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Five-Year Response of Spontaneous Vegetation to Removal of Invasive Amur Bush Honeysuckle Along an Urban Creek

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Five-Year Response of Spontaneous Vegetation to Removal of Invasive Amur Bush Honeysuckle Along an Urban Creek

Rebecca W. Dolan[1,*] and Kelly H. Brown[2,3]

Abstract - Non-native invasive species have major impacts on landscapes worldwide, but their effects in urban areas are not well documented. We quantified the response of naturally regenerating vegetation along an urban creek to removal of the invasive shrub *Lonicera maackii* (Amur Bush Honeysuckle). Over the 5-year study, species richness more than doubled. Most new plants were native, disturbance-adapted, early successional species. Trend analysis of function traits revealed annuals that rely on seed dispersal by wind or externally on animals were significantly overrepresented among new plants in comparison to their proportion in the countywide species pool. Increased species richness did not result in improved habitat quality, as indicated by Floristic Quality Assessment. Eight new invasive species appeared over the course of the study. Active management of this site may be needed in perpetuity.

Introduction

Non-native invasive species are an economic and ecological threat across the globe. The annual economic impact of invasive plants, animals, and other organisms on agriculture, forestry, and fisheries is estimated be to >150 billion dollars in United States (Pimental et al. 2005) and more than 12 billion Euros in Europe (van Ham et al. 2013). Invasive species also threaten habitat integrity in natural areas. Threats include decreased native species diversity (Vilà et al. 2011) and reduced ecological services, including pollination, water purification, soil stability, climate regulation, and flood mitigation (Pejchar and Mooney 2009).

The shrub *Lonicera maackii* (Rupr.) Herder (Amur Bush Honeysuckle [ABH]; Caprifoliaceae) is known to grow outside of cultivation in at least 26 US states and 1 Canadian province (USDA-NRCS 2018). Native to China, ABH is an invasive plant throughout much of its introduced range, including the midwestern United States. Amur Bush Honeysuckle is more pervasive in urban forests compared with rural ones (Borgmann and Rodewald 2004), with stem densities greater in infested urban sites (Trammell and Carreiro 2011).

Once promoted as a desirable species by the USDA Soil Conservation Service (Luken and Thieret 1996), ABH negatively impacts ecological quality of sites by decreasing presence of native plants (Collier et al. 2002, Gould and Gorchov 2000, Hartmann and McCarthy 2007), increasing erosion (Luken and Thieret 1996), and decreasing nesting success of birds (Schmidt and Whelan 1999). A recent review

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by McNeish and McEwan (2016) highlighted a wide range of negative ecological effects of the shrub, from species-level to community- and landscape-scale.

Amur Bush Honeysuckle is associated with lower density and diversity of trees and herbs (Hutchinson and Vankat 1997) and reduced presence of shade-tolerant and early season annuals (Gorchov and Trisel 2003) in forests. Plant cover, tree seedling density, and species richness all increase (Hartmann and McCarty 2004, Runkle et al. 2007) following ABS removal, indicating restoration efforts to remove ABH positively affect natural plant communities.

A major community project focused on ABH removal was begun in Indianapolis, IN, USA, in 2012 (Dolan et al. 2015). It was organized by Keep Indianapolis Beautiful (www.kibi.org), a non-profit organization that "engages diverse communities to create vibrant public places, helping people and nature thrive", and corporate partner Eli Lilly and Company (www.lilly.com). To date, that project has resulted in the removal of ABH from more than 12 ha of the riparian border of Fall Creek (USGS Hydrologic Unit Code 0512020109; https://water.usgs.gov/GIS/huc.html), a tributary of the White River, part of the Mississippi River system. The removal is part of a broader community effort, Reconnecting to Our Waterways (www.ourwaterways.org), focused on making the city's urban waterways a better community asset.

