Identifying restoration practices and landscape variables that increase native plant establishment and mitigate plant invasion

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Table of Contents

List of Figures

List of Tables

Objective 1: Compare the vegetation community of restored with un-restored stream reaches.

Introduction

In environmental restoration programs, trade-offs between competing objectives are common. Many restoration projects are undertaken to increase ecological function in one location to compensate for degradation in another. Unfortunately, prioritizing mandated objectives can sometimes lead to achieving improvement in one area at the expense of another (Noe et al. 2024). Freshwater lotic systems are particularly sensitive to degradation, but globally, freshwater restoration programs are relatively rare and are often assumed to be included in terrestrial restoration initiatives (Cooke et al. 2022, Mohan et al. 2022). Within the United States, an estimated 22-24% of riparian systems are degraded due to urban land use, riparian vegetation disturbances, and decreased streambed sediment size and stability (Kaufmann et al. 2022). In addition to threats from urbanization and pollutants (Johnson et al. 2020), riparian ecosystems are at increased susceptibility to invasive plants due to greater landscape connectivity, frequent disturbance, and movement of plant propagules by flood waters (Schirmel et al. 2016). Globally, invasive species have an estimated economic cost of $423 billion annually and have been implicated in 60% of extinctions (Roy et al. 2024). Specifically in riparian systems, terrestrial invasive plants have been shown to decrease soil stability (Colleran et al. 2020), alter organic matter inputs (Fargen et al. 2015), and decrease density (McNeish et al. 2015) and survival (Custer et al. 2017) of aquatic macroinvertebrate communities.

While most stream restoration projects across the U.S. focus on mitigation and include actions ranging from large dam removal to increase fish habitat to streambank stabilization to protect infrastructure, the focus of stream restoration in the Chesapeake Bay watershed is often on re-establishing stable channel morphology to reduce sediment and nutrient inputs, with revegetation either as a secondary goal or an assumed by-product (Palmer et al. 2010, Fraley-McNeal et al. 2022). However, riparian vegetation performs critical services of in-stream processing of pollutants and organic matter (Sweeney et al. 2004), bank stabilization (Hubble et al. 2010), and supporting biodiversity (Naiman et al. 1993). Despite these clear benefits, evaluation of revegetation status following stream restoration is limited, often only assessed when required by permitting, typically only funded for 3 years following the restoration, and often focuses on basic metrics such as woody stem survival (USACE 2024) and invasive species coverage limits (i.e., less than 10% invasive species coverage; (USACE 2025). Even the main goals of improving ecosystem structure and function are rarely empirically evaluated to determine restoration success (Bernhardt and Palmer 2011). Thus, given the urgent need for widespread stream restoration globally there remain important gaps in our understanding of how restoration impacts riparian vegetation, which would ideally be improved (i.e., more native species/cover) alongside stream (geo)morphology.

The Chesapeake Bay Program is one of the most intensive restoration programs globally, bringing together partners in the United States Environmental Protection Agency (EPA), the Chesapeake Bay Commission, the six states within the watershed, and the District of Columbia (NFWF 2012). Restoration goals within the Chesapeake Bay watershed largely focus on reducing sediment, nitrogen, and phosphorus, to comply with total maximum daily load targets set by the EPA (USEPA 2010). Throughout the Chesapeake Bay watershed, 430 km of stream were restored as of 2024 (Noe et al. 2024), approximately 0.3% of the over 160,934 km of streams within the watershed (Chesapeake Bay Commission 2020). However, the Chesapeake Bay Program also set forth additional goals related to riparian vegetation to be completed by 2025. Of the 31 goals, 18 were not achieved, which raises concern for riparian revegetation efforts (Barnhart et al. 2024). Forest buffer goals have not been met since 2002 and the goal of expanding urban tree canopy by 971 ha was counteracted by a loss of 10,117 ha of tree canopy across the Chesapeake Bay watershed between 2014 and 2025 (Barnhart et al. 2024).

To better understand the consequences of stream restoration for invasion, especially given the documented negative impacts of invasive plants on riparian ecosystems, we compared 46 stream restoration projects in the Chesapeake Bay watershed in Maryland to paired unrestored reaches. While invasive plants are species that have known negative impacts, we chose to focus on non-native plants (~93% of those we observed are also considered invasive) because in urban areas, where many stream restoration projects occur in the Chesapeake Bay watershed, many non-native plants are used in landscaping and while they may not be considered invasive, they have been shown to have negative impacts, particularly on invertebrate communities and species dependent on invertebrates (Narango et al. 2018, Tallamy et al. 2021).

We outlined four possible revegetation outcomes from stream restoration when compared to unrestored reaches (Figure 1). The best-case, “good” scenario for revegetation would be reduced relative richness and cover of non-native species and greater relative richness and cover of native species (green upper right quadrant of Figure 1). Conversely, the “worst”-case scenario would be greater relative richness and cover of non-native species and reduced richness and cover of native species (red lower left quadrant of Figure 1). “Bad” outcomes include restoration projects where relative richness and cover of non-native and native species were either both reduced or both greater on the restored reach (upper left and lower right yellow quadrants of Figure 1). Calculations for relative richness and cover of non-native species (non-native species index; NNSI), invasive species (invasive species index; ISI), and native species (native species index; NSI) return values between 0 and 1, 0 being either no richness and coverage of non-native, invasive, or native species, respectively, and 1 being complete richness and coverage of non-native, invasive, or native species. To aid in interpretation, such that negative values are bad and positive values are good, NSI was multiplied by -1.

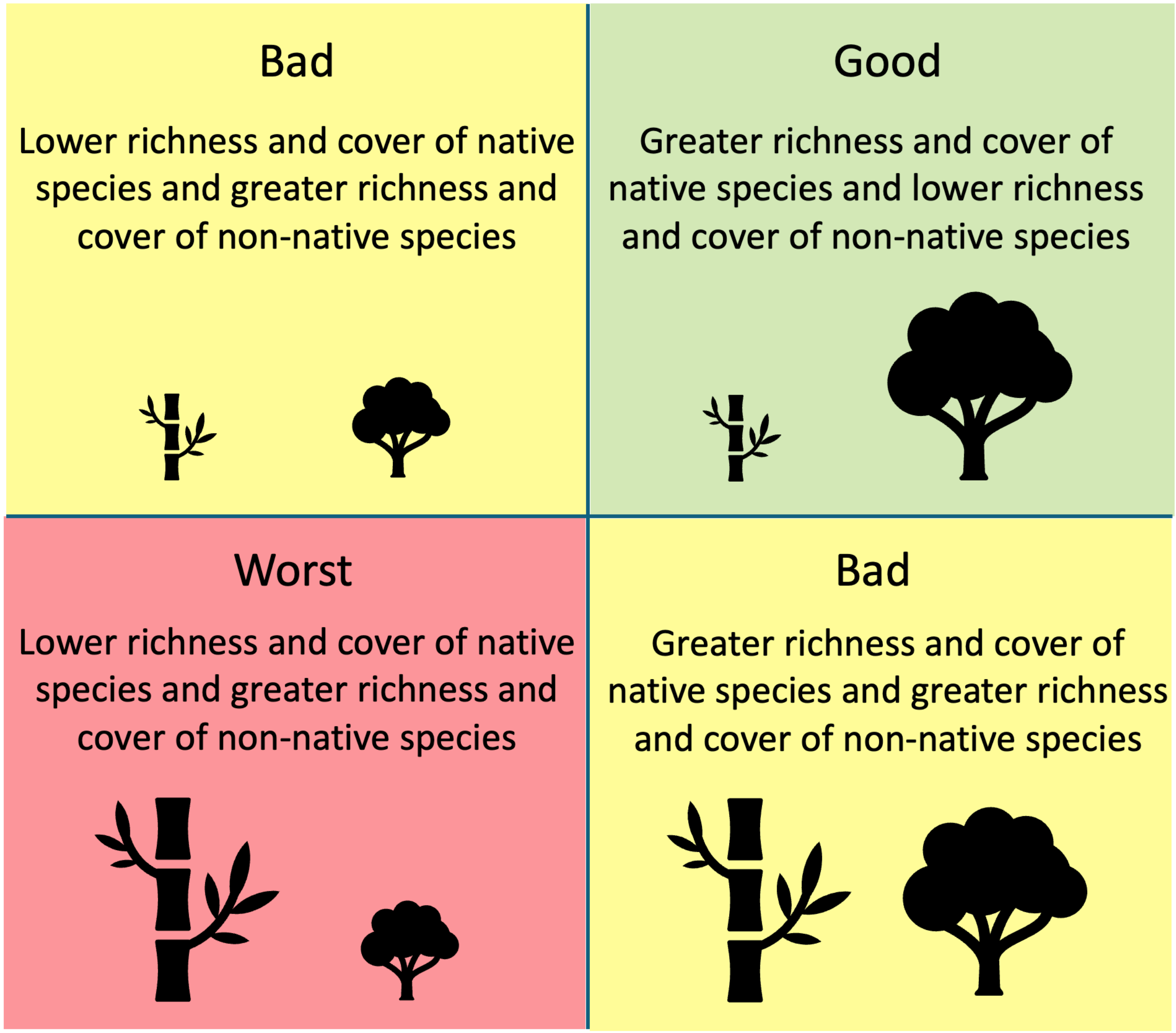


Figure 1. Theoretical stream restoration revegetation outcomes. All comparisons of restoration are in relation to paired unrestored stream reaches. The bamboo icon represents non-native species, and the tree icon represents native species.

Methods

Study Site

Forty-six stream restoration projects were selected for sampling. Various environmental consulting firms were involved in the design, implementation, and monitoring of the selected projects. Several agencies, including the Maryland Department of Natural Resources, United States Fish and Wildlife Service, and Maryland State Highway Authority were involved in the projects either as clients or through permitting. The main stem construction length of the remaining projects ranged from 162-1372 m in Anne Arundel, Baltimore, Calvert, Frederick, Harford, Howard, Montgomery, and Prince George's Counties. Most projects were in residential areas (n=35), but sites are also located in public parks (n=4), rights-of-way (n=2), commercial land (n=2; e.g., shopping center), institutional land (n=2; e.g., community college), and pasture (n=1; Figure 2).

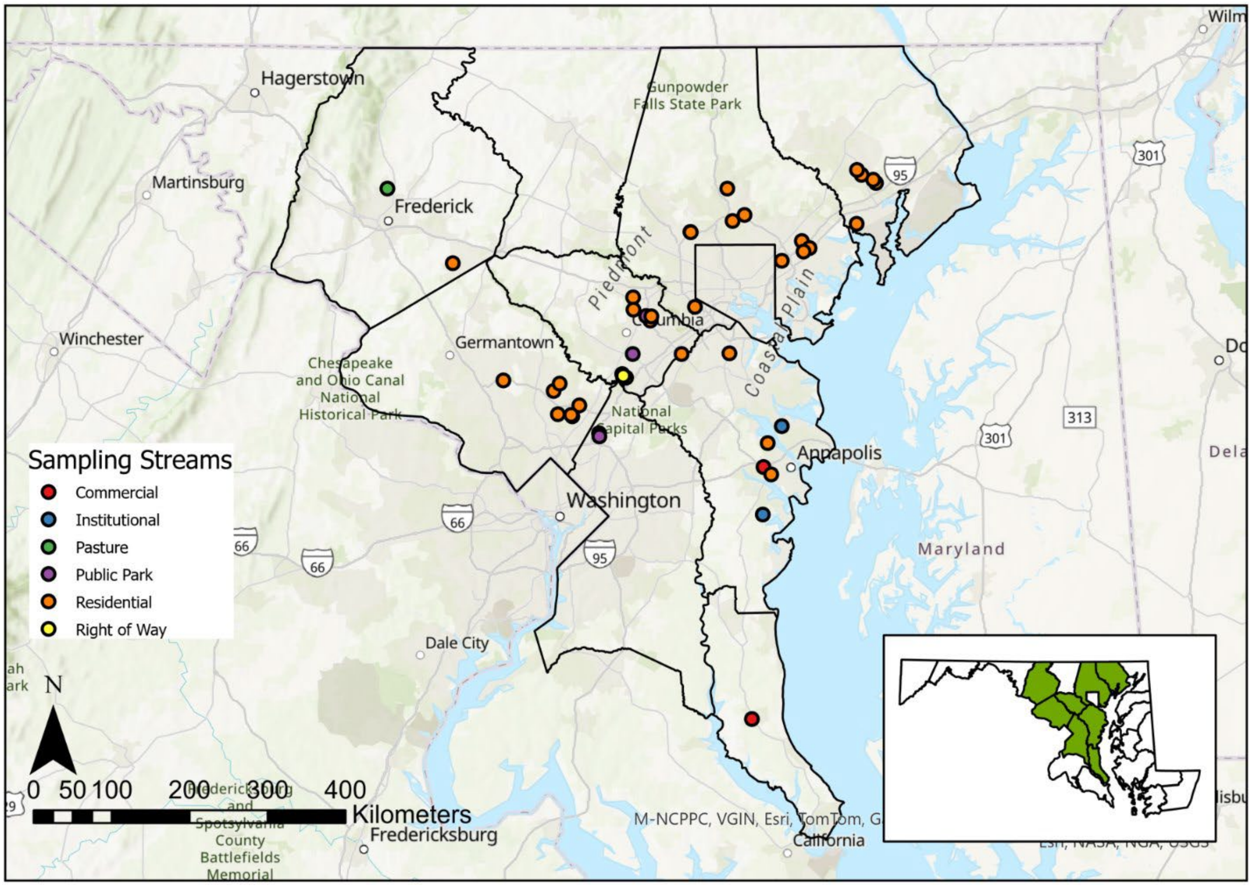


Figure 2. Stream restoration projects sampled within Maryland's Chesapeake Bay watershed (n=46). Site locations are color-coded by their surrounding landscape context. Unrestored stream reaches have the same surrounding landscape context as their paired restored stream.

