality in agricultural soils. *Canadian Journal of Soil Science, 74*(4), 367–385.

Goldblum, D. (2010). The geography of white oak's (Quercus alba L.) response to climatic variables in North America and speculation on its sensitivity to climate change across its range. *Dendrochronologia, 28*(2), 73–83.

Groffman, P. M., Pouyat, R. V., Cadenasso, M. L., Zipperer, W. C., Szlavecz, K., Yesilonis, I. D., Band, L. E., & Brush, G. S. (2006). Land use context and natural soil controls on plant community composition and soil nitrogen and carbon dynamics in urban and rural forests. *For. Ecol. Manage. 236,* 177–192.

Grossman, R. B., & Reinsch, T. G. (2002). Bulk Density and Linear Extensibility. In J. H. Dane, & G. C. Topp (Eds.), *Methods of soil analysis, Part 4- Soil Science Society of America Book Series No. 5* (5th ed., pp. 201–228). Soil Science Society of America.

Grossnickle, S.C. (1988). Planting stress in newly planted jack pine and white spruce. II. Changes in tissue water potential components. *Tree Physiology, 4,* 85–97.

Guerrini, I. A., Croce, C. G. G., Bueno, O. de C., Jacon, C. P. R. P., Nogueira, T. A. R., Fernandes, D. M., & Capra, G. F. (2017). Composted sewage sludge and steel mill slag as potential amendments for urban soils involved in afforestation programs. *Urban Forestry and Urban Greening*, *22,* 93–104.

Hallett, R. A. (2013). Planting forests in NYC: Is the goal Restoration, Reforestation, or Afforestation? *Urbane Walder*, (16), 30–32.

Hamilton, W. D. (1989). Significance of root severance on performance of established trees. *Arboricultural Journal*, *13*(3), 249–257.

Hantz Farms Detroit. (n.d.). What We Have Done. *Hantz Farms*. http://www.hantzfarmsdetroit.com/whatwehavedone.html

Heneghan, L., Miller, S. P., Baer, S., Callaham, M. A., Montgomery, J., Pavao-Zuckerman, M., Rhoades, C. C., & Richardson, S. (2008). Integrating soil ecological knowledge into restoration management. *Restoration Ecology, 16*(4), 608–617.

Herrmann, D. L., Schifman, L. A., & Shuster, W. D. (2020). Urbanization drives convergence in soil profile texture and carbon content. *Environmental Research Letters*, *15*(11), 1–8.

Herrmann, D. L., Shuster, W. D., & Garmestani, A. S. (2017). Vacant urban lot soils and their potential to support ecosystem services. *Plant and Soil*, *413*(1–2), 45–57.

Hilbert, D. R., Roman, L. A., Koeser, A. K., Vogt, J., & van Doorn, N. S. (2019). Urban tree mortality: A literature review. *Arboriculture and Urban Forestry*, *45*(5), 167–200.

Huot, H., Simonnot, M. O., & Morel, J. L. (2015). Pedogenetic Trends in Soils Formed in Technogenic Parent Materials. *Soil Science, 180,* 182–92.

Jarvis, P. G., & Jarvis, M. S. (1963). The water relations of tree seedlings. II. Transpiration in relation to soil water potential. *Physiologia Plantarum, 16,* 236–253.

Jim, C. Y. (1998). Physical and chemical properties of a Hong Kong roadside soil in relation to urban tree growth. *Urban Ecosystems, 2*, 171–181.

Johnson, L. R., Johnson, M. L., Aronson, M. F., Campbell, L. K., Carr, M. E., Clarke, M., D'Amico, V., Darling, L., Tedward, E., Fahey, R. T., King, K., L., Lautar, K., Locke, D. H., Morzillo, A. T., Pincetl, S., Rhodes, L., Schmit, J. P., Scott, L., & Sonti, N. F. (2020). Conceptualizing social-ecological drivers of change in urban forest patches. *Urban Ecosystems, 125*, 254–270.

Kim, G. (2016). The Public Value of Urban Vacant Land: Social Responses and Ecological Value. *Sustainability (Switzerland), 8*(5), 1–19.

