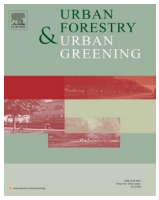
Urban Forestry & Urban Greening 124 (2026) 129564

Contents lists available at ScienceDirect

Urban Forestry & Urban Greening

journal homepage: www.elsevier.com/locate/ufug



Original article

Early establishment performance of hybrid *Populus* and *Salix* in canopy gaps of a post emerald ash borer urban forest

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ARTICLE INFO

Keywords:
Urban forest restoration
Populus
Salix
Monitoring
Genotype selection

ABSTRACT

Urban forests face challenges from invasive species, deer herbivory, and tree mortality, creating gaps where natural regeneration fails. We tested whether fast-growing hybrid *Populus* and *Salix* genotypes could facilitate urban forest restoration in post-emerald ash borer (*Agrilus planipennis*) canopy gaps. We planted six genotypes of each genus using two methods—potted trees and direct-planted cuttings across five 21 × 21-meter plots in Baltimore, Maryland, USA. We assessed initial light availability above each planted tree as canopy leaf area index (LAI). From 2021–2023, we measured survival, growth, stress indicators, and photosynthetic performance for 720 trees.

Potted trees significantly outperformed cuttings, achieving 59% three-year survival compared to only 14% for cuttings. Among potted trees, *Populus* genotypes grew 73% more than *Salix* counterparts with no significant survival difference. Light availability strongly influenced establishment success, *Populus* showed greater sensitivity to shade than *Salix*, exhibiting declining survival and increasing stress as LAI increased. Optimal planting zones occurred where overhead LAI remained below 1.4. Areas where established cover had an LAI > 2.7 presented elevated mortality risk. Genotype variation emerged within both genera, with superior performers identified through comprehensive multi-trait rankings via phyto-recurrent selection.

These results demonstrate that *Populus* and *Salix* genotypes planted as potted stock can successfully establish in challenging urban environments when planted in canopy gaps with adequate light and when appropriate genotypes are selected. The rapid growth offers significant advantages for restoration goals, providing early canopy closure to suppress invasive species while reducing long-term maintenance requirements in urban forest restoration projects.

1. Introduction

Urban forested natural areas in the eastern United States (U.S.) are extensive and provide important biophysical and socio-cultural ecosystem services (Johnson et al. 2020). These urban green spaces are ecosystems dominated by trees and other woody vegetation, where decomposition and natural regeneration processes occur in the understory (i.e., no mowing or other intensive management). Cities contain forests on both public and private land (Sonti et al. 2025), with varying management regimes (Pregitzer et al. 2021; Morzillo et al. 2022). These green spaces provide critical access to nature for local communities (Sonti, 2020) and habitat for native biodiversity (Trammell et al. 2020)), but active management is required to sustain healthy forest ecosystems (Simmons et al. 2016; Piana et al. 2021a).

Mature forests in cities typically have a canopy dominated by native tree species. In contrast, the understory and midstory of these same forests are often dominated by non-native plant species (Pregitzer et al. 2019). Additionally, natural tree regeneration in these forests is limited compared to rural forests (Piana et al. 2021b). This lack of regeneration likely results from seed predation and herbivory by deer (Piana et al. 2019). These factors create a sustainability crisis for mature forests in cities where there is a lack of advance tree regeneration when individual canopy dominant trees are lost due to storms, insects, and/or diseases. The situation becomes even more serious when a whole genus of trees is

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https://doi.org/10.1016/j.ufug.2026.129564
Received 27 February 2026; Received in revised form 27 May 2026; Accepted 10 June 2026
Available online 13 June 2026
1618-8667/Published by Elsevier GmbH.

impacted as in the case of emerald ash borer killing all *Fraxinus* spp. in the forest (Poland and McCullough, 2006). This loss of a whole cohort of canopy dominant trees is exacerbated by the presence of non-native plant species which can quickly take advantage of newly formed gaps, utilizing the additional light, water, and nutrients. To address this sustainability crisis, innovative forest management approaches are needed that can adjust the successional trajectory of these forests towards native tree species while controlling the non-native understory vegetation (Piana et al. 2026).

Urban forest restoration is a labor intensive, long-term process requiring ongoing maintenance and care (Bounds et al. 2015; Simmons et al. 2016). Current practices involve site preparation (invasive plant control) and planting, coupled with continued stewardship. Typical tree planting palettes include a mix of native species that may or may not be tailored to the planting site. This planting stock usually consists of 1- to 2-gallon potted trees, giving the newly planted trees a robust root system and a head start on height growth. Even so, newly planted urban forest restoration sites face risks of the planted trees being overwhelmed by non-native vegetation or deer browse. This approach necessitates continued resource investment focused on tree care and invasive species management.

Urban forest restoration efforts can benefit from the use of fast-growing tree species to reduce maintenance costs and the time needed to maintain deer protection. These early successional species rapidly grow tall enough to prevent terminal leader browsing, which allows them to achieve canopy closure and helps reduce pressure from shade intolerant invasive species. Early successional or pioneer tree species are the first to colonize disturbed rural sites (e.g., after landslides, fires, floods). They grow rapidly and establish new canopies faster than competing vegetation. Hybrid poplar (*Populus*) and willow (*Salix*) trees have been bred for rapid growth, reaching heights of almost 13 m in 6 years (Labrecque and Teodorescu, 2005; Čížková et al. 2010; Nelson et al. 2019). Because of this fast growth, these species are used in short rotation woody crop applications focused on rapid sustainable biomass production (Nelson et al. 2018; Zalesny and Pilipović 2022a), as well as phytoremediation applications (Volk et al. 2006; Pilipović et al. 2019). In addition to their rapid growth, *Populus* spp. and *Salix* spp. have the advantage of easy clonal propagation, significantly shortening the tree supply pipeline and drastically reducing production costs (Lazarus et al. 2015; Volk et al. 2016). To date, hybrid *Populus* and *Salix* genotypes have not been widely utilized in urban forest restoration projects. Furthermore, clone selection methods may allow for improved planting performance in site-specific contexts. For example, in phytoremediation a process called phyto-recurrent selection is used to identify specific clones that can survive under harsh chemical conditions that exist in specific sites slated for remediation (Zalesny et al. 2007). When applied to urban forests, identifying adapted clones may improve survival, rate of growth, and improve the ability of these trees to rapidly close canopy gaps, suppress invasive plant establishment, and serve as a stop gap for further forest restoration activities.

In this study, we leveraged the rapid growth and ease of propagation of hybrid *Salix* and *Populus* trees to conduct a forest restoration study in Baltimore, Maryland, USA. Our objectives were to: (1) assess the viability of hybrid *Salix* and *Populus* plantings for forest restoration in urban areas, (2) compare performance of *Salix* and *Populus* genera, (3) identify high-performing genotypes within these genera, (4) evaluate the feasibility of direct planting of hybrid *Salix* and *Populus* cuttings for restoration efforts.

We hypothesized that:

1. *Hybrid Salix and Populus trees will survive and grow in an urban forest impacted by the loss of mature canopy-dominant ash trees (Fraxinus spp.) killed by the emerald ash borer (Agrilus planipennis).*
2. *There will be differences in performance and survival between Salix and Populus genera.*
3. *There will be growth and survival differences among individual genotypes within and between genera, and between trees planted as cuttings and those grown in pots.*

2. Methods

2.1. Study site

This study was installed in a 4-hectare forest patch owned by Stillmeadow Community Fellowship, a community church located in southwest Baltimore, Maryland, USA (Fig. 1).The forest canopy was previously dominated by mature green and white ash trees (*Fraxinus pennsylvanica* Marsh. and *F. americana* L.) that were killed by emerald ash borer (EAB) prior to study installation in 2021. In 1927, the site was mostly farmland with few trees located primarily along streambanks (Lagrosa et al., 1927). The study site is representative of Baltimore's many small, privately-owned forest patches, some of which are similarly degraded (Baker et al. 2025; Sonti et al. 2025). The Stillmeadow site conditions are commonly found across urban forest patches: steep slopes, degraded soils (construction fill), recent reforestation, EAB related mortality, invasive plants, and deer. Forest patches like this are common throughout Baltimore and other cities in the northeastern U.S. (Harnik et al., 2017).