For each stream restoration project, a comparable unrestored stream reach was selected as an approximation of the unrestored condition of the vegetation community. Ideally, streams to be restored would have been sampled prior to restoration to directly assess the impacts of restoration on vegetation community outcomes. However, because the restoration projects were completed well before the present study's initiation, we were precluded from assessing pre-restoration conditions. Therefore, the selected unrestored reaches serve as proxies for pre-restoration conditions. Most (n = 30 of 46) unrestored reaches were immediately upstream or downstream of the restored reach of the same stream (minimum ~87 m away). However, for those streams where the reach upstream or downstream was not analogous to the restored reach (e.g., bounded by concrete, tributary merges with mainstem), a nearby stream was used for comparison (n = 2 of 46). Remaining unrestored reaches were within the same watershed as their restored reach and were most often tributaries off the same mainstem. In selecting nearby streams, surrounding context (e.g., public park, residential area), stream size, historical vegetation coverage, and forest age were taken into account. Additionally, an even mix of upstream and downstream unrestored reaches were selected to control for potential biases due to flow direction.

Vegetation Sampling

We sampled vegetation from late May through July of 2023 and 2024 during peak summer productivity. Sampling methods captured total species composition, basal area of woody species, and canopy metrics. A 100-m sampling reach was established along the center of each stream reach (n = 92). Within the 100-m sampling reach (Dybkjaer et al. 2012, Gomes et al. 2020), 6 sampling points were established starting at the beginning (0 m), middle (45 m), and end (90 m) of the reach, such that the first point began at 0, 45, or 90 m and the second point began 5 m after the first point on the opposite bank (Figure 3). At each sampling point, two 5-m point intercept transects, one 1 m from the bank edge and one 5 m from the bank edge (n = 12 transects/sampling reach), were sampled to determine species composition and to capture variation in composition moving away from the streambank. Measurements along the transects were taken every 20 cm and species were identified from ground level through the canopy. Transects were always sampled in an upstream to downstream direction. Fixed-radius plots of 0.02 ha were used to measure overstory vegetation, with point center in the center of the two transects (i.e., 3 m from the bank and 2.5 m from either end of the transect) at each of the 6 sampling points (Figure 3; Avery and Newton 1965, Fajvan 2005). We measured diameter at breast height (DBH; 1.37 m) of trees with DBH greater than 11.4 cm and identified them to species to estimate overstory basal area, excluding shrubs or saplings (Waskiewicz et al. 2015). At the same point center used for the fixed-radius plots, canopy metrics including canopy cover, photosynthetically active radiation, and leaf area index were taken using a CI-110 Plant Canopy Imager (CID BioScience, Camas, WA, USA). Nativity of each species was determined using the United States Department of Agriculture Plant List of Attributes, Names, Taxonomy, and Symbols (PLANTS) Database (USDA 2025). Invasive species were determined using the United States Register of Introduced and Invasive Species (US-RIIS; Simpson et al. 2022).

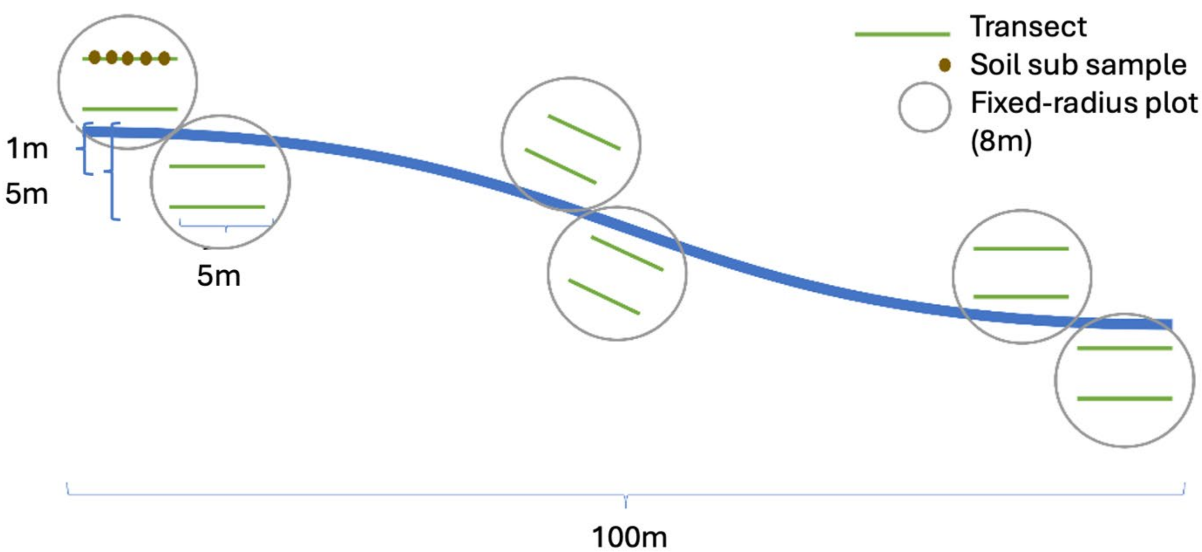


Figure 3. The vegetation and soil sampling design for the 46 paired stream restoration projects. There are six sampling points along each 100 m-sampling reach, beginning at 0 m, 5 m, 45 m, 50 m, 90 m and 95 m. At each sampling point, there were 2 5-m point intercept transects, one 1 m from the streambank and one 5 m from the streambank. Five soil samples were taken along

each transect 5 m from the streambank. At the mid-point between the two transects, a fixed-radius plot of 0.02 ha (8 m radius) was used to measure overstory basal area (DBH > 11.4 cm). Streams were always sampled in an upstream to downstream direction. Figure not to scale.

Soil Sampling

To evaluate the relationship between soil chemistry and fertility and riparian vegetation, we sampled and analyzed soil from each transect 5 m from the streambank. Along each of the six transects per stream that were 5 m from the streambank, five soil samples were collected using a soil probe at a depth of approximately 10-13 cm. The five samples were aggregated per transect resulting in six composite soil samples per stream reach. From the six soil samples taken per stream, four samples were randomly selected for soil analysis through the Virginia Tech Soil Testing Lab. Information provided by the routine soil test includes soil pH, P (using a Melich-1 solution), K, Ca, Mg, Zn, Mn, Cu, Fe, B, and an estimate of cation exchange capacity. Nitrogen availability to plants was not measured because it transforms readily and is thus temporally dynamic.

Analysis

To assess native, non-native, and invasive species prevalence in streamside vegetation communities, we utilized species indices as a means to scale the coverage and species richness of species groups by total cover and species richness. These calculations return values between 0 and 1, 1 being greatest prevalence and 0 being no prevalence. These indices were calculated for native and non-native species using USDA PLANTS data and for invasive species using US-RIIS data. Native species index (NSI), non-native species index (NNSI), and invasive species index (ISI) were calculated as:

$$NSI = \left(\frac{\left(\frac{native\ species\ richness}{overall\ species\ richness}\right) + \left(\frac{native\ species\ cover}{total\ cover}\right)}{2} \right) * -1$$

$$NNSI = \frac{\left(\frac{nonnative\ species\ richness}{overall\ species\ richness}\right) + \left(\frac{nonnative\ species\ cover}{total\ cover}\right)}{2}$$

$$ISI = \frac{\left(\frac{invasive\ species\ richness}{overall\ species\ richness}\right) + \left(\frac{invasive\ species\ cover}{total\ cover}\right)}{2}$$

To leverage the paired design of the study, shifts from the unrestored state to the restored state were determined using differences in the index values. To calculate the difference, using NNSI as an example, the NNSI on the restored (R) reach is subtracted from the NNSI on the unrestored (U) reach within that pair (i.e., the approximate starting condition). If the resulting $NNSI_{U\text{-}R}$ value is positive, there is a greater NNSI (i.e., relative cover and richness of non-native species) on the unrestored reach in the pair and if the resulting $NNSI_{U\text{-}R}$ value is negative, there is a greater NNSI on the restored reach in the pair. Negative values correspond to greater $NNSI_{U\text{-}R}$ and $ISI_{U\text{-}R}$ on restored reaches; however, because negative numbers are associated with negative outcomes, we have chosen to multiply NSI by -1, such that negative NSI_{U-R} values correspond to negative outcomes (i.e., relative richness and cover of native species greater on the unrestored stream in the pair).

On a subset of the data, we analyzed the relationship between the difference in NNSI ($NNSI_{U-R}$), and field assessment scores of the restoration project success from a geomorphic perspective. Geomorphologic restoration success scores (Thompson et al. 2021) were only available for 36 of the 46 restoration projects. We used a linear model, function *lm* in Program R (R Core Team 2025), to model the relationship between the difference in NNSI ($NNSI_{U-R}$) and geomorphologic restoration success. We also assessed relationships between geomorphologic restoration success and raw (i.e., without accounting for unrestored condition) NNSI and ISI using linear models. While a visual assessment of riparian vegetation was included in the field assessment score, the relationship between the scores and NNSI did not change when excluding the vegetation metric.

We used a canonical correspondence analysis (CCA), function *cca* in R package *vegan* (Oksanen at el. 2025), to determine the impact of environmental predictors on community composition for each treatment and shifts within each pair. Environmental predictors included non-native species importance, basal area, measures of light availability (e.g., canopy coverage, photosynthetically active radiation), soil pH, and soil nutrient concentrations in parts per million (e.g., P, K, Mn, Mg, etc.).

Additionally, to more directly determine linear relationships among resource availability, time since restoration, and NNSI, we used general linear models, function *lm* in R. We analyzed fixed effects of phosphorus (in parts per million; ppm), potassium (ppm), photosynthetically active radiation, and time since restoration separately and included a random effect of pair as both restored and unrestored reaches were included in the analyses. Because unrestored reaches did not have a time since restoration, unrestored reaches were assigned the same time since restoration as their paired restored reaches. All significance was determined using $\alpha = 0.05$.

Results

Stream restoration worsens plant invasion in most cases

Overall, stream restoration often resulted in worse vegetation outcomes (bad or worst) compared to the paired unrestored, degraded stream reaches. 37% of projects achieved “good” restoration outcomes (Figure 4), leaving 63% of projects falling short (“bad” and “worst”) of desired revegetation outcomes. Of the 46 paired reaches, 27 (59%) had greater NNSI on the restored reach than the paired unrestored reach (sites with negative NNSI $_{U-R}$ values in Figure 4). Further, half (n = 23 of 46) of all projects had the worst revegetation outcomes such that there was greater NNSI and reduced NSI.