Kinzler, D. (2016). How to tell if a tree or shrub is dead or still dormant. *North Dakota State University Extension-Cass*. https://growingtogetherwithdonkinzler.wordpress.com/2016/04/26/how-to-tell-if-a-tree-is-dead-or-still-dormant/

Kelleher, C., Golden, H. E., Burkholder, S., & Shuster, W. (2020). Urban vacant lands impart hydrological benefits across city landscapes. *Nature Communications*, *11*(1), 1–12.

Knoepp, J. D., Coleman, D. C., Crossley, D. A., & Clark, J. S. (2000). Biological indices of soil quality: An ecosystem case study of their use. *Forest Ecology and Management, 138,* 357–368.

Kozlowski, T. T. (1999). Soil compaction and growth of woody plants. Scandinavian *Journal of Forest Research*, 14(6), 596–619.

Kube, P. (1993). Establishment techniques and early growth of eucalypt seedlings on a high elevation grass site in Tasmania. *Tasforests*, *5,* 63–75.

Kumar, V., Chopra, A. K., & Kumar, A. (2017). A Review on Sewage Sludge (Biosolids) a Resource for Sustainable Agriculture. *Archives of Agriculture and Environmental Science, 2*(4), 340–347.

Kumar, K., & Hundal, L. S. (2016). Soil in the City: Sustainably Improving Urban Soils. *Journal of Environmental Quality, 45*(1), 2–8.

Lal, R. (2015). Restoring soil quality to mitigate soil degradation. *Sustainability (Switzerland)*, *7*(5), 5875–5895.

Larney, F. J., & Angers, D. A. (2012). The role of organic amendments in soil reclamation: A review. *Canadian Journal of Soil Science, 92*(1), 19–38.

Larsen, H. S., South, D. B., & Boyer, J. N. (1988). Foliar nitrogen content at lifting correlates with early growth of loblolly pine seedlings from 20 nurseries. *Southern Journal of Applied Forestry, 12,* 181– 185.

Lautar, K., & Avins, M. (2017, October 17). Protect the forest patches that protect Baltimore. *Baltimore Sun*. https://www.baltimoresun.com/opinion/op-ed/bs-ed-op-1018-forest-patches-20171016-story.html

Layman, R. M., Day, S. D., Mitchell, D. K., Chen, Y., Harris, J. R., & Daniels, W. L. (2016). Below ground matters: Urban soil rehabilitation increases tree canopy and speeds establishment. *Urban Forestry and Urban Greening, 16*, 25–35.

Ledig, F. T. (1983). The influence of genotype and environment on dry matter distribution in plants. In Huxley, P.A. (Ed.), *Plant Research and Agroforestry* (pp. 427–454). International Council for Research in Agroforestry.

López, J. L., Delgado, D. F., Dema, A. V., Maria, S., Vale, G., João, M., & Valiente, Y. (2008). ¿Perduran En El Suelo Las Enmiendas Orgánicas? Análisis En Repoblaciones Forestales. *Cuadernos de La Sociedad Española de Ciencias Forestales, 25,* 281–286.

Lowery, B., & Morrison, J. E. (2002). Soil Penetrometers and Penetrability. In J. H. Dane & G. C. Topp (Eds.), *Methods of soil analysis, Part 4- Soil Science Society of America Book Series No. 5* (pp. 363–388). Soil Science Society of America.

Lu, Q., He, Z. L., & Stoffella, P. J. (2012). Land application of biosolids in the USA: A review. *Applied and Environmental Soil Science, 2012,* 1–11.

Lynch, D. H., Voroney, R. P., & Warman, P. R. (2006). Use of 13C and 15N natural abundance techniques to characterize carbon and nitrogen dynamics in composting and in compost-amended soils. *Soil Biology and Biochemistry, 38*(1), 103–114.

Mabuhay, J. A., Nakagoshi, N., & Isagi, Y. (2006). Microbial responses to organic and inorganic amendments in eroded soil. *Land Degrad. Devel. 17*, 321–332.

