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RESEARCH ARTICLE

Assessing four methods for establishing native plants on urban vacant land

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Abstract Urban greening increases vegetation and can restore ecological functions to urban systems. It has ties to restoration ecology, which aims to return degraded land to diverse, functional ecosystems. Both practices can be applied to maximizing ecosystem services and habitat in vacant lots, which are abundant in post-industrial cities, including Chicago, Illinois (USA), where our study took place. We tested four methods for increasing native plant diversity in vacant lots, ranging from low input to resource-intensive: seed bombing, broadcast seeding, planting plugs, and gardening. After three growing seasons, we assessed the growth of eight target native species and all non-target species. We expected that intensive treatments would have more target species stems and flowers and fewer non-target species, but we found that less-intensive options often produce equal or better results. From this, we recommend broadcast seeding as a viable, low-cost method for improving habitat and biodiversity in vacant lots.

Keywords Broadcast seeding · Chicago · Greening · Native plantings · Seed bombs · Vacant lots

INTRODUCTION

Urban greening increases abundance and cover of vegetation and has benefits for humans and wildlife (Bowler et al. 2010). Greening practices, such as planting street trees or building green roofs, vary in extent and cost (Li et al. 2005; Siwiec et al. 2018), but generally improve provisioning, supporting, and cultural ecosystem services (Li et al. 2005; Ko 2018; Roman et al. 2018). From this perspective, urban greening is a method to restore elements of ecological function in the novel urban ecosystem (Hobbs et al. 2009). As such, urban greening has clear ties to restoration ecology, which aims to return degraded land to diverse and functional ecosystems by reconstituting or rehabilitating land (Packard and Mutel 2005). Native plants are a common and critical element of restoration as they provide essential habitat for a wide variety of wildlife specialists and are well adapted to local climatic conditions. For these reasons, native plants are gaining traction as a focal element of urban greening (Alvey 2006). However, despite these similarities, there is not adequate cross-talk about urban greening and restoration efforts between disciplines (Vogt 2018). In this study, we blur the lines between restoration ecology and urban greening by evaluating urban greening methods with the aim of increasing native plant abundance and diversity in vacant lots.

Vacant lots are common targets for urban greening (Heckert and Mennis 2012). Most post-industrial American cities have substantial vacant land, and nationally about 16% of urban land area is vacant (Newman et al. 2016). In Chicago, IL (USA), where our study took place, the city owns approximately 700 ha of vacant land (Minor et al. 2018). In their unrestored form, vacant lots are ecologically valuable. They are known to provide many ecosystem services and harbor diverse communities (Bonthoux et al. 2014; Mathy et al. 2015; Anderson and Minor 2020), but they also offer abundant area for potential improvements to urban habitat (Schröder and Kiehl 2020). Soils in vacant lots can be dry and nutrient poor, and can potentially serve as refugia for species dependent on nutrient poor habitats (Schadek et al. 2008; Albrecht et al. 2011; Bonthoux et al. 2014; Schröder and Kiehl 2020). Despite these unique ecologies, the socio-demographic distributions of vacant

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lots must be forefront in ecological thinking as this substantial land reserve is concentrated in low-income areas of the city. These neighborhoods, typically home to minority residents, experience ecological injustices such as heavy metal contamination and allergenic pollen exposure at a higher rate than affluent areas (Katz and Carey 2014; Sharma et al. 2015), and also receive less green infrastructure development (Byrne 2018; Ambrey et al. 2017). However, drastically improving the aesthetic or recreational quality of green spaces in low-income areas can instigate displacement of local residents (Rigolon and Németh 2018). Low-input restoration or urban greening may help avoid some of the unintended gentrifying consequences of larger-scale projects (Wolch et al. 2014), while still providing ecological, social, and economic benefits (Anderson and Minor 2017).

Socially, some urban greening methods such as guerilla gardening and community tree planting are known to empower local residents and bolster community cohesion (Ryan 2015). Ecologically, the use of native plants in these greening efforts can help establish critical urban habitat for wildlife that are dependent on these species for food and habitat (e.g., migratory birds, Burghardt et al. 2009; native bees, Threlfall et al. 2015; monarch butterflies, Johnston et al. 2019). By minimizing the risks of eco-gentrification, empowering local residents, promoting ecosystem services such as storm water uptake and soil remediation, and bolstering habitat for wildlife, establishing native plantings in vacant lots has the potential to support a "win–win" outcome for people and urban wildlife (Rosenzweig 2003).

While Lundholm and Richardson (2010) argue that utilizing urban habitat analogues to conserve native species is critical in the Anthropocene, it is unclear how to best establish and support sustainable populations of native plants in cities (Klaus 2013; Mårtensson 2017). To begin to understand how we can improve ecosystem services via urban greening in vacant lots, we need to assess different techniques for establishing native plants. From the restoration literature, we know that certain methods benefit certain types of plants in different conditions (Fahselt 2007; Godefroid et al. 2011). For instance, different families seem to perform better in restorations when planted from seeds versus from plugs (Wallin et al. 2009). Broadcast (or direct) seeding is easier and cheaper, but supplementing with plugs may benefit late-successional species (Hedberg and Kotowski 2010; Kövendi-Jakó et al. 2018). Methods for native plant establishment can and should be informed by knowledge of restoration ecology, but must be validated in urban settings.