From the outset of that project, it was recognized that 3–5 years of repeated treatment would be needed to reduce ABH to levels that would allow reestablishment of desirable understory vegetation through a combination of active (e.g., sowing of seed or planting of plugs) and passive (e.g., recolonization from the seed bank or via natural dispersal) restoration along Fall Creek (Dolan et al. 2015). This project presented the opportunity to conduct a multi-year study monitoring vegetation change at the site to document spontaneous flora that regenerated "naturally" from the seed bank or via seed dispersal into the site. Few studies have addressed vegetation response to removal of ABH in riparian habitats, especially highly altered, narrow borders of streams and creeks in cities, despite the documented potential of invasive shrubs to modify composition and structure and to diminish ecological function of urban riparian systems (Greene and Blossey 2012, Pennington et al. 2010, Richardson et al. 2007). Riparian zones provide important ecological services, including improving water quality and providing wildlife habitat (Sabo et al. 2005).

This paper presents quantitative vegetation analysis of plants growing outside of cultivation in transects along Fall Creek in Indianapolis prior to ABH removal and in the 5 years following removal. We hypothesized ABH was inhibiting the frequency and cover of other plants and predicted that if the number and cover of ABH were significantly reduced, then species richness and habitat quality would increase. Our goals were to quantify success of treatments to remove ABH and to document and characterize, based on their functional traits, the plants establishing from the seed bank or via dispersal or vegetative spread following ABH removal. We also sampled vegetation along transects to assess changes in indicators of habitat integrity and quality, based on Floristic Quality Assessment and Shannon diversity, for this urban riparian zone following ABH removal. Our study addresses the knowledge gap in understanding urban riparian vegetation resilience in response

to removal of an invasive shrub. Analysis of functional traits of the plants involved is a novel approach.

Field Site Description

Indianapolis, IN, the twelfth largest city in the United States, is in the Midwestern USA. The city is located in Marion County in the Central Till Plain section of the Central Till Plain Natural Region (Homoya et al. 1985). Historically, mesic upland forest, mostly *Fagus* (beech)–*Acer* (maple) association (Potzger et al. 1956) covered 76% of the county, with small areas of drier upland forest on ridges. Wet-mesic depressional forests were scattered throughout the county, with floodplain forests along major rivers and tributaries.

Fall Creek runs through urbanized regions of the city (Fig. 1). Much of the flood plain of the creek is owned by the city. During the early 1920s, a parkway and boulevard system was developed in Indianapolis along its urban waterways, including Fall Creek. The plan included the planting of native and non-native plants to beautify the area (Dolan et al. 2015). The current riparian border is composed of trees and understory species, with a variety of widths from 5 m to 45 m. The study area is public land surrounded by private property, much of which is heavily invaded by ABH.

Methods

Removal of Amur Bush Honeysuckle

On 11 October 2012, trained volunteers from the pharmaceutical company Eli Lilly and Company removed >760 m^3 of ABH from 12 ha of land along Fall Creek by cutting and lopping the shrubs. Rodeo Herbicide© (active ingredient glyphosate), manufactured by Dow Agrosciences, was applied to cut stumps. It was selected based on efficacy in killing ABH (Hartman and McCarthy 2004, IPSAWG 2006), short half-life in soil (Giesey et al. 2000), and approved use near waterways (Dow AgroSciences 2017).

Cuttings were dragged to the curb for chipping and disposed off-site. Volunteers spread seed of native grasses (e.g., *Panicum virgatum* L. [Prairie Switch Grass] and *Sorghastrum nutans* (L.) Nash [Indian Grass]) and forbs (e.g., *Heliopsis helianthoides* (L.) Sweet [False Sunflower] and *Silphium perfoliatum* L. [Cup Plant]) not found growing at the study site at the time. Details of the project and its logistics are in Dolan et al. (2015). In subsequent years, an environmental consulting firm was contracted to continue eradication efforts including lopping and herbicide treatment in fall before mature berries were present, as well as foliar herbicide applications in early spring prior to other vegetation leafing out.

Vegetation sampling protocol

In the late summer of 2012, before ecological restoration work targeting ABH removal took place, we established three 100-m long transects parallel to Fall Creek (Fig. 1). We counted ABH individuals within 25-m^2 quadrats located every 10 m along the transects and recorded the percentage of the quadrat covered by ABH. We estimated aerial cover using a modified Daubenmire cover scheme (Daubenmire

1959, McCune and Grace 2002), where 1 = 1–7%, 2 = 8–25%, 3 = 26–50%, 4 = 51–75%, 5 = 76–93%, 6 = 94–100%. We also recorded all species present in the quadrats. These data established the pre-treatment condition of the vegetation. We were restrained to late summer sampling due to short lead-time in being invited to participate in this project.