Madrid, F., Murillo, J. M., Lopez, R., & Cabrera, F. (2000). Use of urea to correct immature urban composts for agricultural purposes. *Communications in Soil Science and Plant Analysis, 31*(15–16), 2635–2649.

McInnes, K. J. (2002). Soil Heat. In J. H. Dane & G. C. Topp (Eds.), *Methods of soil analysis* (pp. 1183–1198). Soil Science Society of America.

McPhearson, T., Kremer, P., & Hamstead, Z. A. (2013). Mapping ecosystem services in New York City: Applying a social-ecological approach in urban vacant land. *Ecosystem Services*, *5*, 11–26.

Minckler, L. (1965). White oak (Quercus alba L.). In H. A. Fowells (Ed.), *Silvics of forest trees of the United States*. (p. 623-637). comp. U.S. Department of Agriculture, Agriculture Handbook, 271.

Minore, D., & Weatherly, H. G. (1990). Effects of site preparation on Douglas-fir seedling growth and survival. *Western Journal of Applied Forestry, 5,* 49–51.

Moll, G., & Ebenreck, S. (1989). *Shading our cities: a resource guide for urban and community forests*. American Forestry Association. Island Press.

Nehls, T., Rokia, S., Mekiffer, B., Schwartz, C., & Wessolek, G. (2013). Contribution of bricks to urban soil properties. *Journal of Soils and Sediments*, *13*(3), 575–584.

New York City Department of Parks and Recreation. (n.d.). Freshkills Park. https://www.nycgovparks.org/park-features/freshkills-park

Norby, R. J., O'Neill, E. G., & Luxmoore, R. J. (1986). Effects of atmospheric CO2 Enrichment on the growth and mineral nutrition of Quercus alba seedlings in nutrient-poor soil. *Plant Physiology, 82*(1): 83-89.

Nowak, D. J., Bodine, A. R., Hoehn, R. E., Low, S. C., Roman, L. A., Henning, J. G., Stephan, E., Taggart, T. & Endreny, T. A. (2016). *The urban forest of Philadelphia*. (Resource Bulletin NRS-106). U.S. Department of Agriculture, Forest Service, Northern Research Station.

[a]Nowak, D. J., & E. J. Greenfield. (2018). Declining urban and community tree cover in the United States. Urban For. *Urban Greening, 32,* 32–55.

[b]Nowak, D. J., & E. J. Greenfield. 2018. U.S. urban forest statistics, values and projections. *J. For., 116*(2), 164–177.

Nowak, D. J., Mcbride, J. R., & Beatty, R. A. (1990). Newly Planted Street Tree Growth and Mortality. *Journal of Arboriculture, 16*(5), 124–129.

Oldfield, E. E., Felson, A. J., Auyeung, D. S. N., Crowther, T. W., Sonti, N. F., Harada, Y., & Bradford, M. A. (2015). Growing the urban forest: tree performance in response to biotic and abiotic land management. *Restoration Ecology, 23*(5), 707–718.

Oldfield, E. E., Felson, A. J., Wood, S. A., Hallett, R. A., Strickland, M. S., & Bradford, M. A. (2014). Positive effects of afforestation efforts on the health of urban soils. *Forest Ecology and Management, 313,* 266–273.

Olson, N. C., Gulliver, J. S., Nieber, J. L., & Kayhanian, M. (2013). Remediation to improve infiltration into compact soils. *Journal of Environmental Management*, (117), 85–95.

Ow, L. F., & Ghosh, S. (2017). Urban tree growth and their dependency on infiltration rates in structural soil and structural cells. *Urban Forestry and Urban Greening*, *26*, 41–47.

Pavao-Zuckerman, M., & Pouyat, R. V. (2017). The effects of urban expansion on soil health and ecosystem services. In C. Gardi (Ed.), *Urban Expansion, Land Cover and Soil Ecosystem Services* (pp. 123–145). Routledge.

Phillips, T. H., Baker, M. E., Lautar, K., Yesilonis, I., & Pavao-Zuckerman, M. A. (2019). The capacity of urban forest patches to infiltrate stromwater is influenced by soil physical properties and moisture. *Journal of Environmental Management*, *246*, 11–18.

