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Vacant lot plant establishment techniques alter urban soil ecosystem services

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ABSTRACT

Vacant residential lots are ubiquitous in cities. While there is increasing interest in enhancing the aboveground habitat and biodiversity of vacant lots via restoration, vacant lot restoration may also affect the properties of and ecosystem services provided by soil. We assessed the effects of four vacant lot plant community establishment techniques (seed bombing, broadcast seeding, plug planting, and intensive gardening) and unaltered lawn on three critical ecosystem services provided by urban soils: carbon sequestration, nutrient retention, and water infiltration. We found that aboveground-focused treatments had belowground consequences. Consistent with other "urban grassland" studies, lawns exhibited the highest carbon storage among our treatments. However, soil carbon may increase in our other treatments over time – a common phenomenon in disturbed urban soils. We also found that nutrient retention – particularly nitrogen retention – increased with treatment intensity, likely due to increased plant uptake and microbial immobilization in our plots with prairie plantings. Finally, our most investment-intensive treatment, intensive gardening, resulted in decreased water infiltration, likely due to soil disturbance and increased bare soil resulting from frequent watering and weeding. Thus, treatments did not have consistent positive or negative effects on soil ecosystem services, emphasizing the multifunctionality and trade-offs associated with urban soil ecosystem processes. However, assuming low soil carbon and organic matter pools in our broadcast seeding and plug planting treatments recover over time, these two treatments may optimize aboveground plant community establishment and belowground ecosystem service provision in urban vacant lots.

1. Introduction

Despite an increase in the extent of urban land and the proportion of people who live in urban areas worldwide (Seto et al., 2011; United Nations, 2019), most post-industrial American cities have substantial vacant land (Bowman and Pagano, 2004). For instance, in the Mid-western USA, about 21 % of urban land area is vacant (Newman et al., 2016), and in Chicago, IL (USA), where our study took place, the city owns approximately 780 ha of vacant land (Minor et al., 2018). While vacant lots encompass many types of former land use – ranging from the yards of unoccupied homes to severely contaminated brownfields – many vacant lots are formerly residential sites where buildings have been demolished (Schilling and Logan, 2008). In vacant lots that were never built or where structures have been razed, the processes of urban development and divestment profoundly disturb the soil (Chen et al., 2013; Herrmann et al., 2017). Land management activities such as fertilization and irrigation further influence the historical legacies and current ecologies of vacant lots (Beniston et al., 2016; Pouyat et al., 2009). Despite these disturbances, vacant lots have the ability to provide critical ecosystem services including supporting plant growth, storing carbon, and retaining stormwater (Anderson and Minor, 2019; Grewal et al., 2011; Herrmann et al., 2017). The ecosystem services provided by vacant lots may be amplified if native plant communities are restored (Anderson and Minor, 2017; Kim, 2016). For instance, "greened" lots may enhance habitat connectivity throughout cities; increase plant, bird, and insect biodiversity; mitigate urban heat island effects; and improve human well-being (Anderson and Minor, 2017; Kim, 2016).

Recent studies of the ecosystem services provided by restored vacant lots have largely focused on those driven by increased vegetation abundance and plant diversity (Anderson and Minor, 2017). However,

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as the "brown" infrastructure of cities, urban soils provide important climate-, water-, and atmosphere-regulating services including carbon (C) sequestration, nutrient retention, and water infiltration. In Chicago, residential urban soils store four times more C than vegetation (Jo and McPherson, 1995), and urban soils often sequester more C than those in native and cultivated ecosystems (Pouyat et al., 2002; Raciti et al., 2011). Urban ecosystems are also nitrogen (N) "hot spots" due to high levels of atmospheric N deposition and fertilizer application, which can generate nitrous oxide (N_2O), a potent greenhouse gas, and lead to runoff and N-induced eutrophication of waterways (Decina et al., 2020; Groffman et al., 2009; Lovett et al., 2000; Smith et al., 2018). Urban soils play a critical role in the retention of this N and other nutrients, such as phosphorus (P) (Hobbie et al., 2017). Finally, cities commonly face hydrologic issues, including flooding and combined stormwater overflow, due to high impermeable surface cover. In simulations of a Chicago watershed, even 10 % permeable surface coverage mitigated small storm flood risk (Zellner et al., 2016), suggesting that soils in vacant lots may play an important role in water infiltration. Given that soils connect aboveground vegetation diversity and structure with belowground processes and functions, it is paramount to consider urban soils when evaluating the ecosystem services provided by vacant lots.

Fine-scale plant management and restoration techniques modify aboveground habitat and community structure, which in turn affect the ecosystem services provided by soils (Byrne, 2007). Plants alter soil properties via the quantity and quality of their litter inputs as well as through nutrient uptake and evapotranspiration. For instance, soil in restored "prairie gardens" in Madison, WI (USA), tended to have higher organic matter content and water infiltration capacity than adjacent lawns (Johnston et al., 2016). Similarly, in experimental rain gardens, prairie vegetation increased nutrient retention and reduced runoff compared to turfgrass (Nocco et al., 2016). In addition, many soil properties are purposefully managed in urban ecosystems (e.g., organic matter levels via mowing or top dressing with compost, soil moisture via irrigation, soil pH via liming, and nutrient availability via fertilization), primarily with aboveground outcomes in mind. In vacant lots in Youngstown, OH (USA), different amendments and urban agricultural management techniques led to variable soil quality and plant yield (Beniston et al., 2016). In general, plot-scale garden management has a greater effect on soil properties and functions than land use classification or city-scale urbanization intensity (Tresch et al., 2019, 2018; Ziter and Turner, 2018). As such, restoration treatments and management decisions in vacant urban lots may have large-scale consequences for soil ecosystem service provision.