2.2. Tree species

We chose hybrid *Populus* and *Salix* that are commonly used in phytoremediation applications such as brownfields, landfills, mine sites, and other liability lands (Burges et al. 2018). These genera are early successional, are easily propagated vegetatively, exhibit fast growth, have extensive root systems, and elevated hydraulic control potential that make them ideal for phytoremediation and associated phytotechnologies (Licht and Isebrands, 2005; Zalesny et al. 2019a; Pilipović et al. 2020) We planted six clones from each genus into the study design (see Fig. 2 and Table 1).

2.3. Experimental design

Our study employed a randomized complete block design, incorporating two treatments across the Stillmeadow property, as detailed in Fig. 2. These treatments consisted of two planting strategies, potted plants and cuttings. We organized the experiment into plots, each measuring 21 × 21 m, and replicated each plot five times.

To ensure uniformity and minimize external variations, we planted the trees at a spacing of 1.2 m apart, allotting 81 trees to each treatment. To mitigate edge effects, we focused our measurements on 36 trees situated at the core of each plot quadrant, maintaining a buffer of two trees from the periphery, as illustrated in Fig. 2.

The final experimental design included equal numbers of cuttings and potted trees within each genus (n = 720; 180 *Salix* cuttings, 180 *Salix* potted trees, 180 *Populus* cuttings, 180 *Populus* potted trees), with 30 individuals per genotype per planting type.

2.4. Site preparation and planting

Plots were established on relatively flat terrain, with slopes of less than 10%. To prepare each plot area, all large dead ash trees were cleared, and the understory vegetation was removed using a brush saw. Additionally, a 2.4 m tall deer fence was installed around the perimeter of each treatment plot. In February 2020, dormant *Populus* and *Salix* cuttings (70 of each genotype) were gathered from a USDA Forest Service stoolbed in Rhinelander, WI. *Populus* cutting caliper ranged from 6.8 to 10 mm (mean per genotype) and *Salix* cutting caliper ranged from 7.5 to 8.4 mm (mean per genotype). These cuttings were refrigerated at 1.7°C until just before planting. In the following spring, the cuttings were soaked in water (2/3rds of the cutting submerged) for 36 h and

Fig. 1. Location of the study site in Baltimore, MD. Plots 1–5 shown on the map.

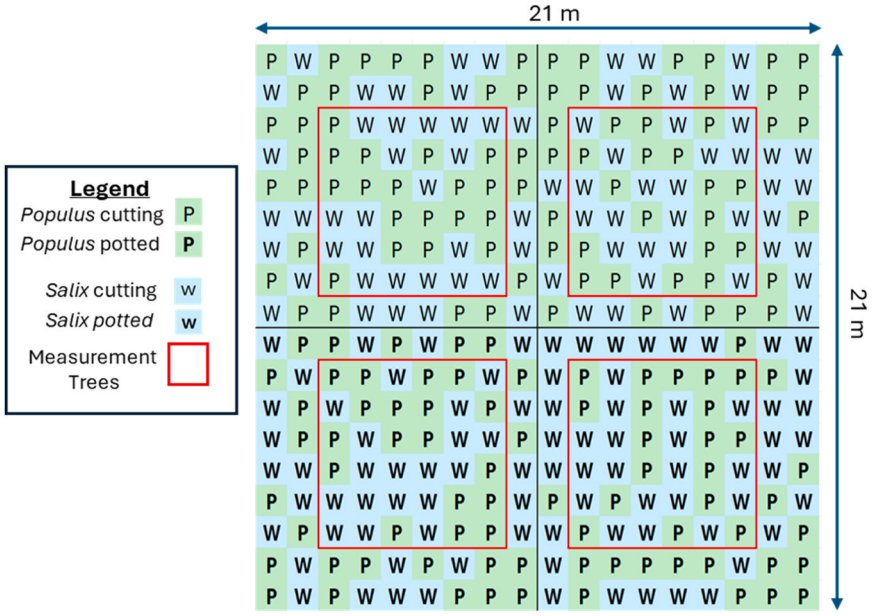

Fig. 2. Example Replicate plot Design: Treatments randomly assigned to each replicate. Light blue squares depict the random placement of Salix genotypes, while light green squares indicate the random placement of Populus genotypes. Bold text signifies potted tree plantings, while unbolded text represents cuttings. Central measurement trees are outlined in red.

then planted in Trade 2-gallon nursery pots (approximate volume = 1.5 gallons, https://reachsupplies.com/product/hp175/) filled with potting soil (Metro-Mix 852, https://www.sungro.com/professional-product/metro-mix−852/). No fertilizer was added. They were grown for one year in an on-site volunteer-managed nursery before planting in June 2021. Potted trees were planted in holes made using a planting auger. In February 2021, dormant cuttings (70 of each genotype) were again collected from the Rhinelander, WI stoolbed, refrigerated, and soaked in water for 36 h prior to direct planting in June 2021. Direct planting of the cuttings involves poking a hole in the ground and inserting the cutting so 1–2 buds are above the ground and closing the hole around the cutting with a trowel.

Before planting, the experimental plots were arranged on a grid with a spacing of 1.2 m between trees (Fig. 2). Planting locations were

Table 1
Populus and Salix genomic groups and genotypes used in this urban forest restoration study. The values in the "Genotype" column are used to refer to each genotype throughout the manuscript.

Genotype	Genomic Group
Populus[a]	
9732–11	*Populus deltoides* × *Populus nigra*
9732–31	*Populus deltoides* × *Populus nigra*
DN34	*Populus deltoides* × *Populus nigra*
DN5	*Populus deltoides* × *Populus nigra*
NM2	*Populus nigra* × *Populus maximowiczii*
DN2	*Populus deltoides* × *Populus nigra*
Salix[b]	
Fabius	*Salix viminalis* × *Salix miyabeana*
Millbrook	*Salix purpurea* × *Salix miyabeana*
Sherburne	*Salix miyabeana*
SX61	*Salix miyabeana*
SX67	*Salix miyabeana*
Tully Champion	*Salix viminalis* × *Salix miyabeana*

[a] Sections and authorities for Populus are: Aigeiros Duby - P. deltoides; Bartr. ex Marsh, P. nigra L.; Tacamahaca Spach—P. Maximowiczii. [b]Sections and authorities for Salix are: Helix Dumortier—S. purpurea L., S. miyabeana Seemen; Viminella Seringe—S. viminalis L.

randomly assigned to each genotype. Volunteers assisted in planting the potted trees and cuttings from May 15 to June 10, 2021. Following planting, the locations of each genotype were verified against the planting map, and the height of all potted trees was measured.

All planted trees and cuttings were periodically hand weeded during each growing season by high school and young adult interns and workforce training program participants at Stillmeadow Community Fellowship. The plantings were watered several times during a drought period in the first growing season, but not in subsequent years. These crews also assisted with construction and maintenance of the deer fence surrounding the plots. The site conditions made it difficult to properly install the deer fence and consistently keep it intact (e.g., slopes, lack of mature trees to attach fence, buried asphalt), and white-tailed deer (*Odocoileus virginianus*) did periodically breach the plot fences during the three years of the experiment.

2.5. Tree measures

At the end of each growing season (September 2021, 2022, 2023) we conducted a field survey of the 36 centrally located trees in each quadrant of every treatment plot for a total of 720 trees. We recorded whether each tree was alive, dead, or missing and took a height measurement. Height was taken from soil level to the apex of the terminal leader. If a tree had multiple stems, we recorded the height of the tallest one.

Tree health assessments were conducted for every tree using methodology adopted from Pontius and Hallett (2014) with modifications made to account for the young age of the trees in this study. Specifically, we assessed trees for:

1. Drought stress as evidenced by dried or brown leaf area or the presence of live branches without leaves;
2. Leaf discoloration not due to drought ;
3. Pest damage, defined as evidence of insect or fungal damaged leaf area ;
4. Tree vigor which is an overall assessment of tree health as described in Table 2;
5. Photosynthetic capacity assessed using the Handy PEA chlorophyll fluorescence meter (Hansatech Instruments Ltd., England, U.K.).