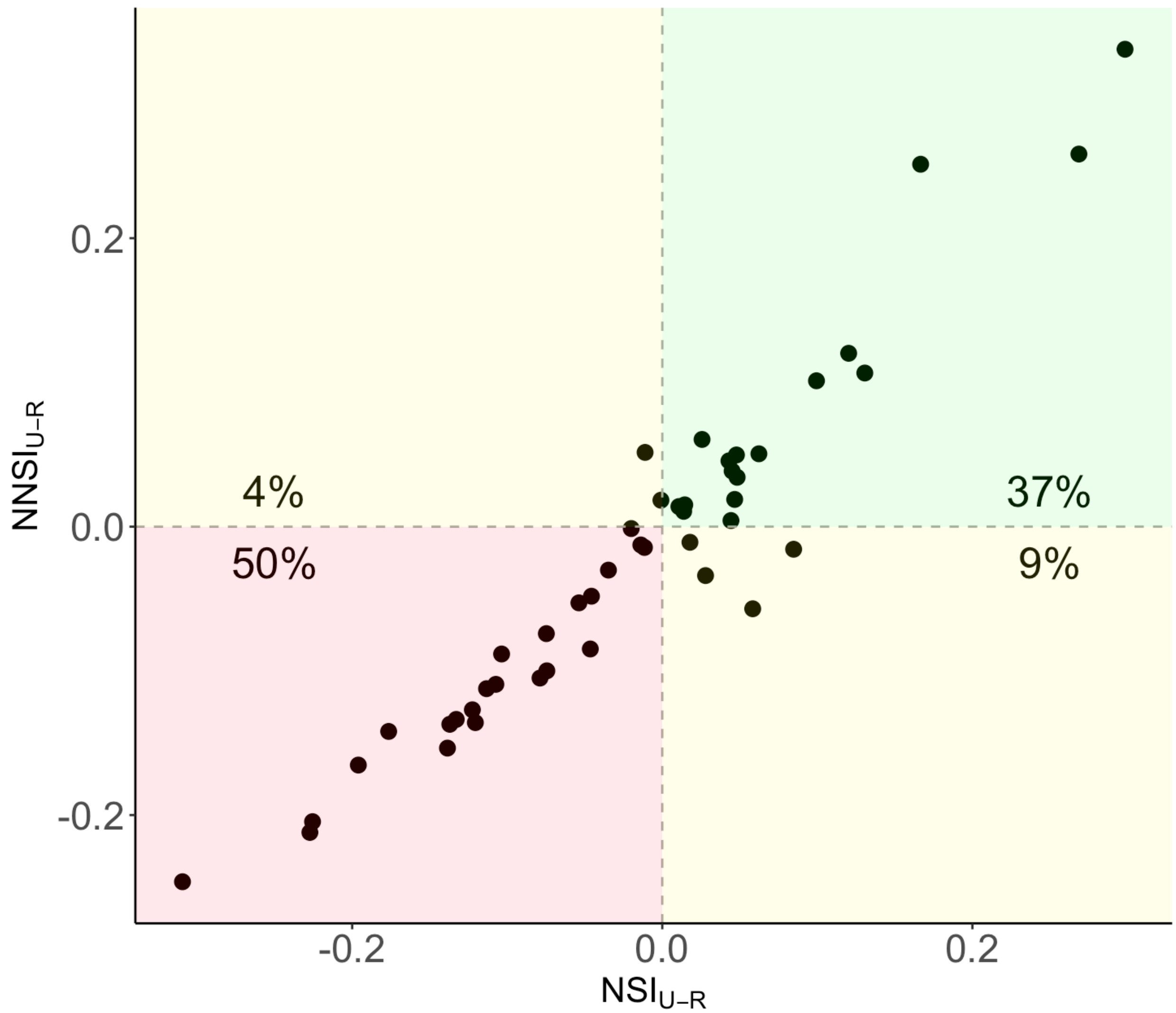


Figure 4. Restoration project outcomes in terms of relative richness and cover of native species (NSI) and non-native species (NNSI). The difference in NSI and NNSI values between paired unrestored and restored reaches is utilized to reduce noise in the data and account for the paired study design (notes as NSI_{U-R} and $NNSI_{U-R}$). While NSI and ISI were calculated using the same equation, NSI values were multiplied by -1 such that negative NSI_{U-R} values reflect negative outcomes (i.e., a negative NSI_{U-R} reflects pairs where NSI was greater on the unrestored site in the pair). Sites with greater NSI_{U-R} and lower $NNSI_{U-R}$ are the most desirable restoration outcomes, highlighted in green in the top right quadrant. Sites with lower NSI_{U-R} and greater $NNSI_{U-R}$ are the worst restoration outcomes, highlighted in red in the bottom left quadrant. Sites in yellow quadrants (top left and bottom right) are sites where NSI_{U-R} and $NNSI_{U-R}$ were either both greater or both lower on the restored reach compared to the paired unrestored reach. Percentages reflect the percent of sites that fall within each quadrant.

Non-native revegetation outcomes are unrelated to geomorphologic outcomes

Because improved channel physical function is often a goal of stream restoration, we assessed the relationship between how well a stream was restored in terms of physical function and revegetation success. A total of 36 of 46 sites had been previously evaluated for geomorphologic restoration success, including bedform location, substrate, cover, bank stability, riparian vegetation, and floodplain attributes (Thompson et al. 2021; Table 1). While riparian vegetation was included in the success score, it was visually assessed during March 2019 and January 2020 and removing the riparian vegetation metric from the overall score had no bearing on the score's relationship with the NNSI. We found no relationship between geomorphologic restoration success and the difference in relative non-native species richness and cover between paired unrestored and restored reaches ($NNSI_{U\text{-}R}$) ($F_{1,34}$ = 0.0047, R^2 = -0.029, p = 0.95; Figure 5). Over half (53%) of the assessed sites had greater richness and cover of non-native species on the restored reach compared to the unrestored reach. The lack of relationship illustrates stream physical function can be restored independently of the vegetation community.

Table 1. The scoring system used to determine geomorphological restoration success for 36 of the stream restoration projects. Sampling was conducted March 2019 and January 2020 (Thompson et al. 2021). Riparian vegetation cover was assessed visually and not systematically sampled.

	1	2	3	4
Bedform location in stream (1-4)	<25% of bed features in proper geomorphologic locations along reach	25-50% of bed features in proper geomorphologic locations along reach	50-75% of bed features in proper geomorphologic locations along reach	>75% of bed features (riffle, run, pool) in proper geomorphologic locations along reach
Substrate	Significant embedded areas, poor gradation, loose, soft areas prominent in >75% of bed	Well graded particle size distribution, minimal embeddedness in 25-50% of bed	Well graded particle size distribution, minimal embeddedness in 50-75% of bed	Well graded particle size distribution, minimal embeddedness in >75% of bed
Cover/refuge	Presence of refuge areas of few types in <25% of reach	Presence of refuge areas of diverse types in 25-50% of reach	Presence of refuge areas of diverse types in 50-75% of reach	Presence of plentiful refuge areas of diverse types (undercut banks, root mats, oxbows/backwaters) in >75% of reach
Bank stability	High bank slopes, good vegetation cover and no evidence of mass wasting in <25% of reach	Low bank slopes, good vegetation cover and no evidence of mass wasting in 25-50% of reach	Low bank slopes, good vegetation cover and no evidence of mass wasting in 50-75% of reach	Low bank slopes, good vegetation cover and no evidence of mass wasting in >75% of reach
Riparian veg cover	<25% of riparian area covered in native vegetation	25-50% of riparian area covered in native vegetation	50-75% of riparian area covered in native vegetation	>75% of riparian area covered in native vegetation
Floodplain	Evidence of flow access, sediment deposition along <25% of reach	Evidence of flow access, sediment deposition along 25-50% of reach	Evidence of flow access, sediment deposition along 50-75% of reach	Evidence of flow access, sediment deposition along >75% of reach

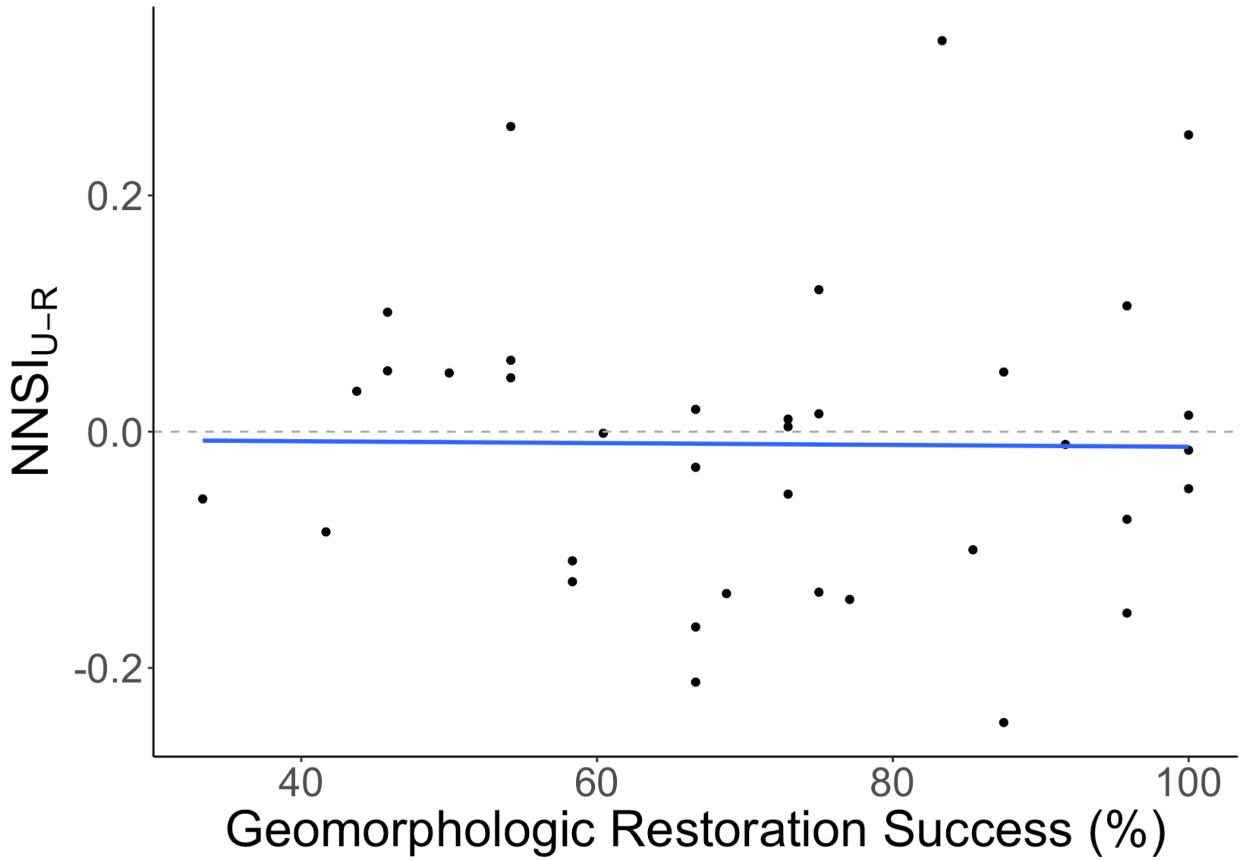


Figure 5. The relationship between the difference in non-native species index (NNSI; unrestored NNSI – restored NNSI; $NNSI_{U-R}$) and geomorphologic restoration success scores of stream restoration physical function (n = 36). Negative $NNSI_{U-R}$ values signify that NNSI was greater on the restored stream in the pair and positive $NNSI_{U-R}$ values signify that NNSI was greater on the unrestored stream in the pair. Geomorphologic restoration success scores were measured as the percent of the original stream restoration design plans that are still intact and functioning. There is no relationship between $NNSI_{U-R}$ and geomorphologic restoration success ($F_{1,34} = 0.0047$, $R^2 = -0.029$, $p = 0.95$).