Vacant lot management and restoration techniques range from low-intensity (e.g., cessation or changes in mowing regimes) to high-intensity (e.g., gardening). In a previous study (Anderson and Minor, 2020), we assessed the effects of four vacant lot plant community establishment techniques – seed bombing, broadcast seeding, plug planting, and intensive gardening (detailed below) – on native prairie plant community composition and structure. In the current study, we further examine the effects of these techniques on three soil ecosystem services: C sequestration, nutrient retention, and water runoff regulation. We also assess the ecosystem services provided by soils under maintained lawns adjacent to the treatments. Here, we ask (1) Do plant establishment techniques affect soil ecosystem services? (2) Are effects consistently positive or negative among ecosystem services? Given that different prairie restoration techniques have been found to alter soil processes in former cropland (Bach et al., 2012), we hypothesized that these different vacant lot plant establishment strategies would have variable effects on soil ecosystem services. Specifically, we hypothesized that planting techniques that effectively establish native prairie plant communities (broadcast seeding, plug planting, and intensive gardening) would enhance soil C storage, nutrient retention, and water infiltration capacity relative to less effective techniques (seed bombing) and lawn (Johnston et al., 2016; Nocco et al., 2016). As such, soil ecosystem services would be positively associated with one another and plant community establishment success.

2. Methods

2.1. Site description

Our experiment took place on the campus of the University of Illinois at Chicago (UIC), which is in the center of Chicago, IL (USA). Chicago is a major city with a temperate climate that experiences four seasons including cold, snowy winters (average −8 °C – 0 °C) and hot, humid summers (average 16 °C – 28 °C, U.S. National Weather Service, 2020). Historically, this area was dominated by tallgrass prairie, but this native ecosystem has been almost completely extirpated from the landscape over the past 200 years due to intensive agriculture and urbanization. The UIC campus was established in its current location in 1965, and the experiment occurred on land that has been unbuilt since at least 1999.

We established eight replicate test gardens in mid-May 2015. Plots were located on two different parts of UIC campus (Anderson and Minor, 2020): four replicates were on the grounds of the Plant Research Laboratory, and the rest were approximately 125 m away in a vacant space adjacent to a parking garage. A map and diagram of the experiment are available in Anderson and Minor (2020). Soils in both locations are broadly classified as "Urban Land" and are shallow, nearly level loamy to clayey Orthents (Soil Survey Staff, Natural Resources Conservation Service, 2020). Original soils were derived from glacial lake plains and ground moraines. Both sites had previous or contemporary native gardens nearby, but our test gardens were established in turf areas in full or almost-full sun.

2.2. Experimental treatments

Each 5.5 m × 5.5 m test garden contained 4 subplots measuring 2 m × 2 m, each surrounded and separated by a 0.5 m weeded and ground-covered buffer. Within a garden, each subplot received one of four experimental treatments designed to establish native plant communities: seed bombing, broadcast seeding, plug planting, or intensive gardening. These methods were selected to represent a gradient of cost and time input, with seed bombing representing the treatment with the lowest investment of resources and intensive gardening representing the treatment with the highest investment of resources.

We left the existing turfgrass in place in one subplot of each garden (for the 'seed bombing' treatment), but in the other three subplots we removed all the turfgrass and tilled the upper 15–20 centimeters of the soil with a conventional rototiller. For the broadcast seeding, plug planting, and intensive gardening treatments, we incorporated 10 kg of hardwood sawdust obtained from a local pallet manufacturer into the top ~30 cm of soil in each of the three tilled subplots to increase the soil C:N ratio and potentially reduce weed growth (Corbin and D'Antonio 2004).

Each plot was bombed or sowed (as seeds) or planted (as plugs) with an equal proportion of eight native species: *Rudbeckia hirta* L., *Dalea candida* Michx. ex Willd., *Echinacea pallida* (Nutt.) Nutt., *Echinacea purpurea* (L.) Moench, *Solidago rigida* L., *Helianthus occidentalis* Riddell, *Panicum virgatum* L., and *Asclepias tuberosa* L.. We created 5 cm diameter seed bombs by combining 50 g of Crayola™ air-dry clay, 10 g of organic potting soil (MiracleGro Nature's Care with Water Conserve™), and 8–10 stratified seeds of each species. We sun-dried seed bombs for 3–4 days and dropped 20 seed bombs into each 'seed bombing' subplot. We broadcast 200 stratified seeds of each species by hand into the 'broadcast seeding' subplots and planted six plugs of each species evenly throughout each of the 'plug planting' and 'intensive gardening' subplots.

All treatments were regularly watered with 30 L of water every 2–3 days for the first 4 weeks of growth in 2015 to aid in plant recruitment and establishment, but after mid-July we watered only the intensively gardened subplots 1−2 times per week. The intensively gardened treatment was also regularly weeded (every 2–3 weeks) during the summer of 2015 and weeded twice each summer (in early June and mid-July) in 2016 and 2017. The lawn areas surrounding the experimental gardens were occasionally mowed, but experienced little foot traffic throughout the experiment. Additional design details can be found in Anderson and Minor (2020).