Drought stress, leaf discoloration, and pest damage variables were recorded as percentages of the canopy impacted using the following classes (1 =0–1%, 2 =2–25%, 3 =26–50%, 4 =51–75%, and 5 =76–100%).

Table 2
Vigor rating rubric used in tree health assessment.

Class	Rubric
1	Tree appears to be in reasonably good health with no major branch mortality or large broken branches. **Less than 10% cumulative** defoliation, discoloration, and crown dieback is present.
2	10–25% cumulative defoliation, discoloration, and crown dieback is present **and/or** crown area missing based on visual evidence of large broken (not pruned) or dead branches **less than 26%.**
3	26–50% cumulative defoliation, discoloration, and crown dieback is present **and/or** crown area missing based on visual evidence of large broken (not pruned) or dead branches **less than 50%.**
4	More than 50% cumulative defoliation, discoloration, and crown dieback is present **and/or** crown area missing based on visual evidence of large broken (not pruned) or dead branches **more than 50%.**
5	Tree is dead or missing.

Planting in forest gaps creates variable light conditions. Overhead canopy leaf area index (LAI) was measured once in September 2021 above each planted individual to quantify light availability at each planting location using a LAI-2200C Plant Canopy Analyzer (LI-COR Environmental). LAI values were used to assess how canopy cover influenced survival, growth, and physiological stress during the establishment period. We analyzed the relationship between LAI and plant performance using correlation analysis, linear regression, and by comparing LAI values between performance groups (e.g., survived vs. dead individuals). The periodic weeding mentioned above helped to maintain the initial light conditions measured in September 2021.

2.6. Data analysis

2.6.1. Survival analysis

Survival rates were calculated as the percentage of survivors relative to the original planting numbers (180 individuals per genus × planting type combination). Genus and planting type effects on survival were analyzed using generalized linear mixed-effects models (GLMM) with binomial distribution and logit link function, incorporating plot as a random effect to account for spatial clustering. Genotype effects within each genus × planting type combination were evaluated using chi-square tests followed by GLMMs when significant effects were detected. Post-hoc pairwise comparisons were conducted using estimated marginal means with Tukey adjustment for multiple comparisons at a probability value of $\alpha = 0.05$. The influence of light availability on survival was evaluated by comparing overhead canopy LAI between individuals that survived versus those that died using Welch's *t*-tests, which do not assume equal variances. These comparisons were conducted for overall survival (2021 and 2023) and stratified by planting type (trees and cuttings). For cuttings, we additionally stratified analyses by genus to identify genus-specific light responses. Effect sizes were calculated using Cohen's d (effsize package in R) to quantify the magnitude of differences in light availability between survival groups.

2.6.2. Growth analysis methods

Annual height measurements were taken at the end of each growing season, and growth increments were calculated as the difference between consecutive annual measurements. Total growth over the three-year establishment period was calculated as the difference between final height (September 2023) and initial planting height (June 2021) for surviving potted trees only, as cuttings had poor survival after 3 years.

Annual growth differences between planting types (cuttings vs. potted trees) were analyzed using linear mixed-effects models with annual growth as the response variable, planting type as a fixed effect, and plot as a random effect. Species differences in total growth were analyzed using linear mixed-effects models with total growth as the response variable, genus as a fixed effect, and plot as a random effect.

Pairwise comparisons between groups were assessed using *t*-tests.

The relationship between overhead canopy leaf area index (LAI) and height growth was evaluated using Pearson correlation analysis for 2021 growth, 2023 growth, and average annual growth. Higher LAI values indicate greater canopy cover and reduced light availability. Linear regression was used to quantify the change in growth per unit increase in canopy LAI. Additionally, mixed-effects models were fitted with average annual growth as the response variable, LAI, genus, and planting type as fixed effects, and plot as a random effect to test the significance of LAI while controlling for spatial variation.

To evaluate whether the LAI effect differed between genera, we fitted separate mixed-effects models within each genus, with average annual growth as the response variable, LAI and planting type as fixed effects, and plot as a random effect. We also tested for a LAI × genus interaction in the overall model to statistically confirm whether the two genera responded similarly to overhead canopy conditions.

Genotype effects on total growth of potted trees were evaluated separately within each genus using one-way ANOVA followed by Tukey's HSD post-hoc tests when significant differences were detected ($p < 0.05$). Mixed-effects models incorporating plot as a random effect confirmed genotype effects while controlling for spatial variation.

Statistical analyses were performed in R version 4.3.0 (R Core Team, 2013) using the lme4 (Bates et al. 2015) and lmerTest (Kuznetsova et al. 2017) packages for mixed-effects models, and the emmeans package (Lenth and Piaskowski, 2017) for post-hoc comparisons. All models included plot as a random effect to account for spatial clustering in the blocked experimental design. Significance was assessed at $\alpha = 0.05$.

2.6.3. Stress index analysis

To compare levels of health and vitality for each tree (cuttings and potted trees) in the study, we calculated a Z-score for individual health variables (drought, discoloration, pest, and vigor). These Z-scores were then averaged for each tree, creating a stress index score (Pontius and Hallett, 2014). Low values represent healthier trees, and high values represent trees that are showing signs of more severe stress.

Stress index levels were compared between planting types (cuttings vs. potted trees) and genera (*Salix* vs. *Populus*) in 2021. The analysis was completed on trees that were alive at the end of the 2021 growing season (n = 517), The dataset comprised 163 cuttings (71 *Salix,* 92 *Populus*) and 354 potted trees (178 *Salix,* 176 *Populus*). A linear mixed-effects model was fitted using the lmer function in the lme4 package (Bates et al. 2015) with stress index as the response variable, planting type and genus as fixed effects (including their interaction), and plot as a random effect. Post-hoc pairwise comparisons were conducted using the emmeans package with Tukey adjustment for multiple comparisons.

Additionally, Welch's *t*-tests were performed to compare stress levels between planting types both overall and within each genus, as these tests do not assume equal variances between groups. Effect sizes were calculated using Cohen's d, with values of 0.2, 0.5, and 0.8 representing small, medium, and large effects, respectively (Cohen, 2013). The relationship between overhead canopy LAI and physiological stress was evaluated using Pearson correlation analysis, conducted overall and stratified by genus to identify genus-specific responses to light availability. Correlations were calculated for both 2021 stress levels and temporal changes in stress between 2021 and 2023.

Stress index levels were also compared among genotypes within each genus and planting type in 2021, and among potted tree genotypes in 2023. The 2021 analysis included all individuals that survived through 2021 (n = 517), while the 2023 analysis was restricted to potted trees that survived through 2023 (n = 214). Only genotypes with a minimum sample size of three individuals were included in statistical comparisons to ensure reliable estimates.

Genotype performance was evaluated using one-way ANOVA within each genus × planting type combination, with physiological stress index as the response variable and genotype as the fixed effect. Post-hoc pairwise comparisons were conducted using Tukey's HSD test when ANOVA indicated significant genotype effects ($p < 0.05$). To assess temporal stability of genotype performance, stress changes between 2021 and 2023 were calculated for potted trees, and genotype rankings were compared between years. Effect sizes were quantified using Cohen's d, and genotypes were ranked from lowest (healthiest) to highest (most stressed) stress levels within each group.