Plant habitats in urban environments are inherently different from those in non-urban areas. Many spontaneous urban plants are introduced ruderal species, and soils have high inorganic nitrogen from anthropogenic pollutants (Mattina et al. 2003; Scharenbroch et al. 2005). Previous work in Chicago's vacant lots has demonstrated that a vast majority of plant species in these sites are introduced forbs, and several species are indeed known to be noxious weeds (Anderson and Minor 2020). Furthermore, landscape legacies such as building footprints and post-industrial waste create harsh conditions for plant growth (Skubała 2011; Johnson et al. 2018), and change the hydrology (Niemczynowicz 1999), chemistry (Zhu et al. 2017; Majidzadeh et al. 2018), and physical structure (Byrne 2007) of soils. In addition to these biophysical differences, socially dominated spaces have different goals. Spaces that look tidy and cared for are better perceived by local residents (Nassauer 1995) and result in greater community buy-in and long-term success (Chaffin et al. 2016). And finally, economics matters. Establishment and upkeep of green spaces can be expensive and labor-intensive (Siwiec et al. 2018). Cities might not want to invest resources in vacant lots that could be developed in the near future. Considering these social and economic elements is important when conceptualizing new methods for urban greening.

In this study, we experimentally evaluated four techniques for greening vacant lots. We used four planting methods to establish eight native plant species, which were selected to represent a range of bloom times, aesthetics, and flower colors. The planting methods varied in terms of their expense and time commitment, ranging from very low-cost and quick to more expensive and time-intensive. To better understand how low-input planting methods influences plant communities, we examined the outcome of these treatments after three growing seasons in terms of two factors: (1) the abundance, diversity, and height of the native species we planted ("target species"), and (2) the diversity and height of non-target species in the plots.

MATERIALS AND METHODS

Study site

Our experiment took place on the campus of the University of Illinois at Chicago (UIC), which is located in the center of the city in Chicago, Illinois (USA). Chicago sits on the southwestern shore of Lake Michigan and experiences four dramatic seasons, with cold, snowy winters (average − 8 °C to 0 °C) and hot, humid summers (average 16 °C–28 °C, US National Weather Service). During the summer of 2015 when our plots were established, Chicago experienced a typical summer. The mean daily temperature was 21.4 °C and the area received 30.8 cm of rainfall (0.2 cm above normal). There was one record-breaking rain in mid-June (6.5 cm in 24 h, US National Weather Service).

Historically, northern Illinois was dominated by tall-grass prairies and interspersed savannah, but intensive urbanization over the past 200 years has almost completely displaced this native ecosystem. The university was established in its current location in 1965, and the vacant land for our experiment has been undeveloped for at least the past two decades.

Experimental design

In late spring of 2015, we set up eight replicate test gardens on two different parts of UIC campus (Fig. 1). Four replicates were on the grounds of the UIC Plant Research Laboratory (PRL), and the other four were approximately 125 meters away, across a 4-lane street, in a vacant space adjacent to a parking garage. Experimental gardens were established in areas of seeded turf at both sites; however, it is important to note that both areas had previous or contemporary native plantings nearby. The eight species selected for our experiment were not found with 50 m of our test plots before beginning the experiment. The areas surrounding the experimental gardens received full or almost-full sun, occasional mowing and management, and little foot traffic throughout the 3-year experiment.

Each 5.5 m × 5.5 m test garden contained 4 subplots measuring 2 m × 2 m. Each treatment was randomly assigned to one subplot per garden and each subplot was separated from others by a 0.5 m buffer that was covered with flattened cardboard, to reduce clonal spread of plants and allow easy access. We left the existing turf grass in place in one subplot (for the 'seed bombing' treatment, described below), but in the other three subplots we removed all turf grass and tilled the upper 15–20 cm of the soil with a conventional rototiller. We added 10 kg of hardwood sawdust obtained from a local pallet manufacturer to each of the three tilled subplots to increase the soil C:N ratio, as total N pools in Chicago are high and soil inorganic N levels are enriched in the urban core compared to surrounding natural areas (M. Midgley unpublished data). Adding hardwood sawdust is a cheap, easy, and scalable method to counteract this enrichment has been shown to improve native plant establishment and reduce weed growth (Corbin and D'Antonio 2004; Prober et al. 2005). Within a garden, each subplot received one of four experimental treatments designed to establish native plant communities: seed bombing, broadcast seeding, planting plugs, and intensive gardening. These treatments, which are described in more detail below, 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.