2.3. Soil sampling and processing

We collected soil samples from each subplot on July 5, 2017. We collected two sets of soil samples: soil probe samples, which we sieved and used for soil chemical and biological analyses, and soil core samples, which remained undisturbed for soil physical analyses. We extracted and composited four, 10 cm deep samples from each subplot with a 2 cm diameter soil probe. We also took one 5 cm diameter, 10 cm deep soil core from the middle of each subplot. We additionally collected soil probe and core samples in a similar manner from turfgrass areas ~1 m south of each replicate garden. Soil probe samples were sieved through a 2 mm sieve to remove rocks and roots and to homogenize the soil and were then stored at 4 °C prior to conducting biological and chemical analyses. Soil core samples were air-dried at room temperature (~23 °C) for 4 weeks prior to conducting physical analyses.

2.4. Soil analyses

We used several biological, chemical, and physical soil properties as proxies for three soil ecosystem services: C sequestration, nutrient retention, and water infiltration.

2.4.1. Carbon sequestration indicators

We used soil C concentration, organic matter concentration, aggregate distribution, and aggregate stability as indicators of C sequestration. Larger soil C and organic matter pools indicate greater ecosystem C storage, and greater proportions of large, stable aggregates indicate that soil C pools are less susceptible to decomposition and CO_2 emission.

To quantify organic matter content, we oven-dried 4−5 g subsamples of soil at 105 °C and subsequently ashed them in a muffle furnace at 450 °C for 16 h (Nelson and Sommers, 1996). To measure total C, we oven-dried soil subsamples at 55 °C, ground them to a fine powder, and determined total C concentrations by dry combustion (Nelson and Sommers, 1996; Vario El III, Elementar, Lengenselbold, Germany).

We used air-dried soil core samples to quantify aggregate distribution and wet aggregate stability (Nimmo and Perkins, 2002). We separated aggregates using a rotary sieve and sieves of the following sizes: 8 mm, 4 mm, 2 mm, 1 mm, 0.5 mm, 0.25 mm, and 0.053 mm. We removed non-soil particles (such as rocks and pieces of concrete) from the samples and used soil weights of each size fraction to calculate the proportion of the total soil weight in each size class. Because large clods of soil were frequently found in the 8 mm size class, we excluded the 8 mm size class from our analysis. We calculated mean aggregate size by summing the products of the mean diameter of each size fraction and the proportion of the total sample weight occurring in the corresponding size fraction. For statistical analysis, we assigned each of the seven size fractions to one of four aggregate size classes: large macroaggregates (2−8 mm), small macroaggregates (0.25−2 mm), microaggregates (0.53−250 μm), or clay particles (<0.53 μm).

We measured wet aggregate stability of aggregates in the 1−2 mm size fraction by gently wetting 4 g of evenly spread soil on a 60 mesh screen using capillary action, wet sieving the soil over cans for 10 min (stroke is 1.3 cm at 34 times/min) using a wet aggregate stability tester, and wet sieving the soil remaining on the screen in a can filled with 100 mL of 0.003 M $(NaPO_3)_6$ dispersing solution for several hours until only roots and sand particles remained. We calculated the fraction of wet-stable aggregates as the weight of soil obtained in the dispersing solution divided by the sum of the weights obtained in the dispersing solution and the water.

2.4.2. Nutrient retention indicators

We used soil inorganic N and P pools as indicators of nutrient retention. High soil inorganic N and P concentrations are indicative of potential nutrient runoff into water bodies and N_20 greenhouse gas emissions; as such, lower soil nutrient concentrations are indicative of higher nutrient retention.

To measure inorganic N pools, we extracted inorganic N (ammonium and nitrate) from 4 g soil subsamples with 2 M KCl. We quantified ammonium concentrations using the salicylate-nitroprusside method (Sims et al., 1995) and measured absorbance at 660 nm on a microtiter plate reader (Synergy HTX, Biotek, Winooski, VT). We quantified nitrate concentrations using the VCl_3/Griess method (Hood-Nowotny et al., 2010) and measured absorbance at 540 nm on a microtiter plate reader. Total inorganic N is the sum of ammonium and nitrate.

To measure inorganic P pools, we conducted a partial Hedley fractionation. We sequentially extracted P from 5 g soil subsamples with H_2O (H1; inorganic resin P) and 0.5 M $NaHCO_3$ (H2; bicarbonate P) (Hedley and Stewart, 1982). Though the bicarbonate P pool contains both inorganic and organic P, we only analyzed the inorganic pool. We quantified phosphate concentrations in resin and bicarbonate solutions using the ammonium molybdate-ascorbic acid method (Kuo, 1996; Shaw and DeForest, 2013) and measured absorbance at 880 nm on a microtiter plate reader. Total inorganic P is the sum of resin and bicarbonate P.

2.4.3. Water infiltration indicators

We used bare soil and aggregate stability as indicators of water infiltration. Large areas of bare soil and low aggregate stability are indicators of poor water infiltration capacity. Bare soils commonly form a hydrophobic crust, which make them more susceptible to aggregate instability and leads to reduced water infiltration. When aggregates break apart, small clay particles clog pores in the soil reducing pathways in the soil profile though which water could travel (Abid and Lal, 2009).

In mid-September and early October 2017, we estimated the percentage of bare soil in each subplot by dividing each subplot into 0.5 m × 0.5 m quadrants. We converted this estimation to an area per 0.25 m^2 and summed these 16 estimates as a total bare soil measurement for each subplot.

2.4.4. Additional soil properties

In addition to the soil properties we used as indicators of ecosystem services, we quantified soil C:N, pH, and microbial biomass C because these are “master variables” that shape soil C and nutrient cycling. Soil C:N and pH drive biological and chemical nutrient retention and release dynamics (Gundersen et al., 1998; Price, 2006; Ste-Marie and Paré, 1999), and microbial biomass is a precursor to both soil C sequestration and aggregate formation (Cotrufo et al., 2013).