2.6.4. Stress threshold analysis

To identify potential stress thresholds affecting tree growth, we performed segmented regression analysis using a grid search approach in R (version 4.5.1). We tested candidate breakpoints at 0.02-unit intervals between the 10th and 90th percentiles of the average stress index distribution (2021–2023) for trees surviving through 2023 (n = 214). For each candidate breakpoint, we partitioned the data into two segments and fitted separate linear regressions relating average stress index to 2023 growth. To ensure adequate statistical power, we required a minimum of 10 observations in each segment. The optimal breakpoint was identified as the value minimizing the combined residual sum of squares (RSS) from both segments. This approach identified a breakpoint at a stress index value of 0.97, which separated trees into a low-stress group (n = 64) exhibiting robust growth (mean = 1.23 m) and a higher-stress group (n = 150) with reduced growth (mean = 0.27 m). Additionally, we calculated the theoretical zero-growth threshold by solving the overall regression equation (Growth = 2.107–1.237 × Stress) for zero growth, yielding a critical threshold of 1.704. The significance of growth differences above and below each threshold was assessed using Welch's *t*-tests, and effect sizes were calculated using Cohen's d.

2.6.5. Performance index analysis

Performance Index (PI), a measure of photosynthetic efficiency and plant vigor (Živčák et al. 2014), was analyzed to evaluate differences between planting types and genotypes across genera and years. Three primary comparisons were conducted: (1) potted trees versus cuttings by genus in 2021, (2) temporal changes in potted trees between 2021 and 2023 by genus, and (3) genotype performance within each genus × planting type combination. The 2021 analysis included all individuals alive in 2021 with available PI measurements (n = 490), while the 2023 analysis was restricted to potted trees that survived through 2023 (n = 213).

Statistical approaches varied by analysis type. For trees versus cuttings comparisons by genus, mixed-effects models were fitted using lmer () with species and planting type as fixed effects and plot as a random effect to account for spatial clustering. Individual genus comparisons used Welch's *t*-tests to accommodate unequal variances, with effect sizes calculated using Cohen's d. Temporal changes in trees between 2021 and 2023 were assessed using Welch's *t*-tests for each genus separately. For genotype comparisons, one-way ANOVA was performed within each genus × planting type × year combination using standard aov() models, with only genotypes having a minimum sample size of three individuals included in analyses. When ANOVA indicated significant genotype effects ($p < 0.05$), post-hoc pairwise comparisons were conducted using Tukey's HSD test.

2.6.6. Genotype performance analysis

We conducted a comprehensive multi-trait genotype performance analysis to evaluate restoration potential of *Salix* and *Populus* genotypes in the urban forest environment. The analysis included only potted tree plantings. Four key performance metrics were evaluated: 2023 survival rates, average annual growth 2021–2023 (m), stress index scores, and photosynthetic performance index.

For each species separately, we ranked genotypes from 1 (best) to 6 (worst) across all four performance traits, with survival and growth ranked in descending order (higher values = better ranks), stress index ranked in ascending order (lower stress = better ranks), and performance index ranked in descending order (higher PI = better ranks). We

calculated weighted comprehensive performance rankings using the formula: (Survival rank × 2 + Growth rank × 2 + Stress rank × 1 + Performance Index rank × 1) ÷ 6. This weighting system emphasizes survival and growth as the most critical factors for successful urban forest restoration, reflecting their fundamental importance for establishment success and restoration objectives.

Genotypes were categorized into performance tiers based on weighted average rankings: Superior (≤2.5), Good (2.6–3.5), Average (3.6–4.25), and Poor (>4.25). We identified performance trait specializations and trade-offs by examining genotypes with rank 1 performance in heavily-weighted traits (survival or growth) combined with poor performance in other metrics, indicating specialized rather than balanced performance. Growth-survival trade-offs were specifically evaluated by identifying genotypes ranking first in growth but fifth or sixth in survival, and vice versa.

The weighted ranking methodology provides restoration-focused, quantitative guidance for genotype selection in urban forest restoration applications while identifying both generalist performers suitable for broad application and specialist genotypes optimized for specific site conditions where survival or growth may be prioritized. This methodology is an urban application of the phyto-recurrent selection process used in phytoremediation settings (Zalesny et al. 2007).

3. Results

3.1. Survival

Planting method had a statistically significant interaction with both genus and genotype in affecting initial establishment success. Both *Salix* and *Populus* genotypes achieved excellent 2021 survival as potted trees (>97%, Fig. 3), with individual genotype survival ranging from 93% to 100%. *Salix* cutting genotypes ranged from 27% to 57% survival, while *Populus* cutting genotypes ranged from 23% to 67%. Genotypes `9732–31` (67%) and `Millbrook` (57%) had superior establishment compared to poor performers such as `DN34` (23%).

At the end of the third growing season (2023), potted trees had a 59% survival rate while cuttings had only 14% survival. *Salix* cuttings achieved only 7% survival while *Populus* cuttings reached 22%. In contrast, potted trees of both genera maintained moderate survival rates (*Salix* 61%, *Populus* 57%) with no significant genus difference.

Populus genotypes planted as potted trees had survival ranging from 50% (`9732–11`, `9732–31`) to 70% (`NM2`) however there were no significant differences between genotypes. *Populus* genotypes planted as cuttings had survival rates ranging from 13% (`DN34`) to 40% (`9732–31`) with no significant differences.

Salix genotypes planted as potted trees showed significant genotype differences in 2023 survival rates, with `Millbrook` achieving 80% survival compared to 40% for `SX61`. More critically, *Salix* cuttings demonstrated highly significant genotype effects, with `Millbrook` as the only viable genotype (23% survival) while `SX61` showed complete failure (0% survival).

Light availability, measured as overhead canopy LAI, influenced first-year cutting establishment in a genus-specific manner but did not affect long-term survival. *Populus* cuttings demonstrated significant light sensitivity in 2021, with individuals that died experiencing greater canopy cover compared to survivors (LAI: 2.37 ± 1.24 vs. 2.04 ± 0.89; $t = 2.03$, $p = 0.044$). In contrast, *Salix* cutting survival was not significantly affected by canopy LAI in 2021 ($p = 0.216$). By 2023, canopy LAI no longer predicted survival for cuttings of either genus ($p > 0.05$).

3.2. Growth

During the first growing season, the average growth for planted cuttings was 33.7 cm while potted trees grew 13.4 cm. Cuttings showed significantly higher growth increments despite starting from much smaller initial sizes. By the end of the first growing season, the average height of cuttings was 33.7 cm while potted trees reached 139.4 cm, demonstrating the substantial initial size advantage of nursery-grown stock (Fig. 4).

By the third growing season, this pattern had reversed completely. Cuttings grew an average of only 16.3 cm during 2023 while potted trees grew 55.7 cm. By the end of the third growing season, the average height of surviving cuttings was 96.8 cm while potted trees reached 274.6 cm.

Over the entire three-year establishment period, significant genus differences emerged in total height gain. *Populus* trees achieved 186.5 ± 20.6 cm mean total growth compared to 107.8 ± 11.7 cm for *Salix* trees, representing a statistically significant 78.7 cm advantage ($p < 0.05$) (Fig. 5), representing 73% greater total growth for *Populus* over the study period.

Overhead canopy LAI significantly influenced growth rates throughout the establishment period. During the first growing season, cuttings showed a significant negative relationship between canopy LAI and growth ($r = -0.155$, $p = 0.048$), with individuals receiving more light achieving greater growth. When accounting for spatial variation using mixed-effects models, canopy LAI emerged as a highly significant predictor of average annual growth ($p < 0.001$), with trees experiencing approximately 12.9 cm less annual growth per unit increase in overhead

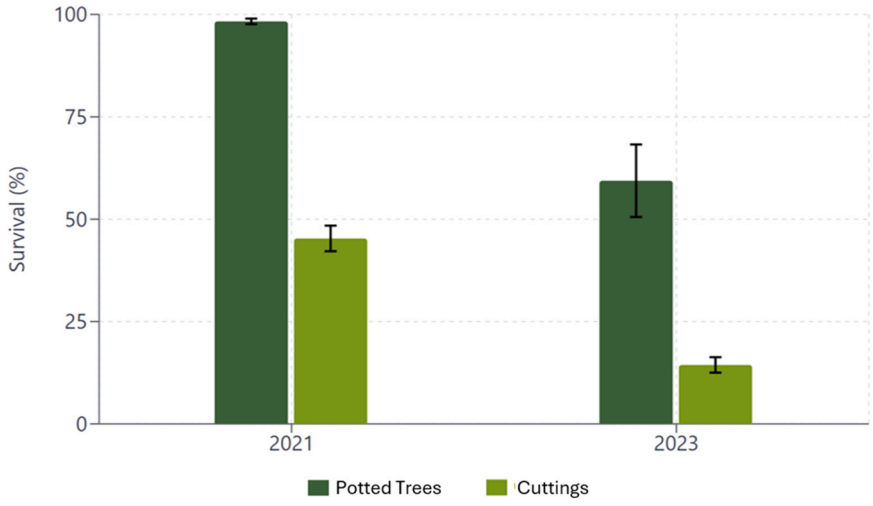

Fig. 3. Percent survival of potted trees versus cuttings (genera combined) at the end of the growing season in 2021 and at the end of the third growing season (2023). Error bars are standard error of the mean (SEM, n = 360). Potted trees had significantly higher survival than cuttings.