Popper, D., & Popper, F. (2010). Smart decline in post-carbon cities: The buffalo commons meets buffalo, New York. In R. Heinberg, & D. Lerch (Eds.), *The post* *carbon reader: Managing the 21st century's sustainability crises.* Watershed Media.

Pouyat, R. (2001). Short- and long-term effects of site factors on net N-mineralization and nitrification rates along an urban-rural gradient. *Urban Ecosystems*, *5*(3), 159–178.

Pouyat, R. V., & Trammell, T. L. E. (2019). Climate change and urban forest soils. In M. Busse, C. P. Giardina, D. M. Morris, D. S. Page-Dumroese (Eds.), *Global change and forest soils: Cultivating stewardship of a finite natural resource. Developments in Soil Science* (Vol. 36, pp.189–211). Elsevier.

Pouyat, R. V, Yesilonis, I. D., Groffman, P. M., Szlavecz, K., & Schwarz, K. (2010). Chemical, Physical and Biological Characteristics of Urban Soils. *Agronomy Monograph*, *55*, 119–152.

Pouyat, R. V., Yesilonis, I. D., Russell-Anelli, J., & Neerchal, N. K. (2007). Soil Chemical and Physical Properties That Differentiate Urban Land-Use and Cover Types. *Soil Science Society of America Journal, 71*(3), 1010.

Pouyat, R. V., Yesilonis, I. D., Szlavecz, K., Csuzdi, C., Hornung, E., Korsós, Z., & Giorgio, V. (2008). Response of forest soil properties to urbanization gradients in three metropolitan areas. *Landscape Ecology*, *23*(10), 1187–1203.

[a]Pregitzer, C. C., Ashton, M. S., Charlop-Powers, S., D'Amato, A., Frey, B. R., Gunther, B., Hallett, R. A., Pregitzer, K. S., Woodall, C. W., & Bradford, M. A. (2019). Defining and assessing urban forests to inform management and policy. *Environ. Res. Lett., 14*(2019)085002.

[b]Pregitzer, C. C., Charlop-Powers, S., Bibbo, S., Forgione, H. M., Gunther, B., Hallett, R. A., & Bradford, M. A. (2019). A city-scale assessment reveals that native forest types and overstory species dominate New York City forests. *Ecological Applications*, *29*(1), e01819.

Pregitzer, C., Sonti, N. F., & Hallett, R. A. (2016). Variability in Urban Soils Influences the Health and Growth of Native Tree Seedlings. *Ecological Restoration*, *34*(2), 106–116.

Pruim, R., Kaplan, D. T., & Horton, N. J. (2017). The mosaic Package: Helping Students to 'Think with Data' Using R. *The R Journal, 9*(1), 77–102.

R Core Team (2020). R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Reitveld, W. J. (1989). Transplanting stress in bareroot conifer seedlings: its development and progression to establishment. *Northern Journal of Applied Forestry, 6*, 99–107.

Robertson, S. (2011). Direct Estimation of Organic Matter by Loss on Ignition: Methods. *SFU Soil Science Lab*, 1–11.

Roman, L. A., Battles, J. J., & Mcbride, J. R. (2016, March). *Urban Tree Mortality: A Primer on Demographic Approaches*. United States Department of Agriculture.

Ros, M., Hernandez, M. T., & Garcia, C. (2003). Soil microbial activity after restoration of a semiarid soil by organic amendments. *Soil Biol. Biochem., 35*, 463-469.

Ross, D. S., Bailey, S. W., Lawrence, G. B., Shanley, J. B., Fredriksen, G., Jamison, A. E., & Brousseau, P. A. (2011). Near-surface soil carbon, carbon/nitrogen ratio, and tree species are tightly linked across Northeastern United States watersheds. *Forest Science*, *57*(6), 460–469.

Ryan, M. G., & Law, B. E. (2005). Interpreting, Measuring, and Modeling Soil Respiration. *Biogeochemistry*, *73*(1), 3–27.