To measure soil pH, we placed 5 g of dry weight-equivalent soil subsamples into a 50 mL centrifuge tube with 40 mL of 0.01 M $CaCl_2$ solution. The suspensions were shaken for 1 h and vortexed immediately prior to analysis. We measured soil pH using a bench top electrode pH meter (Orion 5 Star, Thermo Scientific, Beverly, MA, USA). We determined total soil N by dry combustion concurrently with total soil C (Bremner, 1996) and subsequently calculated C:N ratios.

We measured microbial biomass C by quantifying changes in extractable pools of C after 4 days of chloroform fumigation (Vance et al., 1987). We extracted organic C with 0.5 M K_2SO_4 from 10 g soil subsamples that were either fumigated with chloroform or unfumigated and quantified total organic C concentrations in extracts with high temperature oxidation (1010 TOC analyzer, OI Analytical, College Station, Texas). Soil microbial biomass C is the difference between the concentrations of total organic C in the fumigated and unfumigated subsamples. We adjusted our microbial biomass C values to reflect an extractability of 45 % (Beck et al., 1997).

2.5. Statistical analyses

We used mixed linear models with treatment (lawn, seed bombing, broadcast seeding, plug planting, or intensive gardening) as a fixed effect, garden (1–8) as a random effect, and the above 15 variables of interest as dependent variables to evaluate the effects of treatments on soil properties. All statistical analyses were conducted in R (R Development Core Team 4.0.0., 2020) using the lmertest package for mixed linear models and lsmeans package for *post-hoc* Tukey tests (Kuznetsova et al., 2017; Lenth, 2016). Our figures were created with ggplot2 (Wickham, 2016).

3. Results

3.1. Carbon sequestration indicators

Plant establishment treatments affected many of the soil properties we measured. Specifically, organic matter concentrations were lower in the broadcast seeding, plug planting, and intensive gardening soils than the lawn soils ($P = 0.002$; Fig. 1a). Similarly, total C concentrations were lower in the broadcast seeding and plug planting soils compared to the lawn soils ($P = 0.011$; Fig. 1b).

Physical soil properties also varied among treatments (Fig. 2). While treatments had no effects on the relative abundance of soil aggregates in each size class ($P \geq 0.270$; Fig. 2a) or mean aggregate size (4.07 ± 0.22 mm on average; $P = 0.740$), plant establishment techniques did alter aggregate stability ($P < 0.001$; Fig. 2b). Specifically, intensive gardening decreased aggregate stability compared to all other treatments and lawn.

3.2. Nutrient retention indicators

Soil inorganic N pools varied among treatments, while inorganic P pools did not (Fig. 3). Inorganic N pools tended to decrease with treatment intensity ($P = 0.0175$; Fig. 3a and b). Ammonium pools were significantly lower in the broadcast seeding, plug planting, and intensive gardening treatments relative to the lawn ($P < 0.001$) while nitrate pools were lower in the broadcast seeding soils than the lawn soils ($P = 0.023$). However, since nitrate pools were orders of magnitude greater than ammonium pools, only broadcast seeding treatments had significantly

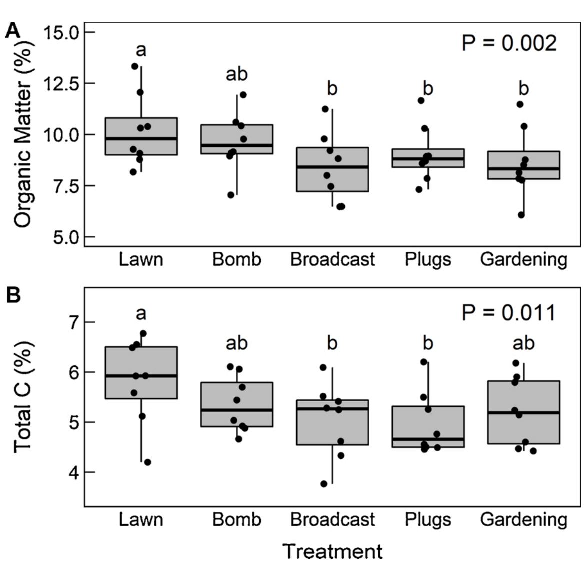

Fig. 1. Boxplots of soil (a) percent organic matter and (b) percent carbon in vacant lot restoration treatments and adjacent unrestored lawns at the University of Illinois at Chicago. Different letters represent significantly different groups based on Tukey's HSD pairwise comparisons at $\alpha = 0.05$.

lower total inorganic N pools relative to lawn ($P = 0.014$; all other *post-hoc* pairwise comparisons $P > 0.05$). In contrast, treatments had no effects on the H1 or H2 inorganic P pools or their sum ($P \geq 0.167$; Fig. 3c and d).

3.3. Water infiltration indicators

Percent cover of bare soil was low overall but varied among treatments; intensive gardening led to more bare soil than any other planting treatments, but the result was not statistically significant ($P = 0.070$). Intensive gardening similarly had the lowest soil aggregate stability compared to all other treatments and lawn, as mentioned above (Fig. 2b).

3.4. Additional soil properties

While soil C:N did not vary among our treatments ($P = 0.220$; Fig. 4a) and treatments had little effect on microbial biomass, pH increased with treatment intensity (Fig. 4). Microbial biomass differed among treatments ($P = 0.049$) and tended to decrease with treatment intensity (Fig. 4b), although the *post-hoc* test found no detectable differences among treatments. In contrast, broadcast seeding, plug planting, and intensive gardening treatments had higher pH soils than those found under lawns ($P = 0.008$; Fig. 4c).