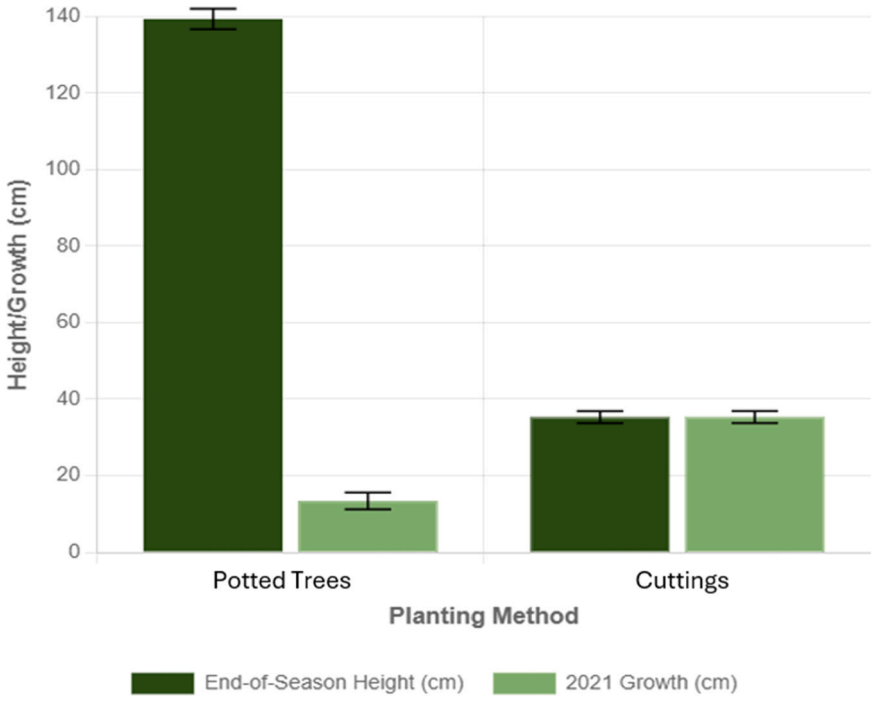

Fig. 4. End of season (year 1) height and growth for potted trees and cuttings. Cuttings grew more in year 1 however potted trees were taller at the end of the growing season. Error bars are (SEM, n = 360).

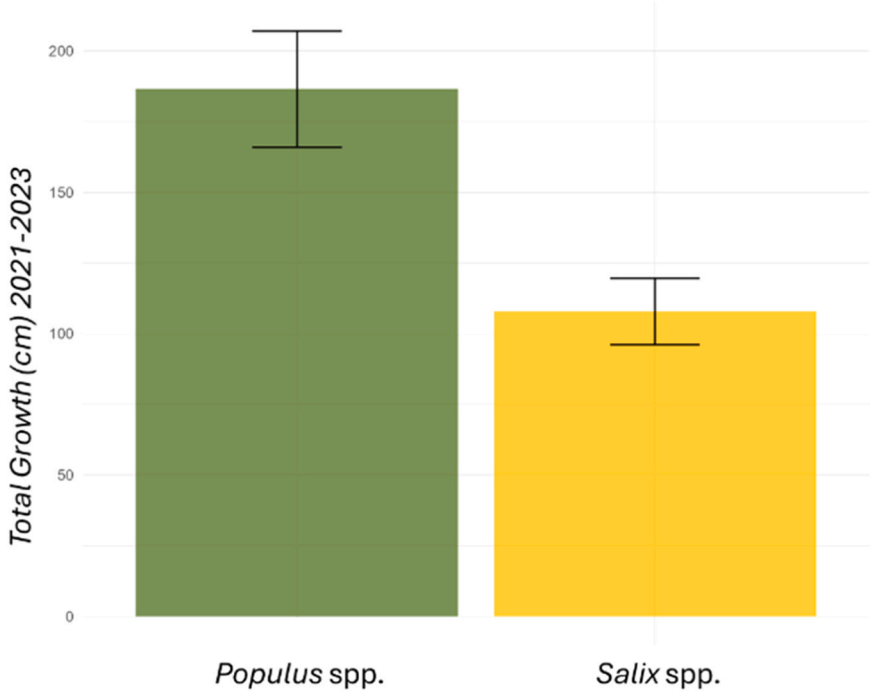

Fig. 5. Potted tree total growth across 3 growing seasons (2021–2023). Populus grew significantly more than Salix. Error bars are SEM, n = 360.

canopy cover.

When tested for differential responses between genera, the LAI × genus interaction was not significant ($p = 0.067$), statistically confirming that *Salix* and *Populus* responded similarly to overhead canopy conditions. Separate genus-specific analyses further confirmed significant negative LAI effects for both *Salix* ($p = 0.004$) and *Populus* ($p = 0.017$), with trees in higher light environments achieving approximately 9.8 cm and 14.3 cm greater annual growth per unit decrease in canopy LAI, respectively. This demonstrates that while both genera can establish under variable light conditions, growth performance was maximized in high-light microsites for both *Salix* and *Populus*.

Within-genus analysis of potted tree growth revealed important genetic variation. Among *Populus* genotypes, total growth of potted trees ranged from 146.9 cm (`NM2`) to 231.0 cm (`9732–11`), though differences were not statistically significant. *Salix* genotypes showed greater variation and significant differences, with total growth of potted

trees ranging from 52.4 cm (`Sherburne`) to 182.9 cm (`SX61`). Genotype `Sherburne` performed significantly worse than all other *Salix* genotypes, while `SX61` achieved the highest growth but with reduced survival (40% survival for `SX61` vs 53–80% for other *Salix* genotypes).

3.3. Tree health

Stress index levels in 2021 differed significantly between planting types, with cuttings exhibiting lower stress than potted trees overall (1.174 ± 0.576 vs. 1.319 ± 0.568; $t = -2.667$, $df = 311.3$, $p < 0.01$). The mixed-effects model revealed a significant main effect of planting type ($F_{1,511} = 7.12$, $p < 0.01$) and a significant planting type × genus interaction ($F_{1,511} = 12.84$, $p < 0.001$), indicating that the planting type effect varied between genera. The random effect of plot accounted for 8.3% of the total variance in stress levels.

Post-hoc analyses revealed that the overall planting type effect was driven entirely by differences within *Populus*, where cuttings showed substantially lower stress than potted trees (0.978 ± 0.430 vs. 1.343 ± 0.536; $p < 0.001$, Cohen's $d = -0.727$, large effect). In contrast, *Salix* showed no significant difference between planting types (cuttings: 1.429 ± 0.640, trees: 1.295 ± 0.599; $p = 0.158$, Cohen's $d = 0.219$, small effect). Within planting types, *Salix* cuttings exhibited significantly higher stress than *Populus* cuttings ($p < 0.001$), while stress levels were similar between genera for potted trees ($p = 0.721$).