Scharenbroch, B. C. (2009). A meta-analysis of studies published in arboriculture & urban forestry relating to organic materials and impacts on soil, tree, and environmental properties. *Arboriculture and Urban Forestry, 35*(5), 221–231.

Scharenbroch, B. C., Carter, D., Bialecki, M., Fahey, R., Scheberl, L., Catania, M., & Raciti, S. (2017). A rapid urban site index for assessing the quality of street tree planting sites. *Urban Forestry & Urban Greening, 27*, 279–286.

Scharenbroch, B. C., & Catania, M. (2012). Soil Quality Attributes as Indicators of Urban Tree performance. *Arboriculture and Urban Forestry, 38*(5), 214–228.

Scharenbroch, B. C., Lloyd, J. E., & Johnson-Maynard, J. L. (2005). Distinguishing urban soils with physical, chemical, and biological properties. *Pedobiologia*, *49,* 283–296.

Scharenbroch, B. C., Meza, E. N., Catania, M., & Fite, K. (2013). Biochar and Biosolids Increase Tree Growth and Improve Soil Quality for Urban Landscapes. *Journal of Environment Quality, 42*(5), 1372–85.

Scharenbroch, B. C., & Watson, G. W. (2014). Wood chips and compost improve soil quality and increase growth of Acer rubrum and Betula nigra in compacted urban soil. *Arboriculture and Urban Forestry*, *40*(6), 319–331.

Scharenbroch, B. C., Smiley, E. T., & Kocher, W. (2014). *Best Management Practices: Soil Management for Urban Trees.* International Society of Arboriculture.

Schuster, W. S., Griffin, K. L., Roth, H., Turnbull, M. H., Whitehead, D., & Tissue, D. T. (2008). Changes in composition, structure and aboveground biomass over seventy-six years (1930–2006) in the Black Rock Forest, Hudson Highlands, southeastern New York State. *Tree Physiology*, *28*(4), 537–549.

Scientific Engineering Response and Analytical Services (SERAS). (1994). Standard operating procedures:Plant Biomass Determination. *Scientific Engineering Response and Analytical Services*, 1–5.

Séré, G., Schwartz, C., Ouvrard, S., Sauvage, C., Renat, J. C., & Morel, J. L. (2008). Soil construction: a step for ecological reclamation of derelict lands. *Journal of Soils and Sediments, 8*(2), 130–136.

Shishiuchi, M., & Adachi, K. (1982). Influence of tractor logging on soil surface condition. I. Effect of soil compaction from tractors on the growth of planted Japanese larch seedlings. *Journal of the Japanese Forestry Society, 64,* 136–142.

Shuster, W. D., Dadio, S., Drohan, P., Losco, R., & Shaffer, J. (2014). Residential demolition and its impact on vacant lot hydrology: Implications for the management of stormwater and sewer system overflows. *Landscape and Urban Planning*, *125*, 48–56.

Shwe, E. (2021, July 15). Finding 'Tree Equity:' New Analysis Shows Neighborhoods That Need More Trees. *Maryland* Matters. https://www.marylandmatters.org/2021/07/15/a-push-for-tree-equity-new-analysis-shows-neighborhoods-that-need-more-trees/

Simmons, B. L., Hallett, R. A., Sonti, N. F., Auyeung, D. S. N., & Lu, J. W. T. (2016). Long-term outcomes of forest restoration in an urban park. *Restoration Ecology*, *24*(1), 109–118.

Six, J., Bossuyt, H., Degryze, S., & Denef, K. (2004). A history of research on the link between (micro)aggregates, soil biota, and soil organic matter dynamics. *Soil and Tillage Research*, *79*(1), 7–31.

Slattery, B., Reshetiloff, K., & Zwicker, S. (2003). Native Plants for Wildlife Habitat and Conservation Landscaping. Chesapeake Bay Watershed.

Smith, J., Hallett, R., & Groffman, P. M. (2020). The state factor model and urban forest restoration. *Journal of Urban Ecology*, *6*(1), 1–9.