Though revegetation outcomes relative to paired unrestored conditions were not related to geomorphologic outcomes, we also tested the relationship between geomorphology and raw invasion metrics (i.e., not accounting for unrestored NNSI or ISI). We found a marginally significant relationship between geomorphologic restoration success and relative richness and cover of invasive species (ISI; $F_{1,34}=3.38$, $R^2 = 0.064$, $p = 0.075$; Fig. 3). Therefore, invasion outcomes are marginally related to geomorphology, but there remains much unexplained variation.

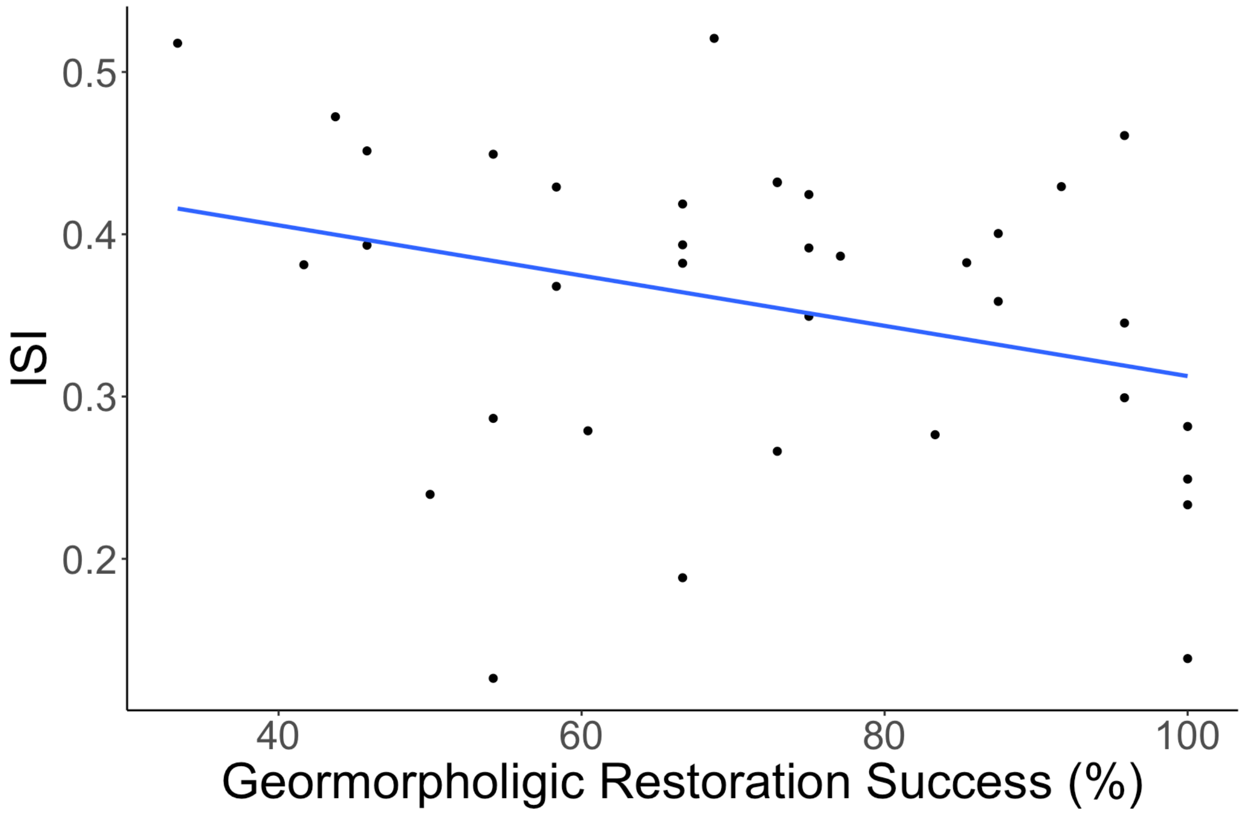


Figure 6. The relationship between the invasive species index (ISI) and geomorphologic restoration success scores of stream restoration physical function (n = 36). Geomorphologic restoration success scores were measured as the percent of the original stream restoration design plans that are still intact and functioning. The relationship between ISI and geomorphologic restoration success is marginally significant ($F_{1,34}$ =3.38, R^2 = 0.064, p = 0.075).

Neither resource availability nor time since restoration explain revegetation outcomes

Because of well established relationships between resource availability and invasion (Davis et al. 2000), we hypothesized that dominance of non-native plants would increase with greater soil nutrient and light availabilities, both of which increase due to construction. While NNSI was positively related to soil phosphorus (ppm; β = 0.0059, 95% CI: 0.0033–0.0085, $F_{1,89}$ = 20.19, R^2 = 0.18, p < 0.001), soil potassium (ppm; β = 0.0013, 95% CI: 0.00070–0.0019, $F_{1,90}$ = 19.12, R^2 = 0.17, p < 0.001), and photosynthetically active radiation (PAR; β = 0.00011, 95% CI: 0.000011–0.00021, $F_{1,90}$ = 4.88, R^2 = 0.041, p = 0.030), there were no differences by treatment for any of these resources (Figure 7A-C).

Additionally, we expected time since restoration to influence non-native species such that more recent restoration projects would have greater non-native species indices due to a pulse in resource availability following a massive disturbance. Because there is no "time since restoration" for unrestored sites, we assigned the same time since restoration as the restored reach to the unrestored reach within each pair, expecting no relationship between the time since restoration and NNSI of unrestored reaches. There was no relationship between relative richness and cover of non-native species (NNSI) and time since restoration ($F_{1,90}$ = 0.16, R^2 = -0.0093, p = 0.69; Figure 7D), such that time since restoration was not related to how restoration projects fared in terms of the non-native species index.

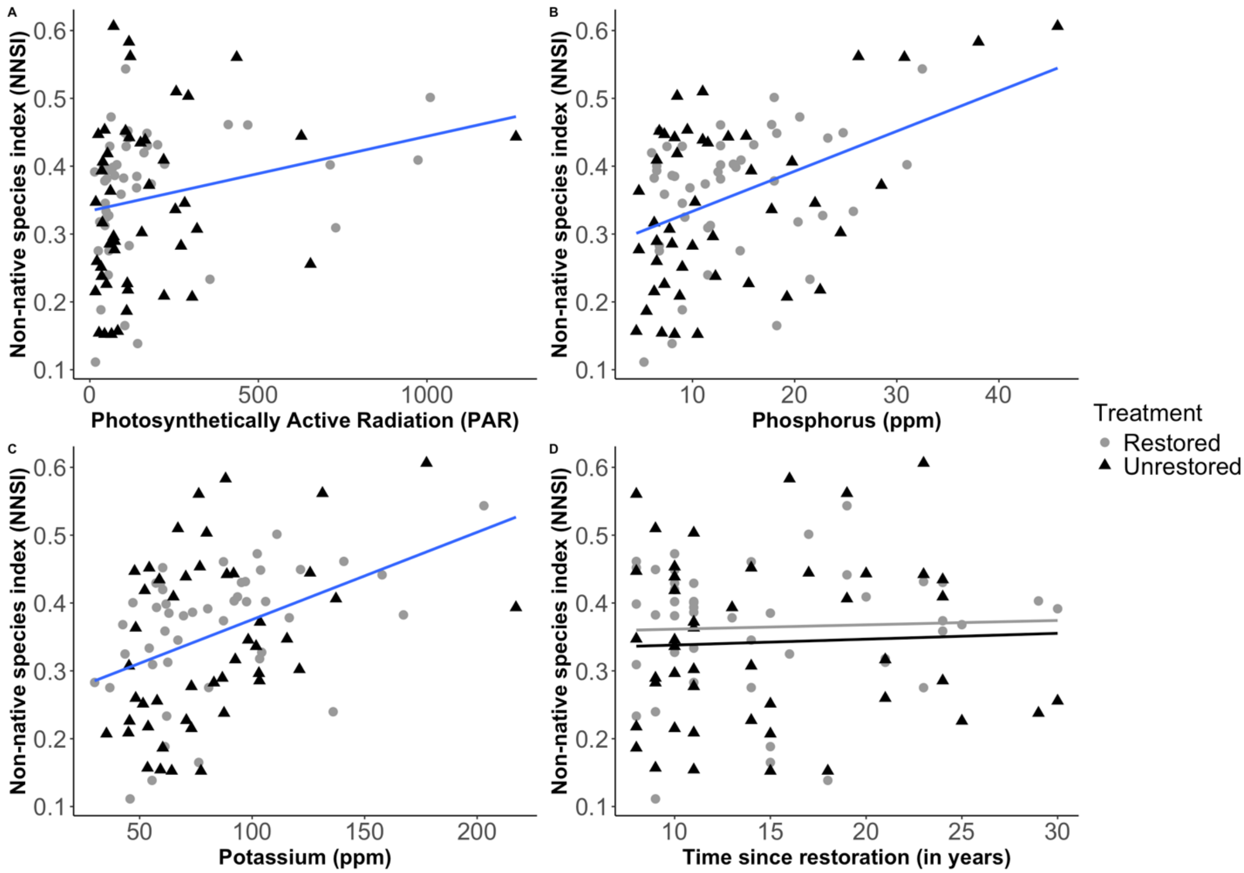


Figure 7. The relationships between A) photosynthetically active radiation, B) phosphorus, C) potassium, and D) time since restoration and the non-native species index (NNSI), a measure of the relative richness and cover of non-native species. NNSI was positively correlated with photosynthetically active radiation ($F_{1,90} = 4.88$, $R^2 = 0.041$, $p = 0.030$), phosphorus ($F_{1,89} = 20.19$, $R^2 = 0.18$, $p < 0.001$), and potassium ($F_{1,90} = 19.12$, $R^2 = 0.17$, $p < 0.001$), but not time since restoration ($F_{1,90} = 0.16$, $R^2 = -0.0093$, $p = 0.69$).

Discussion

We found that stream restoration has the perverse outcome of worsening plant invasion, eve when geomorphologic goals are met. Though stream restoration projects in the Chesapeake Bay watershed as generally undertaken to improve water quality or as mitigation (Noe et al. 2024), there are serious implications of the failure to promote native riparian vegetation. More concerning is that restoration exacerbates an already major problem of invasion along highly degraded urban streams (Johnson et al. 2020). Projects may be achieving their goals in other areas, such as channel physical function, but they are largely failing at limiting non-native plant invasion.

The increase in invasive plants along restored stream reaches compared to unrestored reaches has negative consequences for urban biodiversity and human health. The most common invasive species by coverage was *Microstegium vimineum*, which can have negative effects on the performance of two species commonly planted on restored sites, *Acer rubrum* and *Liquidambar styraciflua* (Goldsmith et al. 2023), and can decrease *Anaxyrus americanus* survival (DeVore and Maerz 2014). There is evidence that prevalence of non-native species has resulted in insect declines (Tallamy et al. 2021), the effects of which are felt by higher trophic levels (Narango et al. 2018). Common forest birds also suffer higher rates of nest predation in non-native shrubs, especially in urban environments (Borgmann and Rodewald 2004). Of concern for human health, several species observed during our plant surveys are associated with increased disease risk, such as *Berberis thunbergii* and non-native *Lonicera* spp., which have been found to increase incidence of Lyme disease and the emerging pathogen ehrlichiosis (Allan et al. 2010, Mack and Smith 2011). These impacts may be particularly acute in urban systems where human contact is higher.

Non-native species dominance could be ephemeral in stream restoration projects, coincident with pulses in soil nutrient and light availability that come with earth and canopy disturbance. However, revegetation outcomes did not change with time since restoration (8-30 years). Additionally, invasive species have been shown to alter successional trajectories, especially when a disturbance increases resources availability and reduces competition (D'Antonio and Meyerson 2002, Theoharides and Dukes 2007). The lack of relationship between time since restoration and revegetation outcomes indicates that post-restoration invasion is a persistent feature of many restored streams. It also highlights a timing mismatch between invasion ecology (decades) and current monitoring requirements (three years).

We observed a tradeoff between native and non-native species richness and cover, such that increasing richness and cover of one comes at the detriment of the other. Therefore, controlling non-native species and managing for native species is a necessary step in improving revegetation outcomes. From our results, the typically required 3–5-year monitoring period is insufficient to prevent long-term invasion, especially as there is no longer funding to address invasion beyond the monitoring period. Given that half of the restored reaches exhibited the worst revegetation outcomes, our results highlight a major shortcoming in post-restoration project management.