Each plot was sowed (as seeds) or planted (as plugs) with an equal proportion of eight native species: black-eyed Susan (*Rudbeckia hirta*), white prairie clover (*Dalea candida*), pale purple coneflower (*Echinacea pallida*),

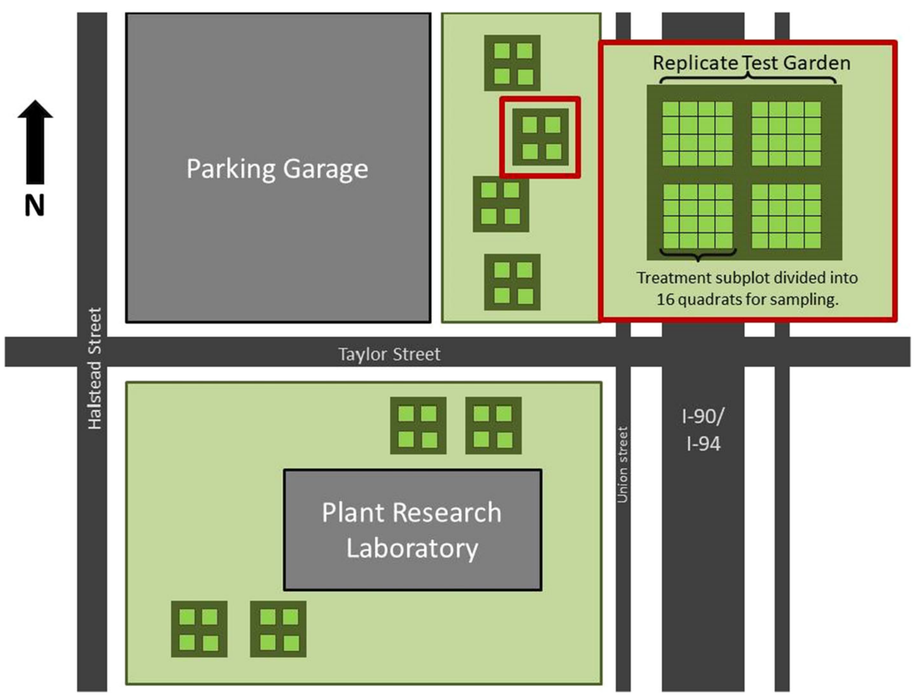

Fig. 1 Schematic of our experimental design, including four gardens each at two sites. Each 5.5 m × 5.5 m garden contained four 2 m × 2 m subplots with one of the following treatments: seed bombing, broadcast seeding, plugs, or gardened. Dark gray lines represent roads and highways. Diagram is not to scale

purple coneflower (*Echinacea purpurea*), stiff-leaved goldenrod (*Solidago rigida*), western sunflower (*Helianthus occidentalis*), switchgrass (*Panicum virgatum*), and butterfly weed (*Asclepias tuberosa*). Seeds were purchased from Prairie Moon Nursery (www.prairiemoon.com). Plugs came from different sources. *D. candida*, *E. purpurea*, *H. occidentalis*, and *A. tuberosa* were purchased from Midwest Groundcovers LLC (www.midwestgroundcovers.com); *P. virgatum*, *S. rigida*, and *E. pallida* were purchased from IonXchange Nursery (www.ionxchange.com). *R. hirta* plants were grown in our greenhouse because heirloom plugs were not commercially available from any local nurseries. The plugs received from Midwest Groundcovers were in their second year of growth while the plugs from IonXchange and our *R. hirta* plugs were first year seedlings.

All seeds for *A. tuberosa*, *E, pallida*, *S. rigida*, and *H. occidentalis* were stratified in a cold, moist environment for 30 days prior to planting or including in seed bombs, per the recommendation of the seed distributor. Henceforth "stratified seeds" refers to the mixture of all eight species, which were cold-treated or not according to Prairie Moon Nursery recommendations.

Treatments

Seed bombing

Seed bombs—small balls made of clay and soil imbued with seeds—are potentially a user-friendly way to establish plants in hard-to-reach locations by tossing them over a fence or out a car window. Making seed bombs is a popular activity at garden centers and family events, and it is potentially a viable method for urban greening by engaging and educating local residents in an effort to distribute native seeds across a large area. Our method closely followed methods found online for such purposes. For each seed bomb, we combined 50 g of Crayola™ air-dry clay with 10 g of organic potting soil (MiracleGro Nature's Care with Water Conserve™). We added 8–10 stratified seeds of each species and rolled the mixture into a sphere approximately 5 cm in diameter. We dried the seed bombs in the sun for 3–4 days and then dropped 20 seed bombs into each subplot, with the turf grass left intact. The remaining turf was not mowed throughout the course of the experiment.

Broadcast seeding

For this treatment, we broadcast 200 stratified seeds of each species into a tilled and weeded subplot. Each species was spread individually by hand as evenly as possible to avoid any potential biases in seed weight and size.

Planting plugs and intensive gardening

For these two treatments, six plugs of each species were fully randomized and spaced evenly throughout the subplot. For both treatments, plugs were planted in tilled and weeded subplots. One of these treatments was left to grow without intervention (referred to as "plugs"), while the other treatment (referred to as "gardened") served as our highest-input technique and a positive control. The gardened treatment was 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.

All four treatments were regularly watered for the first 4 weeks of growth in 2015 to aid in plant recruitment and establishment, but after mid-July, we only watered the intensively gardened subplots 1–2 times per week.

Data collection

In mid-September and early October of 2017, we measured the deliberately planted (i.e., "target") species and non-target plant species within each subplot. To do this, we divided each subplot into 16 $0.5\ \mathrm{m} \times 0.5\ \mathrm{m}$ quadrats (Fig. 1). In each quadrat, we recorded all species present and measured the height of the tallest target and non-target plant (maximum height).