4. Discussion

In urban areas, converting turfgrass-dominated lots to vegetable or prairie gardens often alters soil properties and ecosystem services (Beniston et al., 2016; Grewal et al., 2011; Johnston et al., 2016; Nocco et al., 2016). However, little is known about the effects of different plant establishment techniques on soil functioning in an urban setting. In this study, we assessed the effects of four vacant lot plant community establishment techniques on three soil ecosystem services: C sequestration, nutrient retention, and water runoff regulation. As expected, we found that different vacant lot plant establishment treatments had variable effects on soil properties and associated ecosystem services. However, soil ecosystem services were not positively associated with one another or plant community establishment success. As plant establishment treatment intensity increased, some soil ecosystem services tended to decrease, particularly soil C sequestration and water infiltration. In contrast, ecosystem nutrient retention – inorganic N retention, in particular – increased with treatment intensity.

4.1. Carbon sequestration

Our soil C sequestration metrics – organic matter concentrations, C concentrations, and aggregate distributions and stability – indicate that lawn soils have the greatest potential for C sequestration, while intensive gardening treatments have the lowest capacity to provide C sequestration ecosystem services. This observation is consistent with myriad other studies that have compared C pools in urban lawns to those in natural ecosystems (e.g., Golubiewski, 2006; Raciti et al., 2011; Smith et al., 2018). However, most other studies have compared urban lawns to natural ecosystems in exurban areas, and urban-rural gradient studies indicate that urbanization has a large effect on soil C, even under similar land-use types (Pouyat et al., 2009). In contrast, our study demonstrates that larger soil C pools under lawns manifest even at very fine spatial scales within an urban context.

Soil C sequestration is a function of organic matter inputs created by net primary production and organic matter decomposition and respiration by microbes. As such, there are two likely mechanisms driving higher soil organic matter and C pools in the lawn plots compared to the plant establishment treatments: high grass productivity and limited soil disturbance. High levels of soil C under lawns are often attributed to high net primary productivity of turfgrass, which is enhanced by

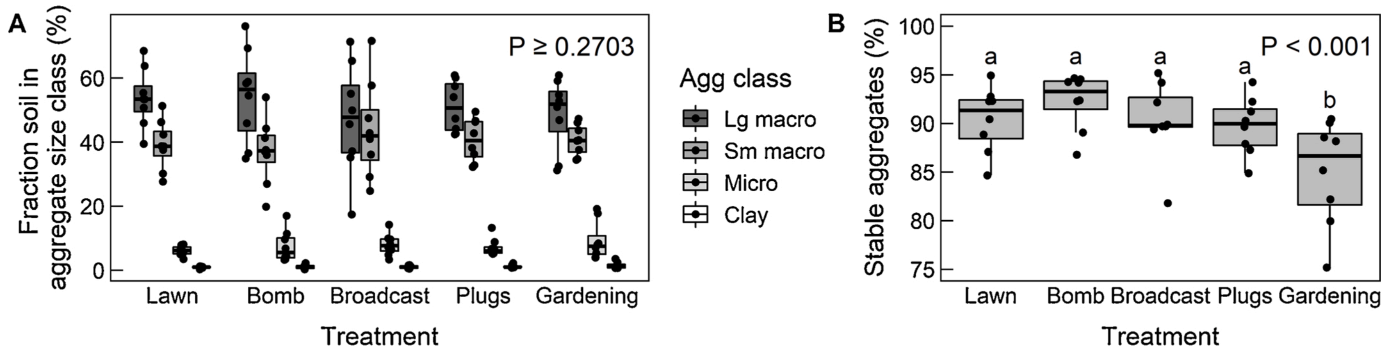

Fig. 2. Boxplots of the (a) relative abundance of aggregates in several size classes and (b) the fraction of stable aggregates in the 1-2 mm aggregate size class in vacant lot restoration treatments and adjacent unrestored lawns at the University of Illinois at Chicago. In (a), four size classes are presented: large macroaggregates (2-8 mm), small macroaggregates (0.25-2 mm), microaggregates (0.53-250 μm), and clay particles (<0.53 μm). Different letters represent significantly different groups based on Tukey's HSD pairwise comparisons at $\alpha = 0.05$.