For each of the next 5 years (2013–2017), we resampled the vegetation to monitor changes following removal of ABH. Annual sampling was conducted at

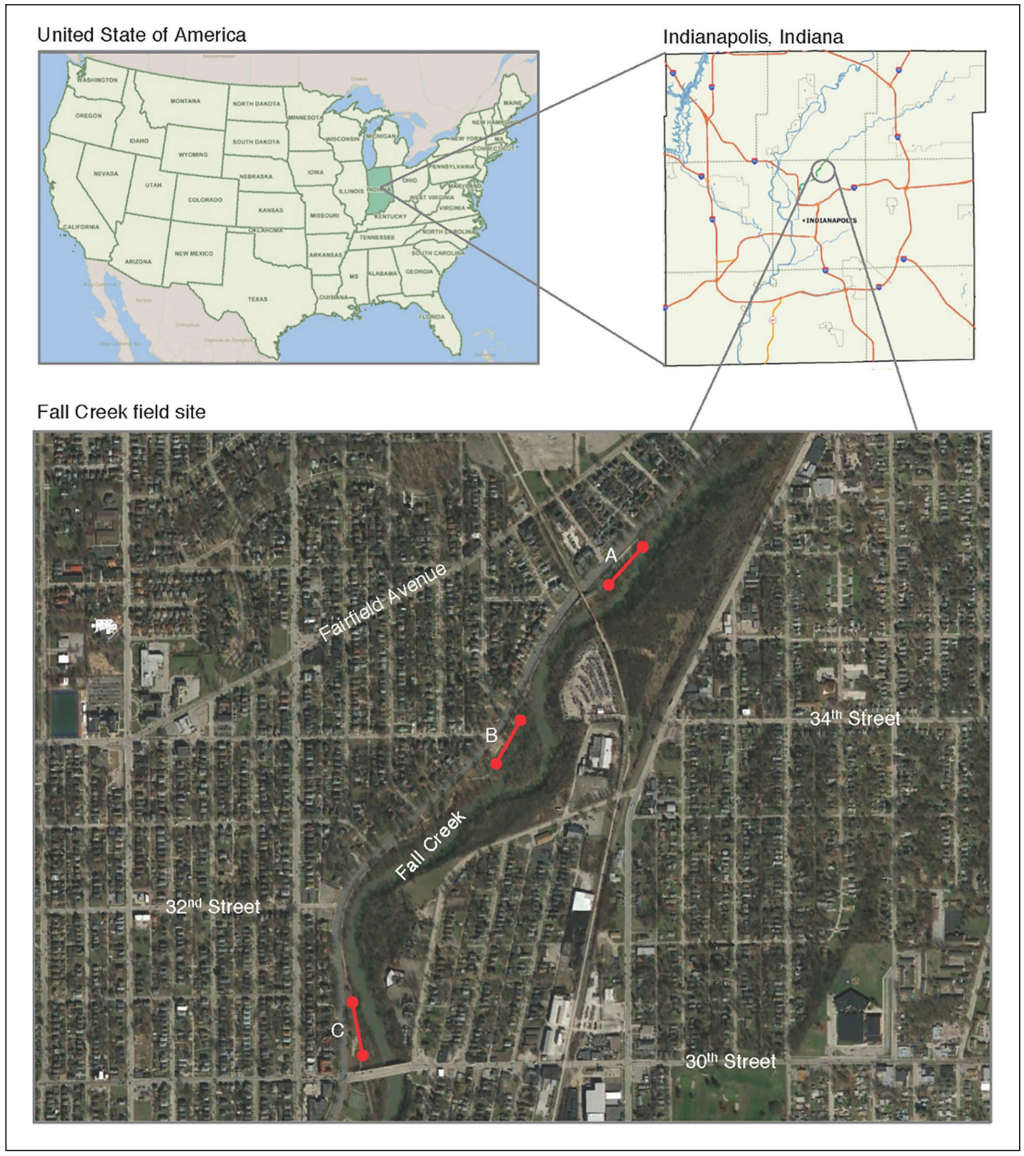

Figure 1. Location of Fall Creek study site in Indianapolis, IN, USA, and the100-m transects (red bars) used to document vegetation change following Amur Bush Honeysuckle removal along the creek (Google Earth 2018). Transect midpoints: A 39°49'17"N, 86°8'9"W, B 39°49'07"N, 86°08'19"W, C 39°48'41"N, 86°8'32"W.

the same time of year to minimize the chances of seasonal phenology differences affecting the ability to detect species' presence. Note that the transects were in the same general location but that the exact same sample quadrats were not surveyed each year. The same data collected in 2012 were collected in the subsequent 5 years. We did not permanently mark transects due to lack of lead time for approval on city-owned property and the high likelihood of vandalism.