Target species

To assess the eight target species, we counted the number of stems of each species in each quadrat and summed them across all 16 quadrats to get a value for the subplot. For *S. rigida* and both *Echinacea* species, which have a basal rosette morphology in early development, we counted each rosette as one stem. We measured the height of the tallest target plant in each quadrat, and then calculated the average maximum height for our target species over all quadrats in each subplot.

We counted the number of flower heads of each target species, including intact flower heads as well as senescing pieces. As some of our focal species had rather small individual flowers, we counted composite flowers, racemes, and umbels as single flower heads. *A. tuberosa* was finished flowering at the time of our surveys so we considered a seed head as one flower. Similarly, we counted a single seed head of *P. virgatum* as one flower.

Non-target species

We identified each non-target species present in each $0.5 \times 0.5\ \mathrm{m}^2$ quadrat. To get a crude estimate of the biomass produced by non-target species, we measured the maximum height of non-target species in each of our 16 quadrats and calculated an average maximum height for each subplot.

Data analysis

Target Species

We analyzed target species as a community and individually. For the community analysis, we used the following metrics to assess the overall success of target species in each subplot: total number of stems, total number of flowers, target species richness, and average maximum height. We used linear mixed-effects (lme, R package “nlme”; Pinhiro et al. 2017, R version 3.5.1) models followed by one-way analysis of variance (ANOVA) to independently assess the differences between treatments in the four target species metrics. Each model included planting treatment as a fixed effect and random effects for the location (PRL vs. parking garage adjacent) and replicate number (1–8). The random effects of location and replicate number were included to account for the nested study design, fine-scale heterogeneity in environmental conditions, and any discrepancies in land-use history among our two garden locations on campus. In essence, the combined random effects account for correlation within groups and allow us to assess the residual variance unexplained by the fixed effect of treatment.

If the fixed effect of treatment was significant, we used a Tukey’s Honest Significant Difference (HSD) test to assess pairwise comparisons between treatments. For each model (stems, flowers, richness, height), we also assessed the significance of the random effects of location (PRL vs. parking lot adjacent) and plot (replicate 1–8). To do this, we used the gls function (package ‘nlme’ R version 3.5.1.) to reconstruct each model with just fixed effects and compared the full mixed-effects model with the fixed-effects model using ANOVA (Mangiafico 2016). We assessed significance for these comparisons at $\alpha = 0.05$.

We also looked at each target species independently to see which treatment performed best. We used the same lme/ANOVA method to analyze the number of stems and the number of flowers independently for each species by running a series of one-way analysis of variance (ANOVA) tests with Tukey’s HSD pairwise comparisons. Since this method is known to inflate the Type I error, we assessed significance at $\alpha = 0.01$.

Non-target species

We used the Shannon diversity index and average maximum height to assess the community of non-target species in our subplots. We used the presence/absence data for each quadrat to calculate a relative abundance (out of 16 quadrats) of each non-target species per subplot. We then used this information to calculate the Shannon diversity index. The Shannon index is useful in this context because, unlike Simpson’s Index, it is more sensitive to changes in species richness than to changes in species evenness (Hurlburt 1971). We made this prioritization because we have a fairly coarse measure of relative abundance (ranging from 0 to 16), but have a precise measurement of species richness.

To understand how different treatments affected the community and structure of non-target plant species, we ran two linear mixed-effects (lme) models using the non-target Shannon diversity index and the average non-target maximum height as our response variables. Just like our target species models, these models included planting treatment as a fixed effect and plot location and replicate number as random effects. We used a Tukey’s HSD test to look at pairwise differences when treatment had a significant effect on the non-target Shannon diversity index.

RESULTS

Target species

Number of stems

We saw a significant difference in the number of target stems between treatments (lme results Table S1: ANOVA df = 1,3, $p < 0.001$, Fig. 2a). Gardened subplots had the most target stems [107 ± 22 (mean $\pm$ SE)], but did not differ significantly from plugs subplots (64 ± 17). Broadcast subplots were similar to plugs subplots (48 ± 11), and also did not differ significantly from seed-bombed subplots, where we saw very little target species growth (1 ± 0.5). The random effects of location and plot number did not significantly change model performance ($p = 0.98$).

Number of Flowers

We saw higher flower abundance in the gardened treatments (lme results, Table S1; ANOVA df = 1,3, $p < 0.001$, Fig. 2b), but gardened subplots (1327 ± 283) did not differ from plugs treatments (926 ± 270). Broadcast subplots had intermediate numbers of flowers (465 ± 148) compared to seed-bombed (9 ± 7) and plugs treatments, but did not differ significantly from either. Residuals for the combined random effects of garden replicate (1–8) and location (PSL or parking lot adjacent) were quite high, and the random effects had a significant effect on model performance ($p = 0.01$), suggesting a strong effect of location at a fine spatial scale on floral abundance.