The few regulations regarding standards of revegetation are insufficient to prevent invasion. These regulations include mandated native woody stem counts (USACE 2024) and invasive species cover limits (USACE 2025), which were not met by any of the 46 restoration projects (i.e., none of the restored reaches had less than 10% invasive species coverage). However, even though woody stems are the focus of revegetation efforts within our study system, included on the planting lists of all projects on our dataset, only 54% of projects had greater native woody species relative richness and cover on restored reaches compared to unrestored pairs (Figure 8). Given woody stem survival and nativity of vegetation are two of few vegetation requirements from permitting agencies within a 3–5-year monitoring period, current monitoring regulations are insufficient to prevent subsequent invasion. While revegetation outcomes improve when considering only woody species, diversity of growth forms in stream revegetation is important to support biodiverse urban ecosystems (Mata et al. 2021).

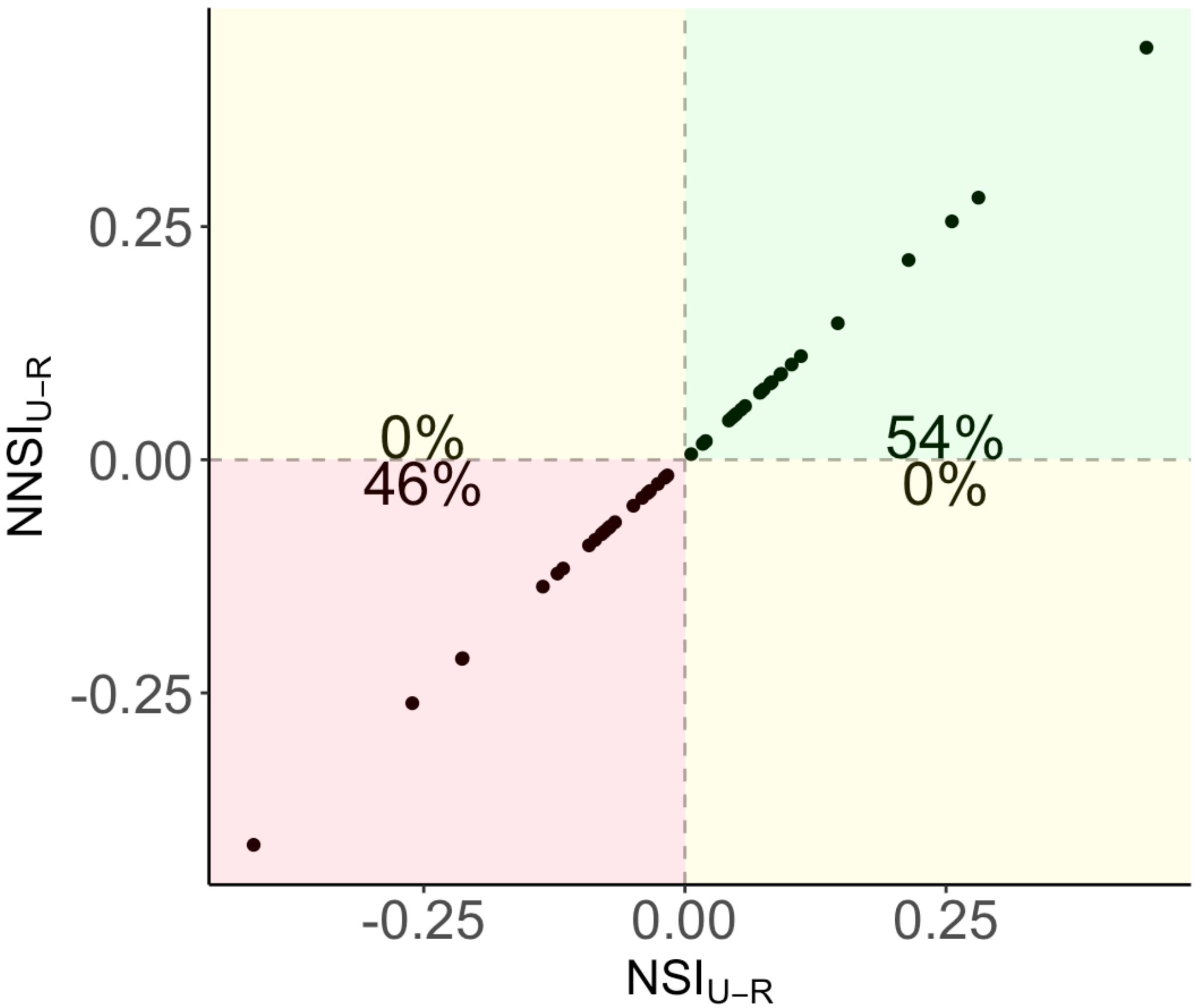


Figure 8. Restoration project outcomes in terms of relative richness and cover of native species (NSI) and non-native species (NNSI) for only woody growth forms. The difference in NSI and NNSI values between paired unrestored and restored reaches is utilized to reduce noise in the data and account for the paired study design (noted as NSI$_{U-R}$ and NNSI$_{U-R}$). Sites with greater NSI$_{U-R}$ and lower NNSI$_{U-R}$ are the most desirable restoration outcomes, highlighted in green in the top right box. Sites with lower NSI$_{U-R}$ and greater NNSI$_{U-R}$ are the worst restoration outcomes, highlighted in red in the bottom left box. Yellow boxes (top left and bottom right) are sites where NSI$_{U-R}$ and NNSI$_{U-R}$ were either both greater or both lower on the restored reach compared to the paired unrestored reach. Percentages reflect the percent of sites that fall within each quadrant.

Improved geomorphologic restoration outcomes were unrelated to revegetation success emphasizing that successful restoration of one attribute does not translate to the health of the entire riparian system (Noe et al. 2024). We expected a positive relationship between geomorphologic restoration success and revegetation success because deviations from historical hydrology may favor non-native species over native species competing in environmental conditions outside of their historical ranges (Catford et al. 2014, Flanagan et al. 2015, O'Briain et al. 2023). However, we found no correlation between limitation of non-native plants and geomorphology and thus, the primary goal of many stream restoration projects is often achieved without also improving the vegetation community.

Because variation in invasion outcomes was not related to geomorphology, other variables such as landscape context might account for between-site variation. Both suitable microsites and availability of propagules are necessary for invasions (Hobbs and Huenneke 1992, Chytrý et al. 2008, Menuz and Kettenring 2013), and, given all sampled stream restoration projects are located in the urban and suburban areas surrounding Washington D.C., Annapolis, and Baltimore, there is likely no shortage of non-native plant propagules (Warren et al. 2015). Non-native species are often associated with urban land use (Loewenstein and Loewenstein 2005, Atasoy et al. 2018) and urban stream systems are important modes of invasive species spread (Säumel and Kowarik 2010, Aronson et al. 2017). Within river networks, downstream locations receive propagule inputs from upstream sources (e.g.., road runoff; Chytrý et al. 2008, Flanagan et al. 2015, Atasoy et al. 2018). Streamside *Microstegium vimineum*, our most common non-native plant, utilizes hydrochory to disperse seed via overland and channel flows (Tekiela and Barney 2013, Aronson et al. 2017). Therefore, because of the known association of invasive plants with urbanization, an influx of invasives post-restoration should be expected and planned for.

While some variation in invasion outcomes was related to soil nutrient and light availability, neither explained differences in outcomes between restored and unrestored reaches. Therefore, project attributes other than resource availability and geomorphology are likely responsible for the increased invasion of restored reaches. However, overall limitations of soil nutrients and a closed canopy are likely to reduce plant invasion. In many systems, the causes of variation in restoration outcomes are still poorly understood (Brudvig et al. 2017). Practitioners hope to predict restoration outcomes to replicate successes and learn from failures (Suding 2011, Perring et al. 2015, Brudvig et al. 2017); however, there has been a lack of comparison of outcomes across projects, limiting the information we can glean from inter-project variation (e.g., landscape context, methodology). For example, most stream restoration projects lack the relevant information to evaluate whether the goals of improved aquatic structure and function have been met (Bernhardt and Palmer 2011), let alone whether vegetation composition has been maintained or improved. Within the Chesapeake Bay watershed, most monitoring focuses on stream channel function, largely overlooking important ecological impacts and outcomes (Noe et al. 2024). In addition, cross-project comparison is difficult due to a lack of standard monitoring methods and rigorous assessments, such as before-after-control-impact studies (Noe et al. 2024). We also recognize the presence of unmeasured sources of variability such as management actions (e.g., composition of vegetative plantings, post-restoration monitoring/management, topsoil import), seed sources, disturbance (e.g., flooding, intensity of restoration), and nitrogen availability.

Given the size, restoration efforts, and monetary inputs of the Chesapeake Bay Program, our results uncover perverse impacts of stream restoration on revegetation for one of the largest and most well-funded restoration initiatives globally. However, this information can be leveraged to argue for improved practices, especially regarding resource limitation, and longer monitoring and management to minimize invasion and enhance overall ecological health of restoration projects.

Objective 2: Provide recommendations on stream restoration techniques and planting practices that facilitate native plant establishment and minimize colonization of invasive plants.

Introduction

Stream restoration projects have the potential to improve biodiversity and ecosystem functioning for instream and downstream ecosystems. However, in the Eastern United States, physical structures or channel form are prioritized over ecological outcomes (Wortley et al. 2013, Palmer et al. 2014). Within these riparian systems, vegetation does not receive the same emphasis as geomorphology, hydrology, or even aquatic biota (Rodríguez-González et al. 2022), especially with respect to restoration goals (Noe et al. 2024), even though it is essential for stream functioning. Riparian vegetation aids in bank stabilization (Hubble et al. 2010), supports biodiversity (Naiman et al. 1993, Hupp and Rinaldi 2007), serves as organic matter inputs, and can maintain stream temperature (Sweeney 1992). Despite the significant influence riparian vegetation has on stream health, U.S. federal laws do not afford protections to riparian vegetation in the same way as wetlands, water quality, or endangered species (Opperman et al. 2017). Stream structure and function are inextricably linked to riparian forests and restoration can be utilized to ameliorate declines in stream and forest health (Sweeney 1992). Therefore, considering ecological processes in stream restoration goals and design has become increasingly common (Bernhardt et al. 2005, Beechie et al. 2010, Palmer et al. 2014).

However, the disturbances caused by stream restoration projects also create opportunities for invasive plant species to infiltrate fragmented riparian forests (see Objective 1). Relative to most other ecosystems, riparian systems are under greater threat from invasive plant species due to increased connectivity, increased movement of propagules, and frequent disturbance (Flanagan et al. 2015, Schirmel et al. 2016). In addition, traits of some invasive plants may confer advantages within riparian systems (Catford and Jansson 2014), such as seed-dispersal adaptations allowing for water-dispersal in Tree-of-Heaven (*Ailanthus altissima*; Planchuelo et al. 2016) and flood tolerance of Chinese privet (*Ligustrum sinense*; Brown & Pezeshki 2000). Restoration of stream and forest function necessitates the limitation of invasive species. Varied negative impacts of invasive plants on stream ecosystems have been documented, including increased evapotranspiration, which reduces baseflows (Galster and Vanderklein 2023), altered biogeochemical cycling (Mineau et al. 2011), streambank instability (Colleran et al. 2020), and eutrophication (Robertson and Coll 2019). Further, climate change could increase invasibility through altered hydrology (Colleran et al. 2020) coupled with increasing constriction of urban riparian corridors (Johnson et al. 2020), giving invasive species with broader environmental tolerances a competitive advantage (Flanagan et al. 2015). While stream restoration often focuses on restoring stream health and geomorphology, practices increasingly need to account for invasive plants, especially as restoration itself could promote invasion through an influx in newly available resources due to site preparation, soil disturbance, and vegetation removal (Davis et al., 2000; see Objective 1).