Statistical analyses

In additon to nativity, we scored 8 functional traits related to important life-history traits (e.g., physiognomy and dispersal) categorically for each species present in the quadrats (Table 1). Traits were those used in a recent examination of the effects of urbanization on the flora of the entire county (Dolan et al. 2017), which are modified versions of Cornelissen at al. (2003) standardized protocols for measurement of plant functional traits worldwide. Traits are not necessarily independent of each other (e.g., growth form and pollination mode).

Using chi-square contingency tables, we analyzed differences in the frequencies of functional trait states for all plants present in quadrats in 2012, before ABH removal, and all plants present in 2017, after 5 years of treatment. Species known to be planted or sown during the course of the study were not included. Our interest here is in naturally recruiting plants. We performed analyses using the Cross Tabulation function in MYSTAT (www.systat.com). We used Pearson chi-square, Yates corrected chi-square, and Fisher's exact test (needed when a least 1 tally value was less than 5, i.e., a sparse cell in the tabulation) to establish

Table 1. Functional traits scored for trend analysis of floristic change along Fall Creek following Amur Bush Honeysuckle removal.

Category/trait	Category/trait	Category/trait
Nativity	Life Form	Dispersal Mode
Native	Phanerophytes	Unassisted
Exotic	Chamaephytes	Wind
	Hemicryptophytes	Internal animal
Growth Form	Geophytes	External animal
Forb	Therophytes	Hoarding
Fern	Helophytes	Water
Graminoid	Hydrophytes	More than 1 value
Shrub	More than 1 value	
Tree		Clonality
Succulent	Life Span	Non-clonal
Vine or climber	Annual	Aboveground
Parasitic	Biennial	Belowground
Aquatic	Perennial	
More than1 value	More than 1 value	Leaf Periodicity
		Evergreen leaves
Spinescence	Pollination	Deciduous leaves
No spines/thorns	Abiotic	Semi-evergreen
Spines/thorns	Biotic	
Missing	More than 1 value	

significance levels. All gave the same results, so we include only the Pearson chi-square analysis here.

We also compared characteristics of the 2012 flora with the flora of the entire county to determine how different the suite of functional traits possessed by plants in our sample quadrats prior to treatment were from traits of the countywide species pool (Dolan et al. 2017). Finally, we pooled traits of all species appearing in quadrats ("new" plants) during 2013–2017 and compared the characteristics of these plants with the countywide flora. This approach allowed us to determine if functional traits of new plants represented a random sample of traits from the available species pool.

To assess plant species diversity, we used Shannon diversity (based on Shannon–Weaver diversity), calculated using PC-ORD (McCune and Mefford 1999). To quantify species' fidelity to high quality habitats reflecting remnant quality, we used natural vegetation coefficients of conservatism (C-values; Swink and Wilhelm 1994). In this method, native species have C-values from 0–10 based on their perceived fidelity to natural plant communities. Higher numbers indicate intolerance of disturbance and restriction to remnants reflecting pre-settlement conditions (Rothrock 2004). C-values used were those developed specifically for the Indiana flora by Rothrock (2004).

We derived floristic quality index (FQI) values from C-values using Floristic Quality Assessment software (Wilhelm and Masters 2004). Floristic quality index is calculated as follows:

$$FQI = \Sigma(CC_i)/\sqrt{(N_{native})},$$

where CC_i = the coefficient of conservatism of plant species i, and N_{native} = the total number of native species occurring in the community being evaluated. We calculated FQI values for each year to provide an integrated metric of species' tolerance of disturbance, as an indicator of over-all floristic quality. Higher FQI numbers indicate greater natural habitat integrity. Changes in FQI values over time at a given location reflect changes in habitat quality, with increasing values indicating site quality improvement from an ecological perspective.