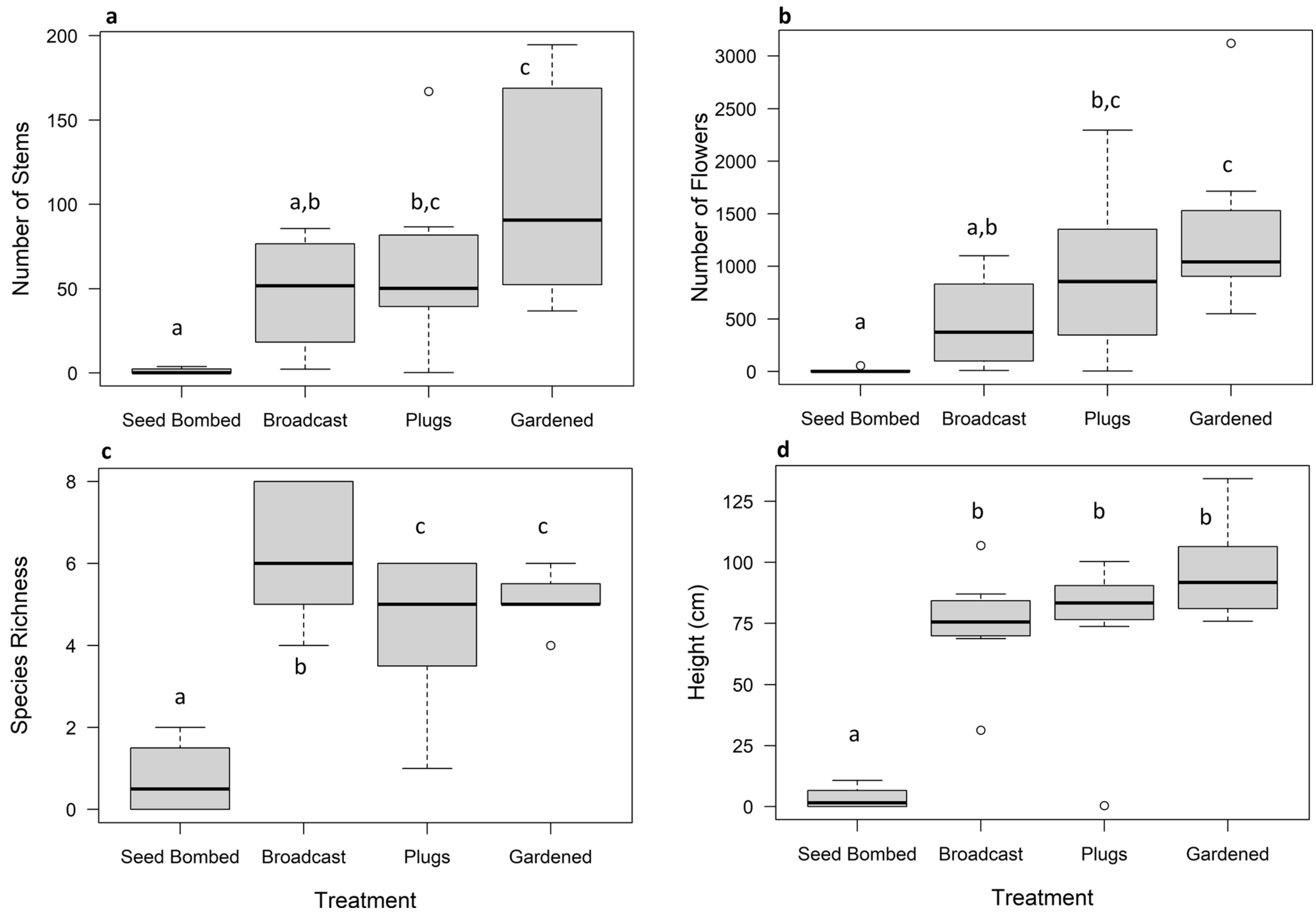

Fig. 2 Boxplots of target species **a** stems, **b** flowers, **c** richness, and **d** average maximum height. Graphs are based on raw data, although ANOVA and Tukey's HSD tests are calculated on fit of linear mixed-effects models (lme). Pairwise comparisons (**a–c**) are notated for significant Tukey's HSD tests. Lines in the box plots show median values, with the first and third quartiles delineated within the box

Species richness

We only ever saw all eight target species in broadcast subplots, which were significantly richer than other treatments in terms of target species (lme results Table S1; ANOVA, df = 1,3, $p < 0.001$, Fig. 2c). Plugs and gardened treatments both had a median of approximately 5 target species, and we rarely saw more than one target species established in seed-bombed plots. The random effects of location and plot number significantly improved model performance ($p < 0.001$).

Height

Target species average maximum height did not differ significantly between broadcast, plugs, or gardened subplots, but target species in these three treatments were all significantly taller than those in seed-bombed plots (lme results Table S1; ANOVA, df = 1,3, $p < 0.001$, Fig. 2d). The random effects of location and plot number did not significantly change model performance ($p = 0.73$).