Light availability emerged as a significant factor influencing stress patterns in a genus-specific manner during the establishment phase. Overhead canopy LAI showed a positive relationship with stress levels in 2021 ($r = 0.16$, $p < 0.001$), indicating that individuals under heavier canopy cover experienced greater physiological stress from light limitation. This relationship was driven almost exclusively by *Populus,* which showed strong positive correlations between canopy LAI and stress for both trees ($r = 0.404$, $p < 0.001$) and cuttings ($r = 0.239$, $p = 0.022$). In contrast, *Salix* showed no significant relationship between light availability and stress ($r = -0.05$, $p = 0.43$), suggesting genus-level differences in shade tolerance and stress response mechanisms. By 2023, tree stress was not significantly related to initial light conditions ($p = 0.086$), indicating successful acclimation to site light conditions among surviving individuals across both genera. These results indicate that cuttings exhibited fewer signs of stress during initial establishment in 2021 compared to potted trees, but this advantage was genus-specific and limited to *Populus*.

Significant genotype effects on signs of stress were detected within most genus-planting type combinations in 2021. Among *Salix* cuttings, `Millbrook` exhibited the lowest stress (1.278 ± 0.144), while `SX61` showed the highest stress (1.733 ± 0.181), representing a 0.455-unit range. For *Populus* cuttings, `NM2` demonstrated superior performance (0.747 ± 0.079) compared to `DN5` (1.173 ± 0.124), with a 0.426-unit difference. In potted trees during 2021, `SX67` outperformed other Salix genotypes (0.949 ± 0.069 vs. `W3` at 1.649 ± 0.123), while `NM2` led *Populus* genotypes (1.242 ± 0.071 vs. `9732–11` at 1.380 ± 0.114).

Temporal analysis revealed substantial changes in genotype rankings between 2021 and 2023. In 2023, `Fabius` emerged as the top-performing *Salix* genotype (1.167 ± 0.118), while the previous top performer, `SX67`, declined to third place following a 48.6% increase in stress levels (0.949–1.410). Similarly, `NM2` became the leading *Populus* genotype in 2023 (1.169 ± 0.138), improving from its 2021 ranking through a 15.1% stress reduction, while the former top performer, NM2, experienced a 24.6% stress increase (1.242–1.547).

3.3.1. Relationships between tree stress and growth performance

A linear relationship was found between tree stress levels and overall growth. Average stress (2021–2023) significantly predicted 2023 growth ($R^2 = 0.243$, $p < 0.001$), with each unit increase in stress reducing growth by 123.7 cm (Fig. 6). Analysis revealed two critical stress thresholds that define tree performance zones. A growth threshold at average stress index 0.97 separated high-performing from declining trees, with those below this level achieving 123 cm average growth compared to 27 cm for trees above the threshold (96 cm difference). A zero-growth threshold was identified at stress index 1.704. This threshold identifies trees with predicted growth of zero or less.

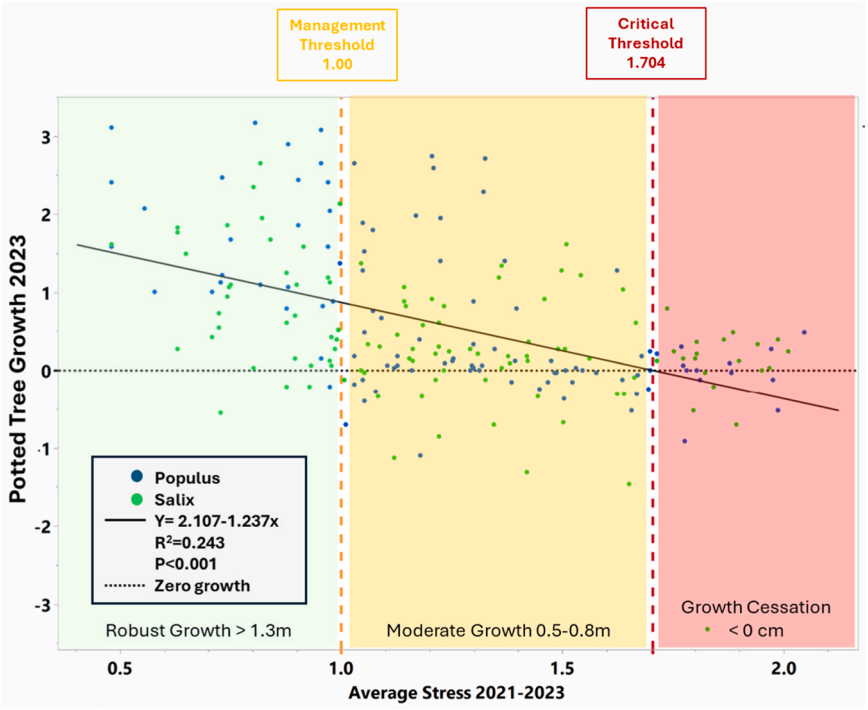

Fig. 6. Potted tree growth in 2023 versus average stress index across all three growing seasons. Stress index thresholds identified based on tree growth.

3.4. Photosynthetic efficiency

Performance Index values in 2021 differed significantly between planting types in *Salix* but not in *Populus*. *Salix* potted trees exhibited higher photosynthetic performance compared to cuttings (28.24 ± 12.49 vs. 23.23 ± 10.64; $t = -3.008$, $p < 0.01$, Cohen's $d = -0.415$), while *Populus* showed no significant difference between planting types (40.05 ± 18.82 vs. 39.23 ± 16.41; $t = -0.351$, $p = 0.726$) (Fig. 7). Temporal analysis revealed non-significant but contrasting trends between genera in potted trees, with *Salix* showing a moderate increase in PI from 2021 to 2023 (+2.81 units, +9.9%; Cohen's $d = 0.225$) while *Populus* experienced a moderate decline in PI (−2.46 units, −6.1%; Cohen's $d = -0.148$).

Genotype effects were significant within most genus-planting type combinations, revealing substantial performance variation. `DN34` emerged as consistently having the highest PI values across 2021 cuttings (44.6 ± 7.9), 2021 trees (49.1 ± 3.5), and 2023 trees (46.8 ± 3.1). In contrast, *Salix* genotype rankings showed temporal instability, with `SX67` leading cuttings (26.0 ± 3.6), `Tully Champion` leading 2021 trees (32.9 ± 2.2), and `SX61` emerging as the leading genotype in 2023 (35.4 ± 2.9). Photosynthetic performance ranges within genera varied considerably, spanning 7.3–13.7 PI units depending on the group examined, indicating substantial genetic variation in photosynthetic capacity and supporting the importance of genotype selection in urban forest restoration planning.

3.5. Comprehensive genotype performance rankings

The multi-trait performance analysis revealed substantial variation among genotypes within potted trees of both genera, with weighted average rankings ranging from 2.33 to 5.17 across all evaluated traits. For *Salix*, `Tully Champion` emerged as the top performer with a weighted average rank of 2.33, demonstrating exceptional balance across the heavily weighted survival (73.3%, rank 2) and growth (0.270 m, rank 3) traits, complemented by a low stress index (2.23, rank 2) and performance index (33.0, rank 2). Among *Populus* genotypes, `DN34` and `NM2` tied for best performance (2.83 weighted average rank), with `DN34` excelling in both growth (0.481 m, rank 3) and performance index (PI = 46.8, rank 1), while `NM2` demonstrated strength in survival (60.0%, rank 2) and growth (0.478 m, rank 4) metrics combined with a low stress index (2.38, rank 2).

Clear performance tiers emerged from the ranking analysis. The superior tier (≤2.5 weighted average rank) contained only `Tully Champion`, establishing it as the sole top-performing genotype. The Good tier (2.6–3.5 weighted average rank) included five genotypes: `DN34` and `NM2` (both 2.83), `Millbrook` (2.83), `9732–11` (3.17), and `NM2` (3.50). `Millbrook` benefited significantly from the survival weighting, rising in the rankings due to its exceptional 80.0% survival rate (rank 1) despite lower performance index scores. The Average tier (3.5–4.25 weighted average rank) contained `Fabius` (3.50), `SX61` (3.50), `SX67` (3.67), and `DN5` (4.17), with `SX61` demonstrating the highest growth rate among *Salix* (0.457 m, rank 1) but this advantage was offset by poor survival (40.0%, rank 6). The Poor tier (≥4.25 weighted average rank) contained `Sherburne` (5.17) and `9732–31`(4.50), both showing critical weaknesses in survival and growth metrics.