Lastly, we also used Floristic Quality Assessment software to assign physiognomic class (e.g., tree, herbaceous perennial, annual, perennial) to our species to allow additional comparisons of the structure of the vegetation present in our quadrats during each sample year. Nomenclature follows Rothrock (2004), which is based largely on the Flora of North America (www.floranorthamerica.org).

Results

Vegetation change following removal of Amur Bush Honeysuckle

Mean density of ABH was 0.52 plants per m^2 in 2012, prior to eradication efforts (Fig. 2a). The first year of treatment did not significantly reduce the density of plants, but continued treatment resulted in an 80% reduction by 2015, with significantly fewer plants. This trend did not carry over to 2016 and 2017; density of plants remained low, but did not continue to decline significantly in those years.

Unlike effects on density, the initial year of honeysuckle removal produced a significant reduction in mean cover class, from 26–50% to 8–25% per quadrat on average (Fig. 2b). There was a slight rebound in coverage in 2014, but still significantly less than in 2012 before treatment. In 2015, mean cover was significantly lower than in 2014, dropping to the 1–7% coverage range. Similar to the 2016–2017

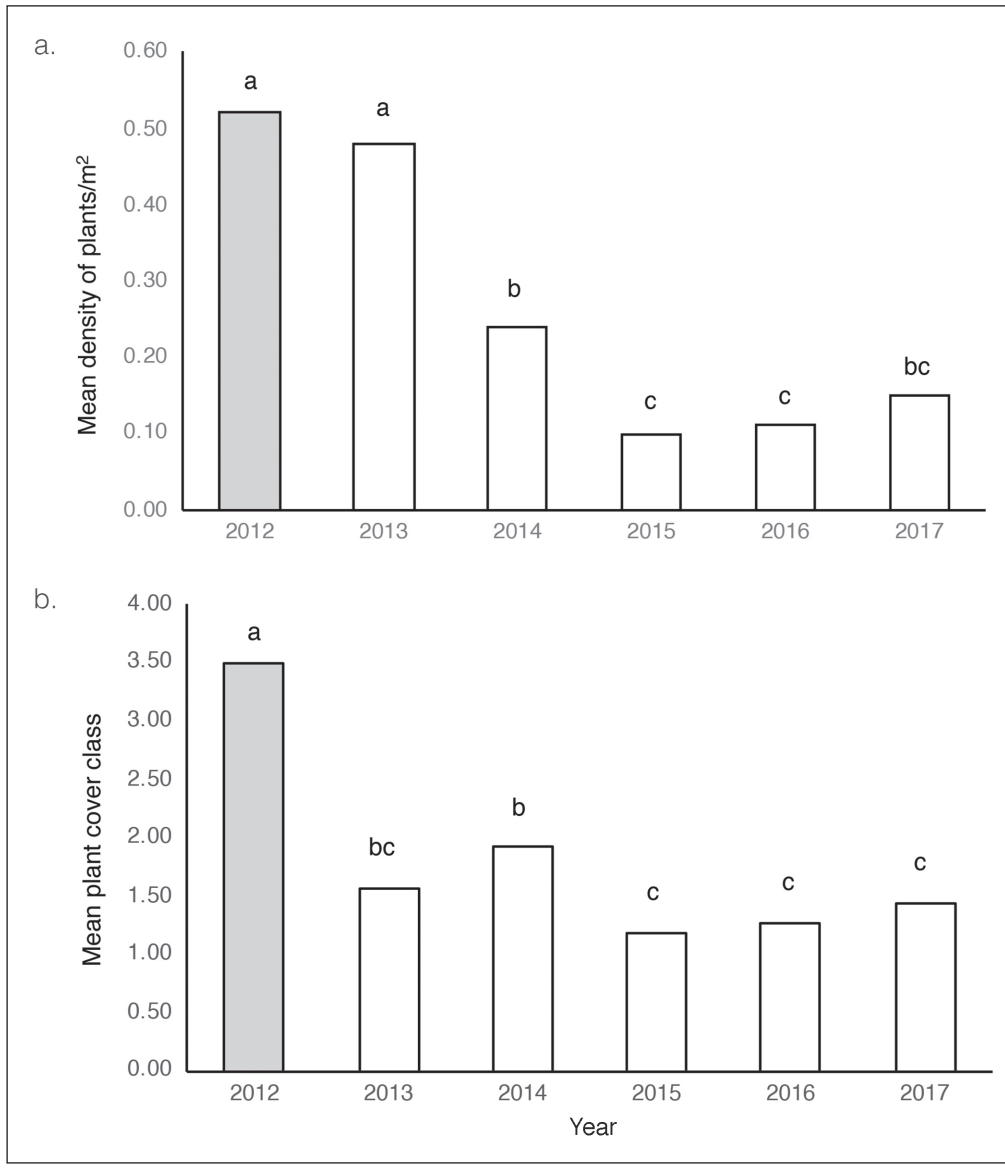

Figure 2. Effectiveness of Amur Bush Honeysuckle removal efforts along Fall Creek. (a) Mean density of plants and (b) mean cover class before removal efforts began in 2012, and in each of 5 years of continued treatment. Cover classes: 1 = 1–7%, 2 = 8–25%, 3 = 26–50%, 4 = 51–75%, 5 = 76–93%, 6 = 94–100%. Bars with different letters are significantly different at $P < 0.05$ based in Wilcoxon signed-rank tests, used for analysis because data did not meet assumptions of normality (MYSTAT).

data for density, cover class remained statistically the same while increasing a bit in absolute value from 2015–2017.

Over the course of our study, a total of 124 taxa were present in our transects. Species presence and frequency in plots are presented in Table S1 (see Supplemental File 1, available at https://www.eaglehill.us/URNAonline/suppl-files/U157-Dolan-s1). Herbaceous plants constituted 60% of the flora, comprised primarily of perennial forbs (35) and annual forbs (14). Woody species were dominated by trees (30), followed by vines (11) and shrubs (8). The vast majority of plants (70%) were native. The number of invasive, non-native species