Individual species

Target species responded differently to the treatments (Table 1). All species had more stems and flowers in either the broadcast or gardened subplots, although not all differences were significant. One species, *A. tuberosa*, only flowered in broadcast plots, where it also had significantly more stems. There was no difference in *E. purpurea* flowers or stems between gardened and broadcast plots; however, broadcasting resulted in a significantly higher number of stems than either seed bombing or planting plugs. While *S. rigida* did better in the gardened treatment than the seed-bombed treatment for both flowers and stems, and better than broadcast in terms of flowers, there was not a significant difference between the gardened and plugs treatments in either metric. *H. occidentalis* also performed

Table 1 Individual responses of target species to four treatments. We used individual lme models with random effects for plot number and location, followed by one-way ANOVAs and Tukey's HSD tests. *F* and *p* values are results from the ANOVA comparison of model fits between treatments. Superscripts indicate significant Tukey's HSD results. Number of stems for basal rosette species (*E. pallida, E. purpurea, S. rigida*) is based on counts of rosettes. Number of flowers indicates the number of flowering heads of compound, umbel, and raceme morphologies, and includes both fresh and senescing flower heads. Bolded numbers indicate the treatment with the highest number of stems or flowers for each species, regardless of statistical significance

Species	Seed bombed Mean ± SE	Broadcast Mean ± SE	Plugs Mean ± SE	Gardened Mean ± SE	*F*	*p*
Stems						
Asclepias tuberosa	0.13 ± 0.13^{a}	$\mathbf{9.63 \pm 4.26^{b}}$	0.00 ± 0.00^{a}	0.13 ± 0.35^{a}	5.00	0.009
Dalea candida	0.00 ± 0.00	**1.63 ± 0.91**	0.00 ± 0.00	0.25 ± 0.71	3.23	0.043
Echinacea pallida	0.00 ± 0.00	**3.13 ± 1.09**	0.88 ± 0.52	1.88 ± 3.80	2.73	0.069
Echinacea purpurea	1.13 ± 1.13^{a}	$\mathbf{35.25 \pm 8.79^{b}}$	5.63 ± 2.15^{a}	$18.38 \pm 16.50^{a,b}$	7.98	0.001
Helianthus occidentalis	0.13 ± 0.13^{a}	$23.00 \pm 8.63^{a,b}$	$43.50 \pm 17.11^{a,b}$	$\mathbf{70.00 \pm 38.84^{b}}$	10.67	< 0.001
Panicum virgatum	4.63 ± 3.03	49.38 ± 21.63	22.63 ± 10.16	**61.75 ± 64.46**	3.54	0.032
Rudbeckia hirta	0.13 ± 0.13	**37.50 ± 12.90**	25.88 ± 20.00	14.75 ± 25.18	1.82	0.173
Solidago rigida	2.50 ± 2.50^{a}	221.63 ± 60.88^{a}	$411.50 \pm 138.23^{a,b}$	$\mathbf{688.25 \pm 476.00^{b}}$	6.62	0.003
Flowers						
Asclepias tuberosa	0.00 ± 0.00	**4.38 ± 2.49**	0.00 ± 0.00	0.00 ± 0.00	3.09	0.048
Dalea candida	0.00 ± 0.00	0.13 ± 0.13	0.00 ± 0.00	**1.75 ± 1.75**	0.95	0.433
Echinacea pallida	0.00 ± 0.00	1.75 ± 0.98	0.00 ± 0.00	**2.13 ± 2.12**	1.16	0.350
Echinacea purpurea	0.88 ± 0.88	24.00 ± 7.02	6.25 ± 2.76	**25.25 ± 12.84**	2.75	0.068
Helianthus occidentalis	0.00 ± 0.00^{a}	72.25 ± 29.65^{a}	111.12 ± 42.95^{a}	$\mathbf{132.75 \pm 32.05^{b}}$	5.51	0.006
Panicum virgatum	8.25 ± 6.93	12.38 ± 5.27	10.12 ± 4.70	**40.88 ± 15.60**	3.35	0.038
Rudbeckia hirta	0.13 ± 0.13	**77.63 ± 34.91**	27.62 ± 19.65	36.50 ± 21.65	1.99	0.138
Solidago rigida	0.00 ± 0.00^{a}	75.66 ± 38.04^{a}	$195.64 \pm 54.33^{a,b}$	$\mathbf{408.69 \pm 116.84^{b}}$	10.19	< 0.001

better in gardened treatments than seed-bombed treatments in terms of both number of stems and number of flowers, and the number of flowers was also significantly higher in gardened treatments than in plugs or broadcast treatments.

Non-target species

We documented 65 non-target species (including turf grass, classified as one species) in our experimental gardens (Table S2). Of these, 26 were native species and 39 were introduced species. On average, each subplot had 13 non-target species, but the variation was quite high, with one subplot containing only four species. The three non-target species with the highest relative abundances across all sampled quadrats ($n = 512$, 16 quadrants per subplot) were Canada thistle (*Cirsium arvense*), bull thistle (*Cirsium vulgare*), and prickly lettuce (*Lactuca serriola*). Bull thistle and prickly lettuce were found almost exclusively at the parking garage adjacent experimental site. Canada thistle, along with two other species (pepperweed: *Lepidium virginicum*, and white mulberry: *Morus alba*), were present in 19/32 subplots, the most of any species. Five non-target species were woody: white mulberry, tree of heaven (*Ailanthus altissima*), silver maple (*Acer saccharinum*), eastern cottonwood (*Populus deltoides*), and Siberian elm (*Ulmus pumila*). All five of these species had large, seed-producing adult specimens within 100 m of our experimental gardens, and are also prominent across the urban landscape in Chicago (Nowak 2010).