Invasive plant risk is related to local and environmental factors, some of which are within restoration practitioner control, and others of which are not. Invasive species have been shown to respond to site-specific factors such as soil nutrient and other resource availabilities with increased abundance (Flanagan et al. 2015), increased invasion (Stohlgren et al. 1999), and differential species establishment (Vasquez et al. 2008). Landscape structure, such as proximity to the nearest urban center and the amount of habitat edge, have also been found to influence invasion (Loewenstein and Loewenstein 2005, Bartuszevige et al. 2006). The interaction between site conditions and landscape context can provide insight into invasibility of stream restoration projects as non-native propagules are often readily available across an urban landscape (Gaertner et al. 2017) and site conditions on restoration projects can be suitable for establishment (e.g., lower overstory cover; Holl & Crone 2004). Unlike resource availability and landscape context variables, those that reflect site disturbance or are related to revegetation and monitoring practices likely also impact invasive plant establishment and are potentially within practitioner control. Practices such as planting greater stem densities or incorporating adaptive management could assist in preventing or addressing invasion of stream restoration projects. Therefore, it is of interest to understand how to manipulate different factors to limit invasion during the project design, construction, and monitoring phases.

The Chesapeake Bay Watershed has been transformed by major investments in stream restoration, but relatively little is known about what factors or practices influence the quality of site revegetation. As of 2024, 430 km of stream have been restored (Noe et al. 2024), approximately 0.3% of the over 160,934 km of streams within the Chesapeake Bay watershed (CBC 2020). Stream restoration projects within the Chesapeake Bay watershed are often undertaken to obtain total maximum daily load (TMDL) credits, which is a relatively new practice, however the composition and regulation of riparian vegetation is often an afterthought (Noe et al. 2024). For example, in many cases, practitioners are required to seed disturbed banks immediately after construction, typically using seed mixes composed of non-native grasses. Because revegetation practices largely prioritize bank stabilization and erosion control, detailed evaluation of practices that may limit or promote invasion are needed.

As stream restoration projects increasingly seek to limit establishment of non-native species, it is necessary to understand the interactions among resource availability, land use, project attributes, planting design, and project monitoring variables and their relative influence of riparian vegetation. Discerning which factors are most important in determining vegetation composition on stream restoration projects can inform judicious use of limited resources and improved revegetation outcomes (i.e., increased cover and richness of native plants). Therefore, to establish best practices for limiting non-native vegetation, we compared restoration practices, site-specific factors, and the broader landscape context across 46 stream restoration projects completed between 1994 and 2016 in Maryland's Chesapeake Bay watershed. Invasive plant species are of increasing concern within the Chesapeake Bay watershed (Barnhart et al. 2024) and may be inhibiting watershed-wide restoration and ecosystem health goals (Fraley-McNeal et al. 2022). Thus, our objectives were:

1. determine which factors (e.g., site-specific, landscape, project attributes) influence plant invasion of stream restoration projects; and
2. provide recommendations to restoration practitioners on how to limit non-native plant invasion of stream restoration projects.

Results from this study will help us to understand how human disturbance impacts invasive plant establishment.

Methods

Study Area

In 1983, Maryland, Pennsylvania, Virginia, Washington D.C., the Chesapeake Bay Commission, and the EPA signed the Chesapeake Bay Agreement and created the Chesapeake Bay Program to counteract the degradation of this important estuary that had been degraded by nutrient and sediment inputs, overfishing and overharvesting, and deforestation (Barth et al. 1989). The other three states within the watershed, Delaware, New York, and West Virginia, joined the agreement in the early 2000s. Following executive order 13508 by President Barack Obama in 2009, restoration of the Chesapeake Bay watershed, largely through compliance with TMDLs of water pollutants, was further mandated through two-year milestones.

We selected 46 stream restoration projects for sampling within the Chesapeake Bay watershed. Streams were restored between 1994 and 2016 and are located within the urban and suburban areas surrounding Annapolis and Baltimore, Maryland, USA, and Washington, D.C., USA. The selected projects had been previously evaluated for geomorphologic restoration success (Thompson et al. 2021). Stream restoration projects lie within the Piedmont and Coastal Plain ecoregions of the eastern USA. All projects are also within the Baltimore-Washington-Arlington Combined Statistical Area, the third largest such area in the USA, home to ~9.9 million people (U.S. Census Bureau 2022). There were numerous environmental consulting firms responsible for the design and monitoring of the restoration projects as well as several state and federal agencies involved in permitting or as clients (e.g., Maryland Department of Natural Resources, U. S. Fish and Wildlife Service). Projects were largely completed either for hydrologic/geomorphologic reasons (e.g., water quality, erosion control, bank stabilization) or as mitigation.

Vegetation and Resource Availability Sampling

During the summers of 2023 and 2024, we sampled the riparian vegetation on the 46 stream restoration projects. We measured species composition and coverage, overstory basal area, photosynthetically active radiation, and took soil samples for soil nutrient analyses (as in Ripa et al.). Six sampling points were established along a 100 m sampling reach oriented about the midpoint of each restored stream, starting at 0, 5, 45, 50, 90, and 95 m and alternating banks. At each sampling point, we sampled vegetation composition and coverage using two point-intercept transects, one 1 m from the streambank and one 5 m from the streambank, each 5 m in length. Species touching the point were identified every 20 cm along the length of the transect, from ground level through the canopy.

At every meter along the transect 5 m from the streambank, we took five soil samples with a soil probe at a depth of 10-13 cm. A total of six aggregated soil samples were taken per project to estimate soil available soil nutrients for each restoration project. Four of the six samples were randomly selected for analysis of soil P, K, Ca, Mg, Zn, Mn, Cu, Fe, and B parts per million (ppm), as well as soil pH and CEC through the Virginia Tech Soil Testing Lab. We did not measure Nitrogen availability because it is temporally dynamic and transforms readily.

At the midpoint between the two transects (i.e., 3 m from the streambank and 2.5 m along the transects), we established a fixed-radius plot of 0.02 ha to measure overstory basal area using diameter at breast height. We considered overstory trees to be trees with a diameter at breast height (m = 1.37 m) greater than 11.4 cm (Waskiewicz et al. 2015). We also used the center point of the fixed-radius plot to measure photosynthetically active radiation (PAR) with a CI-110 Plant Canopy Imager (CID BioScience, Camas, WA, USA).

We followed the United States Department of Agriculture Plant List of Attributes, Names, Taxonomy, and Symbols (PLANTS) Database to determine native status of each species (USDA 2025).

We analyzed potential resource availability variables for covariance with other variables and eliminated correlated variables. Variables were selected based on their potential to influence non-native species cover.

- Phosphorus ppm, Potassium ppm, and base saturation: we expected these variables to reflect increased soil nutrient availability and thus, have positive relationships with non-native cover.
- Iron ppm: as Iron can be indicative of heavy metals or more alkaline soil (Zefferman et al. 2015), we expected increased iron ppm to create harsher environments, potentially limiting invasion.
- Basal area: we expected sites with greater basal area to have lower soil nutrients available and decreased light availability and thus, lower non-native cover.
- PAR: we expected sites with greater light availability to have greater non-native cover.

Surrounding Land Use

Land use variables were calculated using the 2016 National Landcover Database and watershed shapefiles obtained from previous research on the stream restoration projects (Thompson et al. 2021). Variables included watershed area, and areas of high density development, medium density development, low density development, forest, agriculture, and wetland/water as determined from the 2016 National Landcover Database within each watershed. We calculated the proportion of the watershed in the different land cover classes to standardize coverage across different size watersheds. Additionally, we considered the landscape context of the restoration project. We expected sites with greater proportions of high, medium, and low-density development and agriculture to experience greater pressure from invasive species through increased propagule availability. For sites with greater proportions of forest and wetland/water land uses, we expected lower propagule pressure. Most of the projects were in residential areas (n = 35), and the rest of the projects were in public parks (n = 4), rights-of-way (n = 2), commercial land (n = 2; e.g., shopping center), institutional land (n = 2; e.g., community college), and pasture (n = 1; Fig. 1).

Project Attributes

Project attributes were also obtained from the previous work by Thompson et al. (2021). Many of the variables were originally gleaned from design plans. Variables included mainstem construction length, restoration goal, whether habitat was a goal, restoration project design (e.g., natural channel design, regenerative storm conveyance), and structure density. See table S1 for a complete list and descriptions of project attribute variables.

- Construction length and structure density: construction length measures the length of restored stream and structure density was calculated as the number of restoration structures divided by the length of restored stream. We expected both to approximate the area of stream and riparian system that was disturbed by construction, with projects that disturbed a greater area exhibiting greater non-native cover.

- Design: a categorical variable of whether the project was designed as a natural channel design, regenerative stormwater conveyance, or wetland restoration.
- Goal: a categorical variable of whether the project was initiated as mitigation or to address bank stabilization/erosion control/water quality.
- Habitat as a goal: a categorical variable denoting whether improving in-stream habitat was a goal of the restoration project (Y/N). We expected projects that sought to improve habitat to have greater consideration for biotic outcomes, and therefore lower non-native cover.

Planting Design

Planting design variables were obtained from restoration design plans. This category of variables focuses on the actual revegetation and site preparation measures taken during project implementation. Variables included the planted stem density, seeding rate, whether plugs were used, species richness planted, non-native species planted, and whether sites were fertilized, among other variables. See table S1 for a complete list and descriptions of planting design variables.

- Proportion native stems, number of species planted, number of woody/forb/grasslike species planted, number of growth forms planted, stem density planted, seeding rate: We expected each of these variables to have inverse relationships with non-native cover, such that increasing the planted density or diversity would lead to lower non-native cover.
- Number non-native species planted: We expected non-native cover to increase with increasing number of non-native species planted.
- Plugs: a categorical variable denoting whether plugs were planted as part of the planting plan (Y/N). We expected use of plugs to limit non-native cover by giving herbaceous vegetation a head-start.
- Fertilization: whether fertilizer was added (Y/N). Fertilizer is often added to restoration projects post-disturbance to help vegetation establish but could potentially give invasive species a competitive advantage.
- Topsoil: whether topsoil was salvaged from the site, added (of unspecified origin), furnished (brought from an outside source), or not. Sites that had topsoil added or furnished might be more likely to introduce non-native species present in the soil.
- Tree preservation: whether trees within the limits of disturbance were left, fenced, or neither. Trees preserved by either leaving them or by fencing (as added protection) serve to limit soil nutrients, space, and light available for non-native species.
- Number/proportion of planted species detected in surveys: the number or proportion of species noted on the planting plans that we observed during vegetation surveys in the summers of 2023 and 2024. Sites where a greater number/proportion of species were detected potentially had greater survival of planting or better site maintenance which could help to limit non-native species.

Project Monitoring

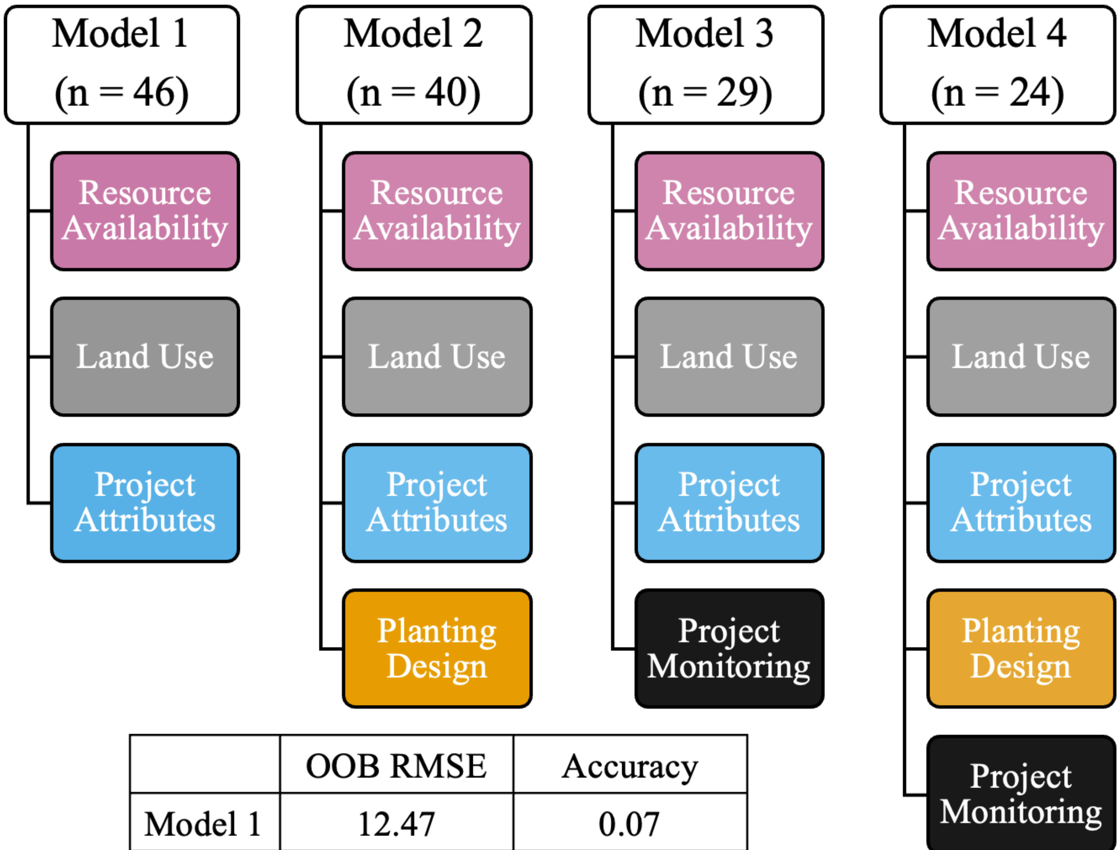
Project monitoring variables were collected from restoration project monitoring reports and focus on post-restoration monitoring and management actions that could impact non-native plant establishment. Examples of project monitoring variables include whether invasive species were monitored, the number of years the project was monitored, whether a reference site was used, and whether other taxonomic groups (e.g., invertebrates, fish, herpetofauna) were monitored. See table S1 for full descriptions of project monitoring variables.