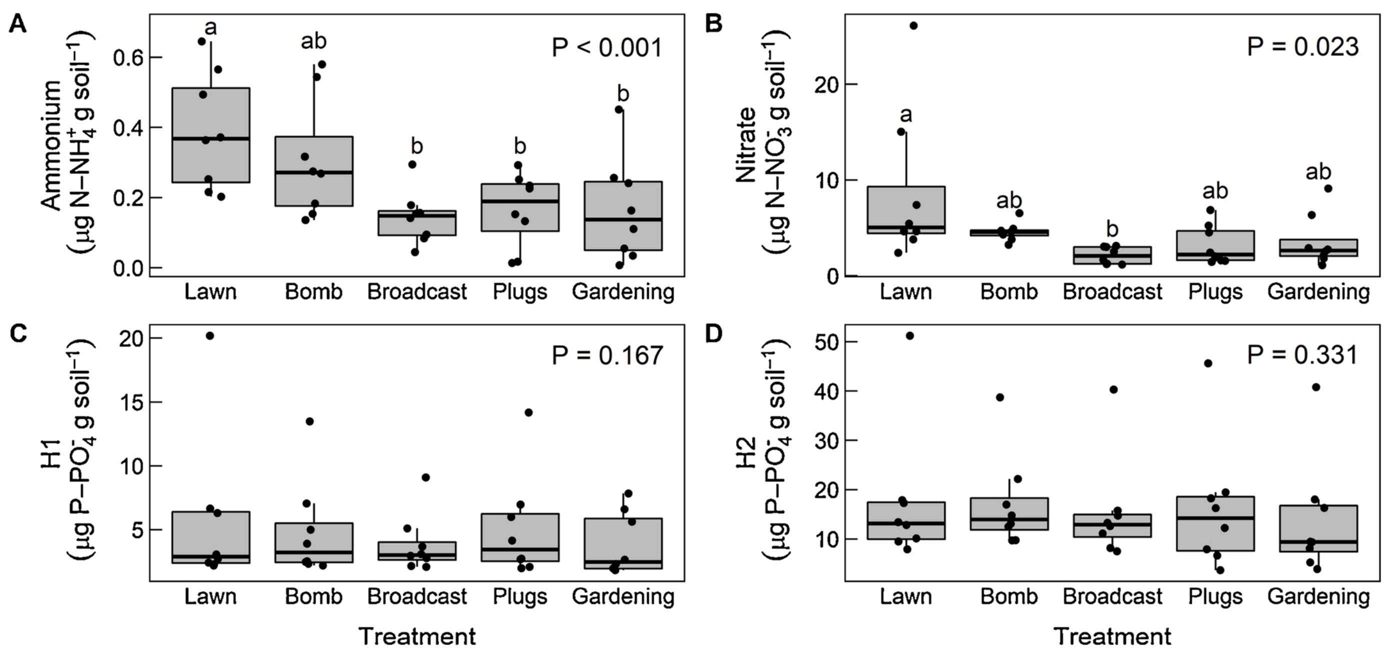

Fig. 3. Boxplots of soil inorganic (a,b) nitrogen and (c,d) phosphorus pools in vacant lot restoration treatments and adjacent unrestored lawns at the University of Illinois at Chicago. In (a,b), two forms of inorganic N (NH_4^+ and NO_3^-) are presented. In (c,d), phosphate sequentially extracted using water (H1) and sodium bicarbonate (H2) solutions are presented. Different letters represent significantly different groups based on Tukey's HSD pairwise comparisons at $\alpha = 0.05$.

fertilizer and water inputs (Campbell et al., 2014; Golubiewski, 2006; Qian et al., 2003; Trammell et al., 2017) and mowing-induced allocation to below-ground tissues (Hamilton and Frank, 2001; Poeplau et al., 2016). Grass clippings left on-site can further increase soil C (Law and Patton, 2017; Poeplau et al., 2016; Qian et al., 2003). However, time since disturbance also has a large impact on C storage in urban soils, particularly under lawn (Campbell et al., 2014; Contosta et al., 2020; Scharenbroch et al., 2005). Our broadcast seeding, plug planting, and intensive gardening treatments were tilled two years before soil sampling, while the lawn and seed bomb treatments were not. Furthermore, the turfgrass in our study was not fertilized or watered. As such, it is possible that organic matter and C concentrations under our lawn controls were higher than in our more-intensive treatments simply because the treatment soils were disturbed more recently.

In contrast to lawn and other treatments (seed bombing, broadcast seeding, and plug planting), intensive gardening led to decreased soil aggregate stability. Soil disturbance breaks up aggregates and mixes microbes, nutrients, and newly-available C together, resulting in respiration-induced decreases in soil C (Chen et al., 2014; Grandy and Robertson, 2007). Additionally, the process of gardening removes roots that stabilize C in aggregates (Jastrow et al., 1998). Thus, aggregates in the intensive gardening treatment were likely less stable due to the frequent interventions occurring in this treatment: watering and weeding. However, stable aggregates develop rapidly upon cessation of disturbance (Jastrow, 1987), which is likely why the treatments that were not subject to ongoing interventions had soil aggregates with similar stability to those found in the undisturbed lawn treatment.

While plug planting and broadcast seeding had relatively low concentrations of organic matter and C, these metrics of C storage may increase over time as plant roots grow, litter inputs increase, and soils develop. This phenomenon has been observed in managed residential lawns (Golubiewski, 2006; Raciti et al., 2011; Smith et al., 2018), restored prairies (Baer et al., 2002; Jastrow et al., 1998), and vacant residential lands (Gough and Elliott, 2012). Microbial biomass C is negatively affected by soil disturbance (Bach et al., 2012), which may explain the decline in microbial biomass C in the intensive garden treatment relative to the undisturbed lawn. As such, microbial biomass C, an important precursor to long-term soil C storage (Cotrufo et al.,