Treatments did not differ significantly in average maximum height of non-target species (lme results Table S3; ANOVA, df = 1,3, $p = 0.324$, Fig. 3a). Broadcast plots had significantly higher non-target Shannon diversity than seed-bombed plots (lme results Table S3; ANOVA, df = 1,3, $p = 0.017$, Fig. 3b), but did not differ from plugs or gardened plots.

DISCUSSION

Greening vacant lots using native plants has the potential to provide ecosystem services in low-income urban neighborhoods by increasing native biodiversity and habitat provisioning cities (Schröder and Kiehl 2020). In this study, we tested four methods for increasing native plant diversity in vacant lots that ranged from very low input to fairly resource-intensive: seed bombing, broadcast seeding, planting plugs, and intensive gardening. After three growing seasons, we found that the less-intensive method of broadcast seeding often produced equal or better target

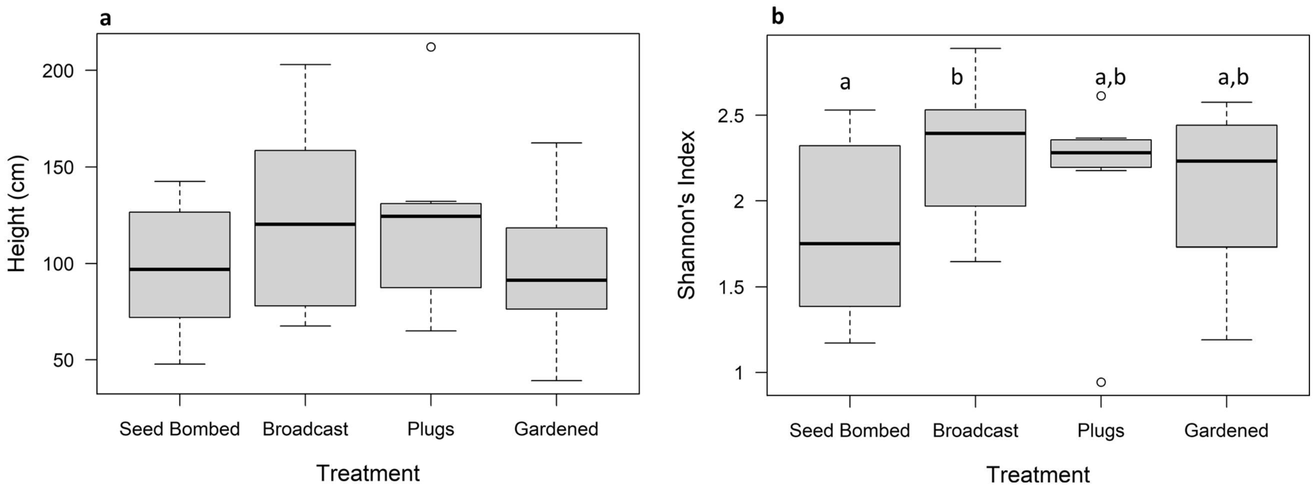

Fig. 3 Boxplots of non-target species **a** average maximum height and **b** Shannon diversity index. Graphs are based on raw data, although ANOVA and Tukey's HSD tests are calculated on fit of linear mixed-effects models (lme). Pairwise comparisons (**a**, **b**) are notated for significant Tukey's HSD tests. Lines in the box plots show median values, with the first and third quartiles delineated within the box

plant establishment and overall diversity when compared to intensive gardening, which we initially predicted would have the highest abundance and diversity of native target species.

There were pronounced differences in plant growth between our treatments, which have consequences for future habitat and ecosystem services provisioning. Two treatments—broadcast seeding and gardening—were overall the most successful, but illustrated a potential trade-off in plant diversity and floral abundance, foreshadowing different long-term community and habitat consequences (Bretzel et al. 2016). Broadcast plots may better support increased plant diversity. Broadcast seeding was the only treatment where all eight target species were present (in 2/8 replicates) after 3 years, compared to a maximum of six target species when planted from plugs or gardened. Broadcast plots also showed a trend towards higher non-target species richness. However, increased diversity in broadcast plots did not manifest itself in more target stems. Indeed, gardened treatments had approximately 50% more target stems and twice as many target floral resources compared to broadcast plots. Non-target species height and diversity were not impacted by weeding, and the Shannon diversity index was equivalent in broadcast, plugs, and gardened treatments, suggesting that local seed banks and establishment rates exert a strong and relatively consistent force on spontaneous plant communities in vacant lots. However, it is worth noting that the Shannon index does not account for community composition so it is possible that there were undetected differences in species composition between treatments.

Seed bombs over turf were a completely ineffective method for establishing native plants. Half of our seed-bombed plots had no target species growth at all, and the other replicates had only a few stems of one or two species. Typically, this was *Rudbeckia hirta*, a wind-dispersed annual that reseeds aggressively (Stevens 1932). Intact turf is a difficult community for new plant species to colonize due to high competition (Fenner 1978), and this feature of our seed-bombed plots likely prevented seeds of the target species from germinating or establishing and seemed to also reduce diversity of non-target species, although the latter effect was not statistically significant. It is likely that seed bombs may have yielded higher target species germination if they were applied to bare soil. However, our ambition with this treatment was to assess the easiest planting method possible, which necessitated leaving the turf intact. While making or distributing seed bombs may be valuable way to engage and educate local residents in urban greening methods, we do not recommend them as a scalable method for establishing native plants in turfed sites. Additional methods of "guerilla gardening" are worth exploring as venues for encouraging environmental awareness, but ecologically are unlikely to make a significant impact on the habitat value of vacant lots at a large scale (Mikadze 2014).