- Years of monitoring: the number of years the project was monitored. Projects that monitor for longer have a longer window to respond to potential invasion.
- Reference site: whether a reference site was used to inform the restoration design or was used as a comparison during the monitoring period. Sites that utilize a reference site might have a better initial project design or be better able to detect when the restoration or vegetation community need correction.
- Invasive monitoring: whether invasive plants were monitored. Projects that specifically monitor invasive plants would be able to detect problematic invaders and to take action if needed to correct course.
- Monitoring of other taxa: whether other taxonomic groups (e.g., fish, invertebrates, herpetofauna) were monitored (Y/N). Projects that monitored other biotic aspects of the stream system may have given greater weight to biotic outcomes (including the vegetation community).

Analysis

We sought to predict non-native plant coverage (%) across the 46 stream restoration projects using the five categories of potential influential factors. We chose non-native plant coverage as our response variable because non-native species are prevalent in urban greenspaces, especially for horticultural purposes, and have been shown to have negative impacts on native communities (Nelson et al. 2017, Narango et al. 2018, Tallamy et al. 2021). Additionally, restoration practitioners in Maryland are mandated to keep invasive species coverage below 10% throughout the required monitoring period (Stream Mitigation Performance Standards and Monitoring Requirements in Maryland 2024), and while not all non-native species are invasive, ~93% of our observed non-native species are invasive. We were unable to obtain several design plans and monitoring reports, leaving gaps in planting design and project monitoring variables. Due to differences in availability across the five datasets (i.e., resource availability, land use, project attributes, planting design, project monitoring) of potential factors that could influence non-native plant coverage, we created four random forest models to best utilize the dataset (Figure 9). Random forest models create "forests" of decision trees to determine variable importance in predicting a response variable. Random forest analyses were conducted in R v4.4.2 (R Core Team 2025) using package *ranger* (Wright and Ziegler 2017).

Datasets for each model were split into training and testing data, approximately 70% training and 30% testing, stratified by non-native cover values. Using the training data, we tuned the models using a full cartesian grid search of hyperparameters to assess out-of-bag (OOB) root mean squared error (RMSE). Each of the four random forest models was run using impurity-based variable importance, n-dimensional hyperparameters, and 2000 trees. Model predictions were then calculated using the impurity model and the testing data to create a confusion matrix of prediction accuracy. Based on prediction accuracy and OOB RMSE, we selected a best model and re-ran the model with permutation-based variable importance. To determine whether certain categories of variables had greater importance than expected by chance, we performed a Chi-square goodness of fit test. We summed variable importance within each category and scaled by the number of variables within that category.



	OOB RMSE	Accuracy
Model 1	12.47	0.07
Model 2	9.20	0.08
Model 3	11.14	0.22
Model 4	8.39	0.63

Figure 9. The datasets included in each of the four random forest models and the number of observations included in each model.

Results

We observed a broad range of non-native species cover across the 46 projects (Figure 10), from approximately 10% to 59%, with a mean cover of 37% ± 12% SD. The most common invasive species by coverage were *Microstegium vimineum* (11% of the total coverage across all species), *Rosa multiflora* (3%), and *Ampelopsis brevipedunculata* (3%).

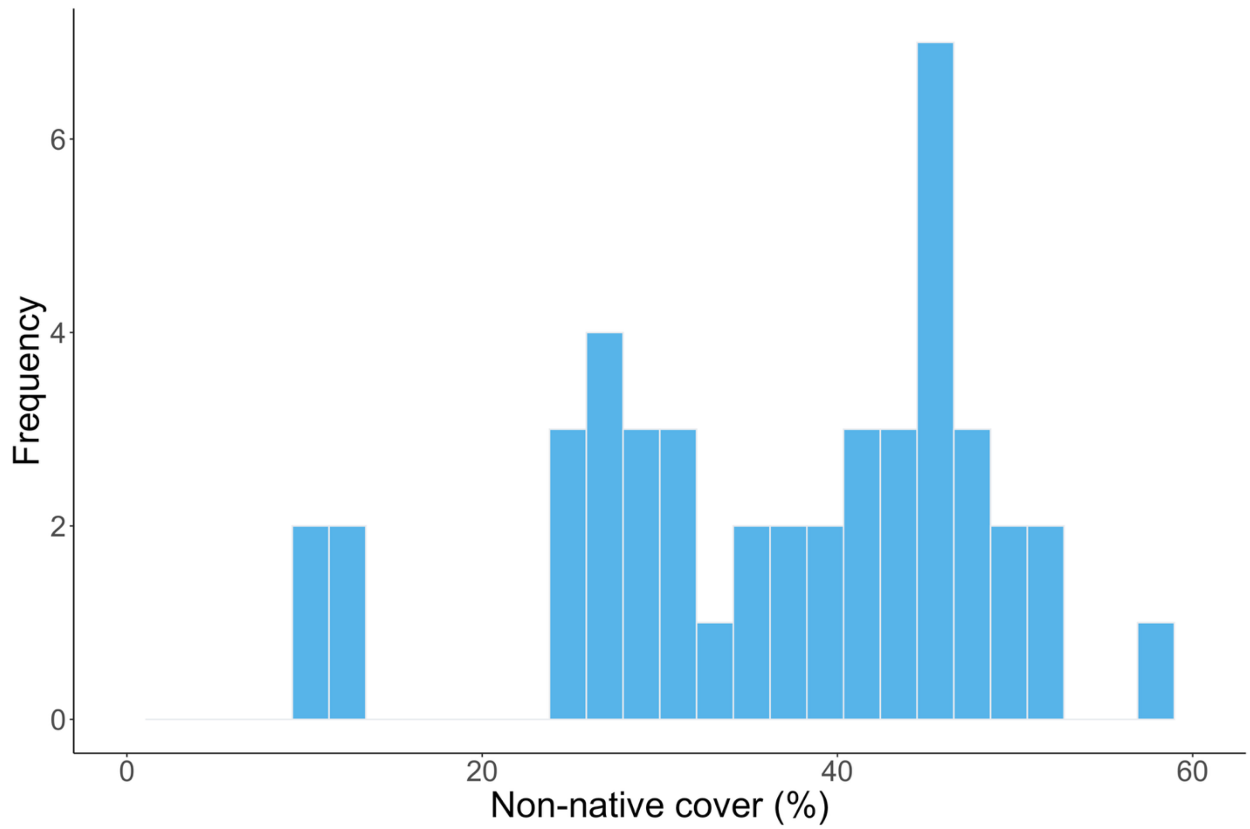


Figure 10. Histogram of the frequency of non-native cover values (%) observed across 46 urban stream restoration projects between 8- and 29-years post-restoration. The average non-native cover was 37% ± 12%.

We used impurity-based variable importance to determine which categories of variable were consistently important across models. Resource availability and land use variables were the most important variables across all models. We found that for model 4, some of the five categories had greater importance than expected (Table 2; $p < 0.0001$). For all other models, some categories were observed more than expected (Table 2).

Table 2. Relative importance of each category of variable for the four random forest models. Chi-square tests were used to determine whether certain categories of variables were more important than others, while accounting for the number of variables in each category. Impurity-based importance values were aggregated by category and divided by the number of variables in that category.

	Category	Observed	Expected	Chi-square	p-value
Model 1	Land Use	79.8	0.33	40.6	1.5e-9
	Project Attributes	78.5	0.33		
	Resource Availability	159.4	0.33		
Model 2	Land Use	52.8	0.25	340.7	2.2e-16
	Planting Design	34.6	0.25		
	Project Attributes	54.9	0.25		
	Resource Availability	260.4	0.25		
Model 3	Land Use	87.8	0.25	8.24	0.04
	Project Attributes	85.1	0.25		

	Project Monitoring	82.1	0.25		
	Resource Availability	116.6	0.25		
Model 4	Land Use	47.3	0.2	117.35	2.2e-16
	Planting Design	15.4	0.2		
	Project Attributes	11.0	0.2		
	Project Monitoring	6.3	0.2		
	Resource Availability	77.7	0.2		

Given the OOB RMSE and accuracy of each model (Figure 9), we have chosen model 4 (the model containing all five categories of data) as our best model with an OOB RMSE of 8.39 and an accuracy of 0.63 (Figure 11). We investigated relationships between non-native cover and the most important variables within each category from model 4. For most resource availability variables (i.e., PAR, soil K parts per million), as availability of the resource increased so did non-native cover, whereas soil Fe and basal area have somewhat inverse relationships (Figure 12A). In general, as proportion of low-density development increased, so did non-native cover (Figure 12B), whereas proportions of medium and high-density development had inverse relationships with non-native cover. Project attribute variables of construction length and structure density were of highest importance in the category, but had only slight positive relationships with non-native cover (Figure 12C). As the proportion of species planted that were detected in vegetation surveys increased, non-native cover decreased, while non-native cover had a positive relationship with the number of woody stems planted (Figure 12D). Projects that used a reference site had lower non-native cover than sites that did not (Figure 12E).