Somerville, P. D., May, P. B., & Livesley, S. J. (2018). Effects of deep tillage and municipal green waste compost amendments on soil properties and tree growth in compacted urban soils. *Journal of Environmental Management, 227,* 365– 374.

Sonti, N. F., Hallett, R. A., Griffin, K. L., Trammell, T. L. E., & Sullivan, J. H. (2021). Chlorophyll fluorescence parameters, leaf traits and foliar chemistry of white oak and red maple trees in urban forest patches. *Tree Physiology*, *41*(2), 269–279.

Standards for the Use or Disposal of Sewage Sludge, 40 C.F.R. § 503.11 (2000).

Struve, D. K. (2009). Tree establishment: A review of some of the factors affecting transplant survival and establishment. *Arboriculture & Urban Forestry, 35*(1), 10–13.

Sullivan, D. M., Cogger, C. G., & Bary, A. I. (2015). Fertilizing with Biosolids. Pacific Northwest Extension Publication.

Thomas, C. (2021). Maryland's 5 Million Tree Program will Help Protect the Shore. Chesapeake Bay Foundation.

Thomas, G. W. (1996). Soil pH and soil acidity. In D. L. Sparks, A. L . Page, P. A. Helmke, R. H. Loeppert, P. N. Soltanpour, M. A. Tabatabai, C. T. Johnston, & M. E. Sumner (Eds.), *Methods of Soil Analysis Part 3: Chemical Methods* (pp. 475–490). Soil Science Society of America.

Treonis, A. M., Austin, E. E., Buyer, J. S., Maul, J. E., Spicer, L., & Zasada, I. A. (2010). Effects of organic amendment and tillage on soil microorganisms and microfauna. *Applied Soil Ecology, 46*, 103–110.

U.S. Bureau of Reclamation. (1991). *Engineering geology field manual*. US Department of the Interior, Bureau of Reclamation.

U.S. Composting Council. (2001). *Field guide to compost use*. U. S. Composting Council.

USDA-NRCS. (2021). Web Soil Survey.

U.S. Department of Agriculture Natural Resources Conservation Service. (2017). Mulching. Code 484. NRCS NHCP, Washington, DC.

U.S. Environmental Protection Agency. (2000). Biosolids technology fact sheet: Land application of biosolids. EPA 832-F-00-064. USEPA.

Weil, R., & Brady, N. (2017). *The Nature and Properties of Soils* (15th ed.). Pearson Education.

Werner, M. R., & Dindal, D. L. (1989). Earthworm community dynamics in conventional and low-input agroecosystems. *Revue d'Ecologie et de Biologie du Sol, 26*, 427–437.

Xiao Q., & McPherson, E. G. (2011) Performance of engineered soil and trees in a parking lot bioswale. *Urban Water J, 8*, 241–253.

Yang, J., & McBride, J. (2003). A Unique Technique for Street Tree Planting in Beijing. *Arboricultural Journal, 27*(1), 1–10.

Yeomans, J. C., & Bremner, J. M. (1991). Carbon and nitrogen analysis of soils by automated combustion techniques. *Communications in Soil Science and Plant Analysis*, 22(9-10), 843–850.

Zalesny, R. S. Jr., Hallett, R. A., Falxa-Raymond, N., & Grassi, C. (2012). *Improving ecosystem health and functioning through phytotechnologies at Freshkills Park, Staten Island, New York,* USA. 9th Conference of the International Phytotechnology Society: Phytotechnologies—plant-based strategies to clean water, soil, air and provide ecosystem services, Hasselt University, Diepenbeek, Belgium.

Zalesny, R. S., Hallett, R. A., Falxa-Raymond, N., Wiese, A. H., & Birr, B. A. (2014). Propagating native Salicaceae for afforestation and restoration in New York City's five boroughs. *Native Plants Journal*, *15*(1), 29–41.

Zukswert, J. M., Hallett, R., Bailey, S. W., & Sonti, N. F. (2021). Using regional forest nutrition data to inform urban tree management in the northeastern United States. *Urban Forestry and Urban Greening*, *57*, 1–17.