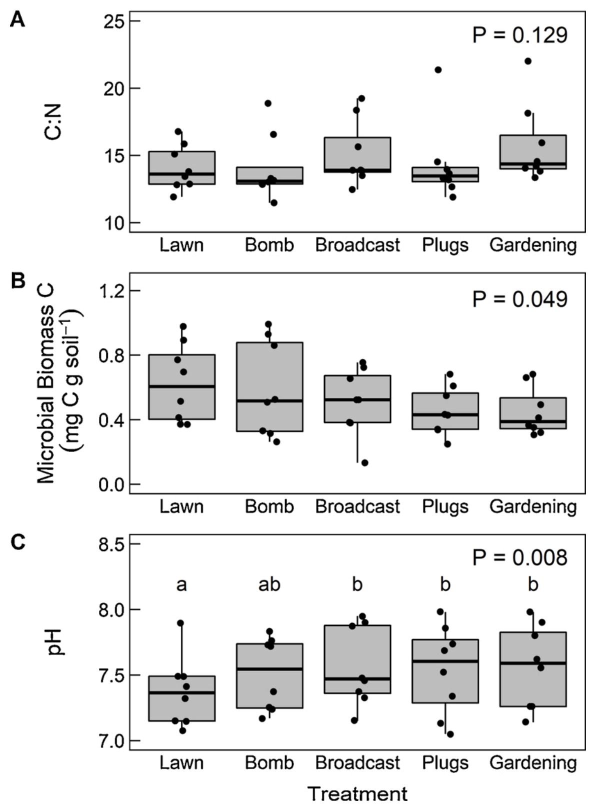

Fig. 4. Boxplots of soil (a) carbon to nitrogen ratios, (b) microbial biomass carbon, and (c) pH in vacant lot restoration treatments and adjacent unrestored lawns at the University of Illinois at Chicago. Different letters represent significantly different groups based on Tukey's HSD pairwise comparisons at $\alpha = 0.05$.

2013), may also recover in restored vacant lots over time.

4.2. Nutrient retention

The broadcast seeding treatment lowered concentrations of total soil inorganic N and nitrate, while ammonium concentrations declined in the broadcast seeding, plug planting, and intensive gardening treatments, indicating that the plant establishment treatments led to enhanced N retention. Low concentrations of soil inorganic N can sometimes be negative for plant growth. However, vacant lots have soil N levels similar to those found under residential lawns (Herrmann et al., 2017), which commonly have as great or greater N levels than those in agricultural or wildland soils (Golubiewski, 2006; Pouyat et al., 2002, 2009; Raciti et al., 2011). This excess N can lead to nitrate leaching and denitrification (Qian et al., 2003; Smith et al., 2018; Trammell et al., 2016). In our study, lawn soils had inorganic N concentrations that were more than double those found in nearby native ecosystems (Taylor and Midgley, 2018), while the treatments created much lower soil inorganic N concentrations. Overall, our plant establishment treatments – broadcast seeding, in particular – appear to create soil N dynamics that approximate those found in native ecosystems.

Decreased soil inorganic N concentrations in our restored plots are likely driven by a combination of plant N dynamics and microbial N immobilization. Plants take up inorganic N from the soil as they grow. While turfgrass grows quickly, mowing returns N to the soil in the form of easily decomposable lawn clippings. In contrast, tallgrass prairie plants take up N and retain it during the growing season. Upon senescence, their recalcitrant litters lead to a slow release of N in the spring that is readily taken up by new growth. Sawdust addition to our broadcast seeding, plug planting, and intensive gardening treatments may have also driven down inorganic N further via microbial immobilization. Though soil pH is often negatively correlated with nitrate production, soil pH was relatively high across all treatments in our study. pH-driven changes in nitrate production rates are detected when pH ranges are wide (from 3 to 10; Ste-Marie and Paré, 1999); the relatively small differences we found among treatments (range from 7 to 8) suggest that pH is not a major driver of varied soil inorganic N concentrations across treatments. As such, inherent differences in plant management and tissue quality between lawn and plant establishment treatments and the addition of sawdust likely drove differences in soil inorganic N concentrations.

While we expect differences in soil C pools between our treatments to diminish over time, we expect differences in inorganic N and P availability to be further enhanced over time. Though a study in Ohio found that soil ammonium concentrations in predominantly grassy vacant lots and community gardens converged over time, this was likely driven by increased fertilizer inputs in the community gardens (Grewal et al., 2011). Furthermore, in a study examining time since development on ecosystem service indicators in Madison, Wisconsin, available P decreased over time in grassland sites, but increased over time in residential sites (Ziter and Turner, 2018). As we did not incorporate fertilization into our treatments, we expect ecosystem nutrient retention to be stable or potentially even increase in our restored plots over time.

4.3. Water infiltration

Our metrics of water infiltration – percent bare soil and percent aggregate stability – suggest that intensive gardening subplots have lower water infiltration capacities than the other treatments or the unrestored lawns. Though indirect metrics of water infiltration, percent bare soil and aggregate stability provide insight into the relative water infiltration capabilities of our treatments. Bare soil is more subject to erosion than vegetated soil. In bare soil, wind and rainfall dislodge soil particles that subsequently fill in and block soil pores (Abid and Lal, 2009). This can lead to the development of surface crusts that restrict water infiltration (Craul, 1985). Removing undesirable plants produces areas of bare soil, thereby decreasing water infiltration into soil. Additionally, as discussed above, frequent weeding and watering lead to decreased aggregate stability. Thus, the intensive gardening treatment likely has reduced capacity to provide water infiltration ecosystem services compared to other treatments and lawn. However, in a comparison of rain gardens planted with turfgrass or native prairie species, median water infiltration rates were much higher in prairie rain gardens, likely due to their deep rooting systems (Nocco et al., 2016). Vacant lots generally have large areas of permeable surfaces areas compared to occupied residential lots (Herrmann et al., 2017); our findings show that vegetation management in vacant lots also has consequences for water infiltration patterns.