Several greening methods may have utility across urban vacant lots. As is the case with all science-based urban greening projects, planting methods must be selected with specific species-, community-, and ecosystem-level goals in mind. Our results demonstrate that individual species establish differently in different treatments and it is likely that long-term community and ecosystem services patterns will be shaped by planting strategy. Notably, *A. tuberosa*—a milkweed species—had significantly more stems in broadcast plots while *S. rigida* and *H. occidentalis* showed a trend toward more stems and flowers when planted from plugs. While the two latter species benefited from being planted as plugs, on the whole, broadcast seeding appears to be a cheap and effective method for establishing a high number of stems of a diverse native plant community at large scales. Seeds that germinate from broadcast seeding may represent the best genetics for a given site, allowing natural selection to act on the plant community in highly variable urban areas.

In addition to performing better in many cases, broadcast subplots also looked different from plugs or gardened treatments. Broadcast subplots had a more random, interspersed distribution of plants, while subplots planted from plugs retained a feeling of the planting grid with sizeable clumps where individual plugs were installed. Understanding aesthetic appeal of urban greening is a critical field of ongoing research. It is well established that landscapes that shows clear "cues to care" with clear edges and vivid flowers increase satisfaction with a site (Nassauer 1995). However, there is recent evidence from Europe that residents do perceive and value biodiversity-focused urban grassland management (Southon et al. 2017; Fischer et al. 2020). Biodiverse urban landscapes often look less tidy than typical tidy American lawns (Byrne 2008), but the scientific and outreach discourse in the USA is also moving towards prioritizing urban biodiversity (Lerman et al. 2018). In this vein, it is important to understand how our treatments for establishing native plants differ in their overall appearance, as a dispersed vs. clumpy aesthetic of vividly flowering plants may be more acceptable in some planting locations than others.

Compared to other methods, gardening resulted in slightly higher stem and flower abundance when considering all target species combined. However, particularly in light of the responses of individual target species described above, this expensive and time-consuming method does not seem to be worth the investment on the landscape scale. Instead, gardening might be suitable for species of conservation priority where a quick increase in the number of flowers (and potentially seeds) is the top objective, although this practice would certainly require species-specific validation. Specific sites may also benefit from gardening, perhaps in or adjacent to community gardens, protected habitats, or designated conservation areas in city parks where an overall increase in flower abundance may have additional benefits for pollination or aesthetics (Hoyle et al. 2017). In these specific contexts, the continued labor required for the intensive gardening method could also support a model of community-led restoration or urban stewardship, similar to that of community gardening (Svendsen and Campbell 2008). In addition to increasing native plant habitats for urban wildlife and bolstering ecosystem services, engaging community residents in urban greening practices has been shown to help build community cohesion among neighbors, which in turn helps to reduce crime, poverty, and environmental injustice (Ryan 2015).

Increasing native plant diversity is gaining traction in cities (Kendle and Rose 2000), and our study suggests that broadcast seeding in vacant lots may be a viable way to accomplish this. Prairie plants have long, deep roots that stabilize soils and reduce flooding (Asbjornsen et al. 2007). These ecosystem services could improve environmental conditions in a dense city like Chicago, where heavy rains inundate the combined sewer system and pose a public health hazard. Additionally, native plants support specific pollinator interactions (Salisbury et al. 2015) and are better habitat for native birds (Chace and Walsh 2006). It is worth noting that there are critics of focusing on native plantings in cities in the face of changing climates (Niinemets and Peñuelas 2008). These critics say that native plants are not well suited to salty, polluted urban environments and that introduced or cultivated species often perform better in terms of net primary production (NPP) and carbon sequestration. However, we argue that there is little to lose and much to gain by broadcast seeding native plants in vacant lots across the city. Broadcast seeding resulted in good recruitment and retention of robust and attractive native species over three growing seasons. Rather than throwing the baby out with the climate-change bathwater, increasing native species richness in vacant lots could be embraced as a front-line strategy to help improve urban habitats and ecosystem services quickly, cheaply, and across a large scale.

CONCLUSIONS

Our study demonstrates that planting method makes a difference in community composition, floral success, and ultimately ecosystem services provided by a site. In comparing the target and non-target plant growth after three growing seasons in plots established via seed bombing, broadcast seeding, planting plugs, and intensive gardening, we found pronounced differences in target species abundance and number of stems and flowers. While seed bombing was completely ineffective for establishing native plant species, broadcast seeding and intensive gardening may have utility in different urban contexts, given site-specific priorities. Broadcast seeding resulted in a more diverse native plant community, while intensive gardening resulted in the highest floral resources. While environmental interventions should always be goal-oriented, our findings suggest that installing native plants in large vacant spaces via broadcast seeding can be an inexpensive and ecologically viable method for urban greening.

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