The weighted ranking system prioritized establishment-critical traits while maintaining sensitivity to physiological performance, providing a restoration-focused framework for genotype selection Table 3. These results provide quantitative evidence supporting Hypothesis 2 (significant performance differences between *Salix* and *Populus* genera) and Hypothesis 3 (substantial variation among genotypes and between planting methods), with the weighted comprehensive ranking system identifying `Tully Champion` as the optimal primary choice, followed by `DN34` and `NM2` as excellent secondary options for urban forest restoration applications.

4. Discussion

Urban forested natural areas can be challenging to restore and manage. In this case, the study site in Baltimore, Maryland (Fig. 1) lost all canopy dominant ash trees, thereby increasing available light to the forest floor. This benefitted invasive plants and vines, both native and non-native. Restoration of this site means establishing a closed canopy forest consisting of native tree species. Standard practice dictates planting a site appropriate mix of native tree species in one- or two-gallon pots 1.2 m on center, a higher density than is typically

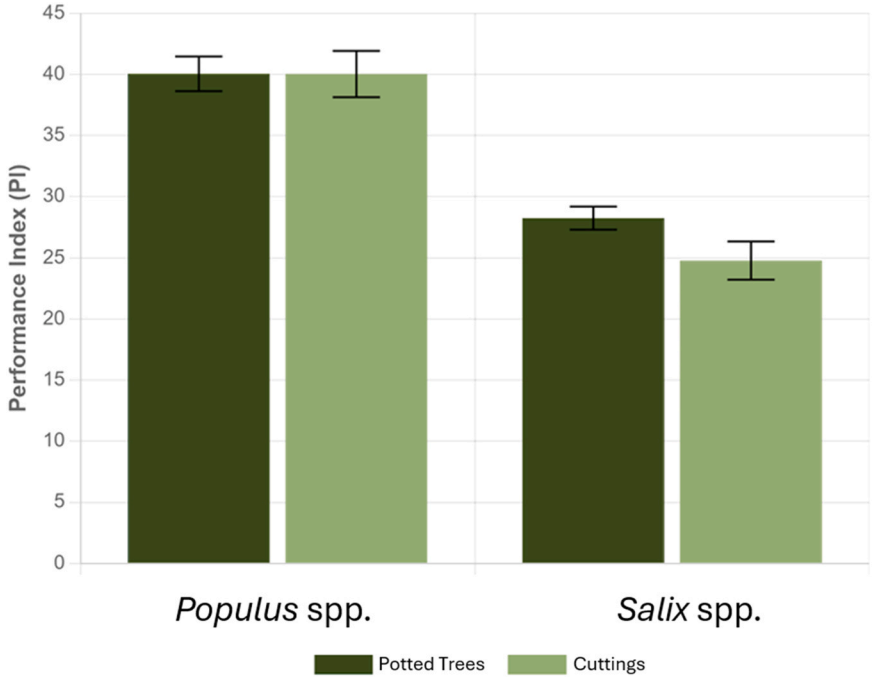

Fig. 7. Performance index for Populus and Salix planted as potted trees and cuttings. Error bars are SEM, n = 180.

Table 3
Genotype ranks for each variable. Survival and growth were weighted as they are more important for restoration success.

***Salix* spp.**												
Rank	**Genotype**	**n**	**Survival Rate (%)**	**Growth Rate (m/yr)**	**Stress Index**	**PI**	**Survival Rank**	**Growth Rank**	**Stress Rank**	**PI Rank**	**Average Rank**	**Category**
1	W6	30	73	0.27	2.23	33.0	2	3	2	2	2.33	Superior
2	W2	30	80	0.26	2.03	27.1	1	4	1	6	2.83	Good
3	W1	30	40	0.32	2.48	29.0	4	2	4	5	3.50	Good
3	W4	30	67	0.46	3.00	35.4	6	1	6	1	3.50	Good
5	W5	30	57	0.23	2.34	30.2	3	5	3	3	3.67	Average
6	W3	30	53	0.13	2.88	29.7	5	6	5	4	5.17	Poor
***Populus* spp.**												
Rank	**Genotype**	**n**	**Survival Rate (%)**	**Growth Rate (m/yr)**	**Stress Index**	**PI**	**Survival Rank**	**Growth Rank**	**Stress Rank**	**PI Rank**	**Average Rank**	**Category**
1	P3	30	57	0.48	2.75	46.8	3	3	4	1	2.83	Good
1	P5	30	60	0.48	2.38	36.5	2	4	2	3	2.83	Good
3	P1	30	50	0.58	2.79	36.6	5	1	5	2	3.17	Good
4	P6	30	70	0.37	2.34	33.4	1	6	1	6	3.50	Good
5	P4	30	57	0.39	2.71	36.3	3	5	3	4	3.83	Average
6	P2	30	50	0.55	2.87	35.0	5	2	6	5	4.17	Average

practiced in rural settings (Bounds et al. 2015) This approach is expensive and labor intensive. For this reason, we designed a study that utilizes fast growing early successional species that can be clonally propagated from cuttings.

Our three-year study demonstrates that hybrid *Populus* and *Salix* can successfully establish in post-emerald ash borer urban forest gaps, though performance varies significantly by planting method, genus, and genotype. In support of our first hypothesis, we found evidence supporting the feasibility of using fast-growing, clonally propagated species for urban forest restoration, with implications for both short-term canopy recovery and long-term successional trajectories.

4.1. Establishment success and planting method effects

There was a dramatic contrast between establishment success of potted trees and cuttings. While both genera achieved excellent initial survival rates when planted as potted trees (>97%), cutting survival was substantially lower. The 51% survival rate for *Populus* cuttings compared to 39% for *Salix* cuttings, declining further to 22% and 7% respectively by year three, highlights the harsh establishment conditions in urban forest environments (Bounds et al., 2015; Brunzel et al., 2009; Pauchard et al., 2006).

Light availability emerged as a critical factor influencing both survival and growth during establishment, though its effects varied by planting method and genus. For *Populus* cuttings only, greater canopy cover (LAI) was significantly related to higher mortality in 2021. However, by 2023, canopy LAI no longer predicted survival for cuttings of either genus, indicating that light availability was critical only during the initial establishment window. LAI measurements were not taken after 2021, however weeding around each tree was designed to limit changes in light availability over the course of this study. Assuming light conditions did not change, tree survival was not affected by canopy LAI in either year, demonstrating that potted trees possessed sufficient resource reserves to tolerate variable light conditions throughout the study period.

While initial light availability effects on survival were temporary and genus-specific, tree growth was impacted by initial light availability throughout the establishment period for both genera and both planting methods. The persistence of light-growth effects beyond the critical survival window suggests two distinct phases of light limitation: an initial establishment phase where light determines survival success for *Populus* cuttings, followed by a sustained growth optimization phase where light availability influences height increment rates for all surviving individuals. This dual role of light—as both a survival filter during initial establishment and a growth determinant throughout development, has important implications for microsite selection in urban forest restoration.

These results contrast sharply with short rotation woody crop applications where cutting establishment typically exceeds 90% (Pilipovic et al., 2021; Zalesny et al. 2021). Urban forest conditions present a unique combination of stressors including soil compaction, altered hydrology, air pollution, and intense competition from established non-native vegetation that rural agricultural settings lack (Mejia et al., 2024, Pregitzer et al., 2016).

The importance of light availability documented in our study further distinguishes urban forest restoration from agricultural and phytoremediation settings, where canopy competition is typically minimal or controlled through vegetation management (Zalesny et al., 2021). Furthermore, the sloped terrain and narrow trail access at our study site do not allow for systematic mowing and maintenance typically practiced in short rotation woody crop applications (Dickmann, 2006). The initial height advantage provided by potted stock (139.4 cm versus 35.3 cm for cuttings at the end of the 1st growing season) proves decisive in helping trees compete for light and avoid being overwhelmed by competing vegetation, particularly in shadier microsites where light limitation compounds other establishment stressors.