4.4. Plant establishment techniques present trade-offs in ecosystem services

Our results corroborate the aboveground findings of Anderson and Minor (2020), which demonstrate that selecting a technique to maximize ecosystem services and minimize inputs is not a clear-cut decision. In a previous study at this site, Anderson and Minor (2020) found that seed bombing was a relatively ineffective treatment for establishing native plants; four of the eight seed-bombed plots had no target species growth, and the other replicates only had one or two target species (of eight planted). In contrast, broadcast seeding and plug plantings maximized aboveground ecosystem services while minimizing inputs (i.e., time for maintenance; watering). However, selection of one of these techniques over the other depends on one's definition of success. Plug plantings led to a greater number of stems and flowers and increased species richness of target species, while broadcast treatments led to the successful establishment of all eight target species. Furthermore, species success varied between broadcast and plug treatments; *Asclepias tuberosa* and *Echinacea purpurea* had greatest growth and flowering when established from seed while *Solidago rigida* and *Helianthus occidentalis* fared best when planted as plugs.

Results from the present study suggest that broadcast seeding and plug planting strategies also maximize three soil ecosystem services provided in vacant lots: soil C storage, nutrient retention, and water infiltration. Nutrient retention and water infiltration were high in the broadcast seeding and plug planting treatments, and while some soil C storage metrics (namely, soil C and organic matter concentrations) were low in these treatments, we expect them to increase over time. Seed bombing tended to have intermediate effects on soil properties (in between lawn and more intensive treatments), but also had limited desirable aboveground effects. While our ecosystem service proxies have their limitations (described below), in combination with our aboveground data, they suggest that broadcast seeding, planting plugs, or some combination of the two will likely maximize belowground ecosystem services and desirable aboveground outcomes. Combined with the aboveground outcomes observed by Anderson and Minor (2020), there are clear tradeoffs in investment and ecological function which require holistic integration to achieve the desired goals of a restoration/greening project.

4.5. Caveats and future directions

There are two caveats with our conclusions: we evaluated the effects of our plant establishment treatments at only one site, and we sampled two years after plot establishment. Initial properties of vacant lots can vary widely (Newman et al., 2016). They differ in their historical land use, time since structure demolition, current management, and local and neighborhood context (Herrmann et al., 2017; Scharenbroch et al., 2005; Ziter and Turner, 2018). Soil properties also vary within vacant lots due to fine scale land use and management, such as historic locations of the residential structure and yard (Herrmann et al., 2017). Future studies should assess the effects of different restoration treatments in vacant lots that vary in initial properties. Additionally, future studies should examine changes in soil ecosystem services over time. As discussed above, some of our findings may be ephemeral, while others may be stable, and some differences among treatments may be enhanced, while others might be dampened over time. Short-term studies at fine spatial scales are common in ecology, but testing the effects of our treatments across a wide range of vacant lots that vary in their initial properties and assessing the effects of our treatments over time are needed to broadly apply our findings.

There are also challenges associated with applying our proxy metrics to ecosystem services provision. A complete assessment of nutrient retention, for instance, would include plant nutrient uptake and nutrient leaching fluxes. Similarly, direct measurements of water infiltration rates may be better and simpler than aggregate stability and bare soil assessments. Finally, some of our metrics are closely tied to our experimental choices – bare soil created by weeding and nutrient concentrations reduced by sawdust addition. Minimally, our results demonstrate that, in the short term, tilling and sawdust addition decrease soil C, organic matter, and inorganic N pools and increase pH, and weeding decreases aggregate stability and increases bare soil area. Clearly, converting lawn into prairie gardens alters soil properties, and the conversion and management methods employed have varied effects on soils. Collecting process measurements to complement our pool measurements and monitoring these metrics over time will be key to conclusively identifying vacant lot management strategies that maximize ecosystem services.

4.6. Conclusions

While early urban ecology studies focused on the effects of urbanization on "natural" ecosystems along urban-rural gradients (Lovett et al., 2000; Pouyat et al., 2009; Zhu and Carreiro, 2004) and subsequent studies examined land use effects on ecosystem processes within cities (Groffman et al., 2009; Ziter and Turner, 2018), recent research highlights the large effects of fine-scale management on urban ecosystem services. In fact, fine-scale drivers may be more important than land use classification or location, particularly in developed areas (Tresch et al., 2019; Ziter and Turner, 2018). Our study shows that native plant community establishment in vacant lots, and even the choice of establishment strategy, has consequences for soil ecosystem services. Notably, we found that most methods of installing native plants at least temporarily reduce C sequestration while enhancing N retention. Furthermore, intensive gardening may inhibit water uptake due to increased bare soil and decreased stable aggregate formation. The choice to green a space and the method by which that is undertaken alter the ecosystem services provided by a lot. Individual choice has also been shown to alter the ecosystem services provided by lots in community gardens (Bretzel et al., 2018; Tresch et al., 2019, 2018). Thus, in the face of large-scale environmental changes, the fine-scale actions of individuals and small groups have the potential to improve ecosystem services in a major way (Cerra, 2017).

While the potential for large-scale ecosystem service enhancement through small-scale actions is encouraging, in some cases, "restoration" practices may have trade-offs, leading to unintended ecosystem disservices (negative or unintended consequences) (Pataki et al., 2011). As demonstrated by the present study, restoration-induced enhancement of a given ecosystem service does not necessarily reflect restoration effects on other services (Ziter and Turner, 2018). Furthermore, fine-scale management and aesthetic decisions that modify habitat structures are driven by the cultural and socioeconomic contexts of individuals and communities (Byrne, 2007). At our site, assuming soil C sequestration metrics rebound over time, broadcast seeding and plug planting treatments optimized above- and belowground ecosystem services. However, adoption of vacant lot restoration practices that maximize belowground ecosystem service provision requires alignment of these techniques with societal values.

Author statement

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Declaration of Competing Interest

The authors have no competing interests to declare.

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