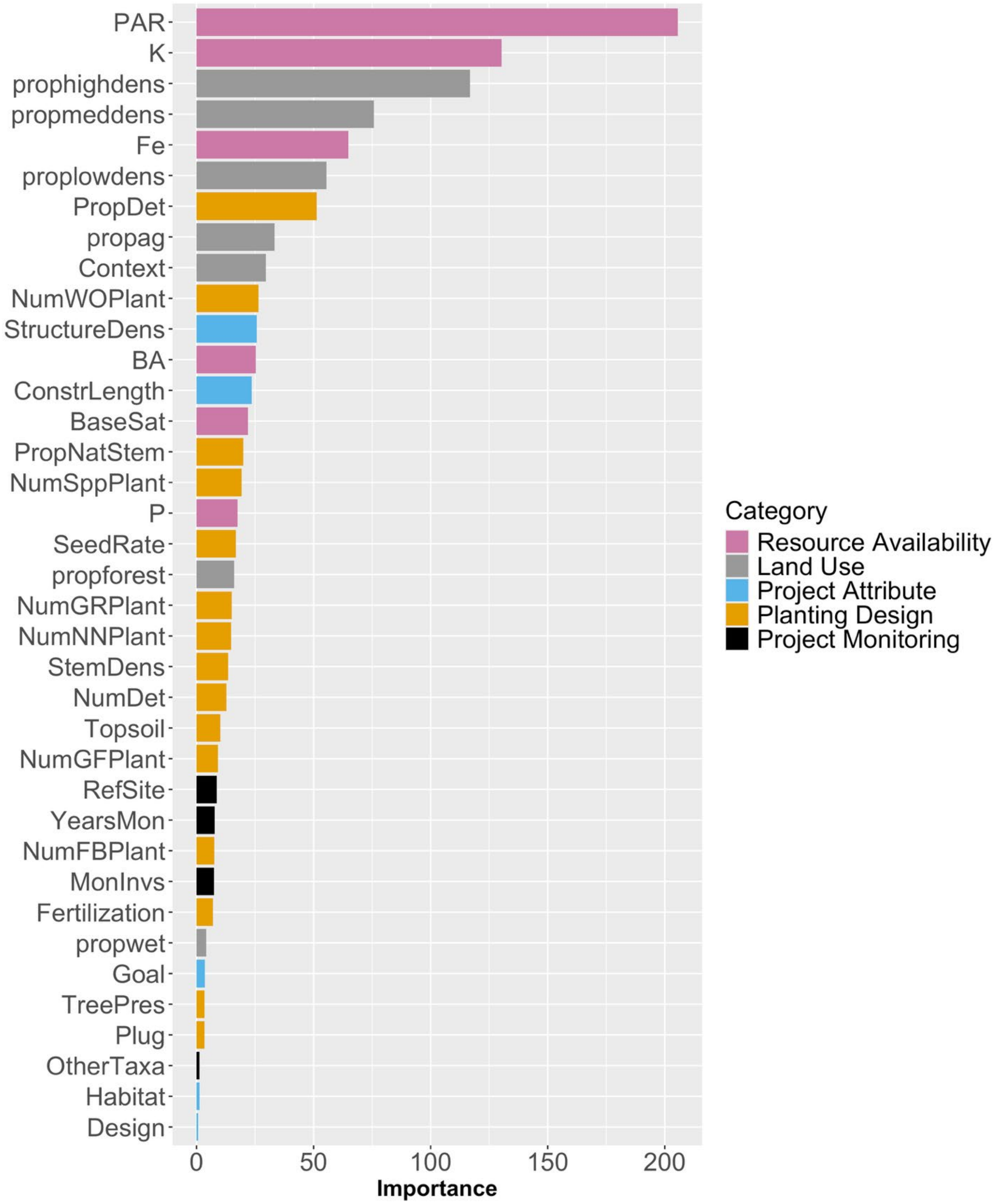


Figure 11. Impurity-based variable importance for the best random forest model which includes resource availability, land use, project attribute, planting design, and project monitoring variables.

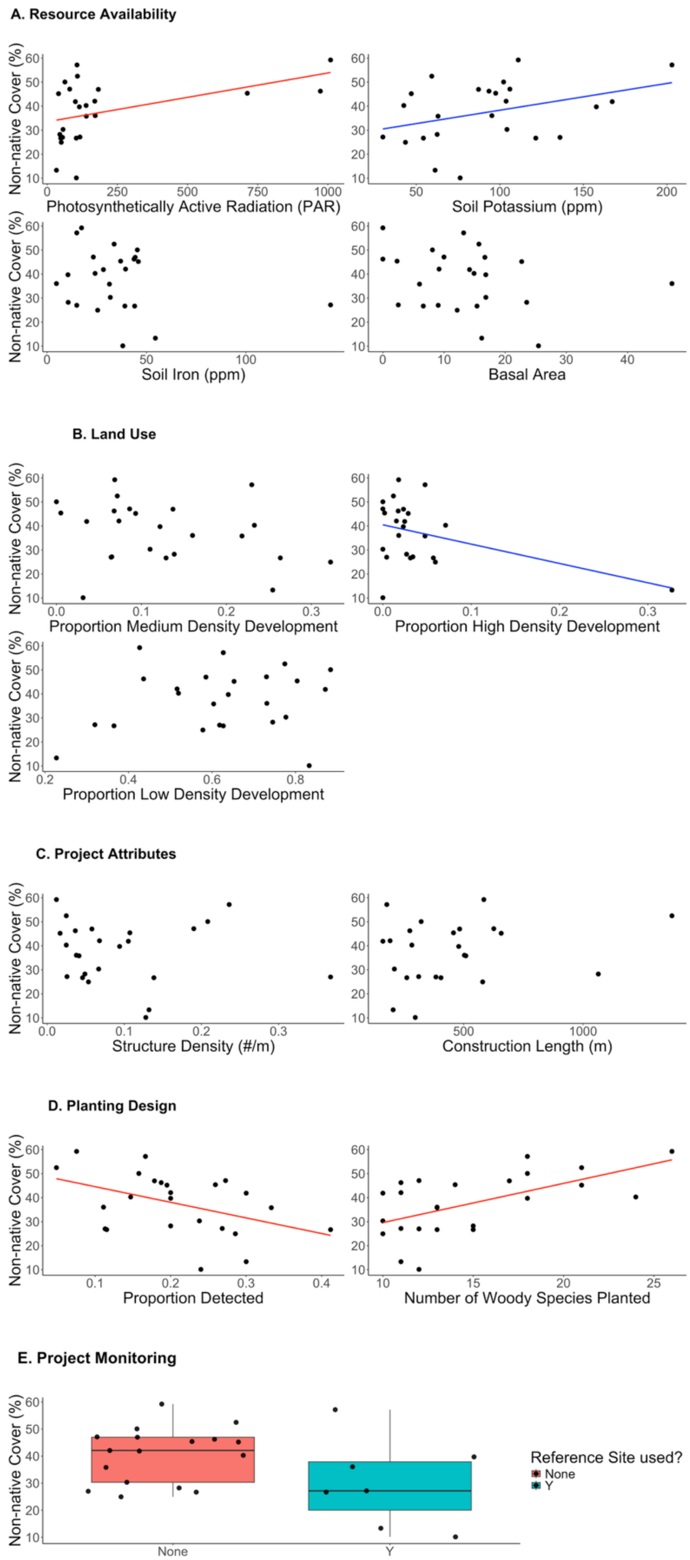


Figure 12. Scatterplots and boxplots of most important predictor variables from random forest model 4 within each variable category. A) Scatterplots of the relationships between resource availability variables of photosynthetically active radiation, soil potassium, soil iron, and basal area and non-native cover (%). B) Scatterplots of the relationships between land use variables of

proportions of low-, medium-, and high-density development in the watershed and non-native cover (%). C) Scatterplots of the relationships between project attributes of construction length and structure density and non-native cover (%). D) Scatterplots of the relationships between planting design variables of the proportion of species detected in vegetation surveys and number of woody species planted and non-native cover (%). E) Boxplot of the relationship between project monitoring variable of whether a reference site was used and non-native cover (%). Blue lines denote linear relationships with $p < 0.10$ and red lines denote linear relationships with $p < 0.05$.

Discussion

Urban stream restoration projects are hindered by urbanization and increased resource availability in limiting invasive plants. However, several project variables within practitioners' control were also important for predicting invasion. We observed an inverse relationship between the proportion of planted species detected on vegetation surveys and non-native cover. Additionally, projects that used a reference site had lower average non-native cover. While manipulating surrounding land use and resource availability are often not feasible, we have identified other aspects of stream restoration projects that can be controlled to limit invasion.

The positive relationship between resource availability and invasion has been identified many times before (Davis et al. 2000, Ehrenfeld 2008, Flanagan et al. 2015). We found positive relationships between soil potassium and light availability and non-native cover, similar to Gurevitch et al. (2008), who also found invasive plants responded more to increased availability of such resources than native plants. Light availability, measured as canopy cover, has also been previously identified as an important driver of populations of three common wetland invaders (i.e., *Arthraxon hispidus, Microstegium vimineum, Typha angustifolia*; Hunter & DeBerry, 2023). Our most common invader, *M. vimineum*, has been shown to perform poorly in excessively low-light conditions, such as under a midstory canopy without sunflecks (Cole and Weltzin 2005). While previous studies have recommended limiting nutrients in restoration (e.g., removing topsoil, planting low-nitrogen species, leaving mature trees; D'Antonio et al., 2016; Davis et al., 2000; Menuz & Kettenring, 2013; Vasquez et al., 2008), stream restoration design plans often include directives to add or furnish topsoil and fertilize bare ground immediately post-construction. Though broad resource limitation, especially on urban stream restoration projects, is challenging, there are some actions practitioners can take to limit resources available to potential invaders.

Proportions of high, medium, and low-density development were the most important variables for predicting non-native cover within the land use category. Because these measurements were calculated at the watershed scale (range: 0.098 – 95.13 km^2, mean: 6.17 km^2), they provide a broad snapshot of human disturbance around the restoration project sites, rather than pressure at the site-level. Restoration is necessary in urban areas, but urbanization can also limit the effectiveness of restoration, even beyond vegetation, due to flashy flows and increased sedimentation and pollution (Bernhardt and Palmer 2007). Urban and suburban areas and residential development, as captured by low and medium density development, are known hotspots of invasive spread (Duguay et al. 2007, Ehrenfeld 2008). Given many of the stream restoration projects ($n = 35$) are in residential areas, there are many opportunities for non-native plants to establish in stream corridors. For instance, suburban gardens are more likely to be planted with non-native species than native (Ward and Amatangelo 2018) and many invasive plants were originally introduced for horticultural or ornamental reasons (Bell et al. 2003, Niemiera and Holle 2009). Stream restoration project design and management should plan face hurtles associated with urbanization.

In combination with increased resource availability and surrounding urban land use, the disturbance associated with stream restoration potentially aids in invasive plant establishment (see Objective 1). Our proximate measures of disturbance due to restoration, especially mainstem construction length, could capture human disturbance. Anthropogenic disturbance has been shown to positively impact non-native plant species diversity and abundance, particularly when soil is disturbed, nutrients are added, and invasives are already present in the community, as in urban areas (Jauni et al. 2015). Similarly, Aryal et al. (2022) also found that anthropogenic disturbance was a major factor in invasion success in urban forests. Because of the intensive construction stream restoration often entails, the associated soil and overstory disturbances, especially over longer reaches, could increase invasibility by creating high disturbance, high productivity conditions (Huston 2004). Further, the worst invasions often occur post-disturbance, prior to native or planted species establishment (Guo et al. 2018). While most stream restoration projects in the Chesapeake Bay watershed are required to seed, often non-native grasses, immediately after disturbance to reduce soil erosion, such actions may be introducing potential invaders or be insufficient to effectively sequester excess resources to prevent other invaders.

Though urban stream restoration could lead to increased invasion (see Objective 1) through human disturbance and associated increases in resource availability, these restoration projects are still necessary, either as mandated mitigation or for ecological reasons (e.g., water quality, erosion control). Site selection to avoid projects in resource-rich, highly human-modified locations is often not an option. Therefore, understanding how to manipulate planting design or project monitoring factors is likely more useful for practitioners seeking to limit non-native plant cover on stream restoration projects. For example, the relationship we observed between decreased non-native cover on sites where we observed a greater proportion of planted species between 8- and 29-years post-restoration could be reflective of sites with better revegetation establishment. Incorporating improved site maintenance into planting plans (Sweeney and Czapka 2004) could promote survival and growth of planted species through reduced competition with non-native species. Most of the restoration projects examined were only required to monitor for 3 to 5 years, however, there is evidence that lack of monitoring and maintenance beyond the requirements can lead to both decreased geomorphological function and poor vegetation outcomes (Moore and Rutherfurd 2017).

Identifying when the project recovery trajectory is off track is an important aspect of managing and maintaining restoration projects. While only nine projects used a reference site either to inform restoration or for comparison, whether a project used a reference site was an important predictor of non-native species coverage. Though seldom used in stream restoration to inform planting plans, reference sites, communities, and/or models are a key component of most terrestrial restoration plans (Gann et al. 2019). In addition to the general lack of reference sites and longer-term monitoring, no sites utilized adaptive management, only one mentioned managing against invasives, and three had an invasive management plan. Thus, there is a pressing need to plan for invasive species monitoring and management and to document implementation of such actions. Current stream restoration monitoring and management practices and regulations seem insufficient to prevent lasting invasion (see Fig. 2 and Objective 1). Invasive species will likely become more competitive with a changing climate (Johnson et al. 2020), increased nutrient availability (e.g., from urban and agricultural runoff), human disturbance, and altered hydrological regimes (Flanagan et al. 2015). Given that invasive species and the factors that promote their persistence are wicked problems (Rittel and Webber 1973), invasive plant management should be considered and planned for from restoration project outset (Guo et al. 2018).

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