The superior first-year growth performance of cuttings (33.7 cm versus 13.4 cm for potted trees) demonstrates their potential vigor when establishment succeeds. However, this advantage disappears by year three as surviving potted trees achieve 55.7 cm annual growth compared to only 16.3 cm for cuttings. This suggests that while cuttings initially allocate more resources to shoot growth, this may be insufficient to overcome the combined effects of competition, light limitation, and other stressors in the long term. The documented negative relationship between canopy LAI and growth for both planting methods indicates that surviving cuttings continued to experience light-limited growth, potentially explaining their reduced height increments in later years compared to the taller potted trees that could access higher light strata. In addition, the shorter cuttings were more vulnerable to deer browse that occasionally occurred when the fencing failed.

4.2. Genus-level performance differences

Our results provide clear support for the second hypothesis, revealing distinct performance patterns between *Populus* and *Salix* that have important implications for species selection in urban restoration projects. *Populus* planted as potted trees demonstrated superior overall performance, achieving 73% greater total growth than *Salix* (186.5 cm versus 107.8 cm) over the three-year establishment period. This growth advantage, combined with better cutting survival rates, positions hybrid *Populus* as the more robust choice for sites like Stillmeadow where there is intense competition from invasive plant species.

The stress response patterns revealed interesting genus-specific differences in how trees respond to different establishment conditions and light availability. *Populus* cuttings showed substantially lower stress levels than potted trees during initial establishment, suggesting that *Populus* may be better adapted to the direct planting approach when site conditions allow for successful establishment. However, this advantage was contingent on light availability—*Populus* exhibited strong positive relationships between canopy LAI and stress levels across all genotypes, indicating that individuals under heavier canopy cover experienced significantly greater physiological stress from light limitation. This sensitivity to available light provides a mechanistic explanation for the observed pattern that *Populus* cuttings with lower overall stress were disproportionately those that established in high-light microsites. In contrast, *Salix* showed no significant relationship between light availability and stress, indicating that *Salix* may be less suitable for direct planting not due to light sensitivity, but rather due to other establishment limitations. This fundamental difference in stress physiology—with *Populus* responding strongly to light gradients and *Salix* remaining relatively unaffected—suggests genus-level differences in shade tolerance mechanisms during establishment.

These physiological differences align with the ecological strategies of these genera. *Populus* spp. are typically more aggressive colonizers of disturbed sites and open habitats and may be better equipped to handle the variable conditions present in urban forest gaps (Reitschmiedová et al. 2022) particularly when sufficient light is available. The strong light-stress relationship observed in *Populus* spp. is consistent with their evolutionary ecology as early successional species that thrive in high-light gaps but experience physiological stress under canopy competition. *Salix* spp., while also pioneer species, often establish in riparian environments where water availability is less limiting than in upland forest restoration sites (Isebrands and Richardson, 2014), and their light-independent stress response suggests greater physiological flexibility across light gradients, though this flexibility did not translate to improved establishment success in this urban forest context.

4.3. Genotype selection and performance optimization

Phyto-recurrent selection (Zalesny et al. 2007) is a genotype selection technique that has been used in phytoremediation, agroforestry, and other phytotechnology applications globally (Pilipovic et al., 2019; Zalesny et al. 2019b; Zalesny and Pilipović 2022b) to select *Populus* and *Salix* genotypes that are best suited to a particular site. Here we set the stage for similar genetic selection techniques for forest restoration projects. We found substantial variation among genotypes within both genera strongly supporting our third hypothesis thereby providing actionable information for restoration practitioners. The comprehensive performance ranking revealed clear winners: `Tully Champion` among *Salix* genotypes and `DN34` and `NM2` among *Populus* genotypes emerged as superior performers across multiple traits. These genotypes demonstrate that careful selection can significantly improve restoration outcomes.

The trade-offs between growth and survival evident in several genotypes highlight the complexity of genotype selection. `SX61` achieved the highest growth rate among *Salix* (0.457 m) but suffered from poor survival (40%), while `Millbrook` demonstrated excellent survival (80%) with moderate growth. Similarly, `9732–11` showed exceptional growth among *Populus* (0.578 m) but poor survival (50%). These patterns suggest that restoration goals and site type should drive genotype selection, with fast-growing genotypes appropriate for sites where they need to outcompete fast growing invasive plant species and survival-focused genotypes better suited for poorer sites where invasive plants are less of a problem.

The temporal instability in genotype rankings between years emphasizes the importance of multi-year evaluation for restoration planning. Environmental conditions, competitive pressure, and tree maturation all influence performance trajectories, making single-year assessments insufficient for predicting long-term success. The identification of critical stress thresholds (0.97 for growth performance and 1.704 for zero growth) provides practitioners with early warning indicators to identify struggling plantings before complete failure occurs. Thus, providing an opportunity for intervention (i.e., weeding, watering) to preserve the planting.

4.4. Implications for urban forest restoration practice

These findings provide actionable guidance for practitioners incorporating hybrid *Populus* spp. and *Salix* genotypes. into urban forest restoration. Superior genotypes (*Tully Champion*, *DN34*, and *NM2*) planted as potted stock in high-light microsites—canopy gaps, forest edges, or areas with minimal overhead competition—offer the highest probability of success. The 59% three-year survival rate for potted trees represents acceptable establishment success for challenging urban environments, particularly given the dense planting strategy (1.2 m spacing) that maintains approximately 4000 trees per hectare even after mortality.

Light availability plays a critical role in establishment success through distinct genus-specific pathways. *Populus* genotypes experience physiological strain under shade and require high-light microsites to minimize stress and maximize survival, while *Salix* genotypes show light-independent stress responses. For growth, both genera benefit from high-light conditions, with individuals in higher light environments achieving 9–14 cm greater annual growth per unit decrease in canopy LAI. Initial microsite selection has both immediate survival consequences and long-term growth implications, as light effects on growth persist through year three.

Direct planting of cuttings should be reserved for large-scale applications where site conditions allow systematic maintenance, high-light microsites can be targeted, and some level of failure is acceptable. *Populus* genotype cuttings require the highest-light locations available due to their light-dependent survival during the critical first year, while *Salix* genotype cuttings may tolerate variable light conditions. However, poor overall survival rates (6.7% for *Salix* genotypes, 22.2% for *Populus* genotypes by year three) indicate that cuttings remain a high-risk approach regardless of genus.

Successfully established trees rapidly reach heights (274.6 cm mean for *Populus* genotype potted trees by year three) that reduce deer browsing risk and shift light competition in favor of planted trees over invasive understory species. Strategic selection of naturally open microsites or thinning of competing vegetation will accelerate this trajectory toward canopy dominance, addressing the primary challenge of achieving rapid canopy closure while minimizing long-term maintenance requirements.

5. Conclusions

Our study demonstrates that hybrid *Populus* and *Salix* can play valuable roles in urban forest restoration, particularly when planted as potted stock and when superior genotypes are selected. The clear performance differences between genera, genotypes, and planting methods provide restoration practitioners with specific guidance for optimizing establishment success. While cutting establishment rates prove too low for most restoration applications, the rapid growth potential of successfully established trees offers significant advantages for achieving restoration goals in challenging urban environments.

The identification of superior genotypes and the development of performance prediction tools based on stress thresholds advance the scientific foundation for evidence-based urban forest restoration. These results support expanding the use of hybrid *Populus* and *Salix* beyond agricultural applications to address the pressing need for innovative approaches to urban forest sustainability in the face of ongoing environmental challenges.

Funding Sources

This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

Many colleagues and partners helped make this project possible. We thank the staff of Stillmeadow Community Fellowship, the Urban Silviculture Workforce Trainees, the Canopy Crew, Gabe LePage, Brandon Wilson Radcliffe, Morgan Grove, Mckay Jenkins, Ryan Vinhal, Elizabeth Rogers, the USDA Forest Service Urban & Community Forestry Program, and countless volunteers who showed up to clear, prepare, and plant the experimental plots.

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