documented that are of management concern in Indiana doubled from 8 in 2012 to 16 in 2017. Most are ranked as being of "high" concern in the state, presenting special management concerns and challenges, due to their demonstrated negative ecological impacts and costs to eradicate. All but 3 invasives were woody; the 3 non-woody invasives were biennial forbs (Table S1; see Supplemental File 1, available at https://www.eaglehill.us/URNAonline/suppl-files/U157-Dolan-s1).

We detected 84 new spontaneously appearing species during the course of the study. Sixty-seven percent were native, with C-values mostly from 0 to 3, indicative of disturbance-adapted early successional species, not specialists indicative of mature remnant natural habitat (Table S1; see Supplemental File 1, available at https://www.eaglehill.us/URNAonline/suppl-files/U157-Dolan-s1).

The total number of species present increased each year following ABH removal (Table 2). More than twice as many species were present after 5 years of treatment than were present before treatment. Percent native fluctuated little through time, varying from 67% to 74%. Floristic quality index values for all years are within 1 unit of each other and are not influenced greatly when presence of non-natives is included in the analysis. Shannon diversity reflected the increase in species richness, increasing 3-fold over initial conditions by 2017.

These floristic changes manifest as an increase in the total number of species per quadrat each year following ABH removal in 2012 (Fig. 3). Increases were primarily due to a larger numbers of native plants. After 5 years of treatment, the average number of native plants per quadrat increased over fourfold (2.5 to 11.0), while the number of non-natives tripled (1.2 to 3.7).

The most common native species, based on frequency, in all years were *Celtis occidentalis* L. (Hackberry) and *Toxicodendron radicans* (L.) Kuntze (Poison Ivy).

Table 2. Floristic Quality Assessment metrics and Shannon diversity for flora in Fall Creek transects 2012–2017.

Trait	2012	2013	2014	2015	2016	2017
Total sp.	35	62	65	67	73	78
No. native sp.	26	45	44	45	52	58
Percent native	74.3	72.6	67.7	67.2	71.2	74.4
Native FQI	15.0	16.3	16.5	14.8	16.2	16.8
FQI with non-natives	13.2	13.8	13.7	12.4	13.8	14.2
Shannon diversity	0.8	1.2	1.9	1.8	2.1	2.4

The most common non-natives were ABH and *Euonymus fortunei* (Turcz.) Hand.-Maz. (Winter-creeper). Winter-creeper increased in presence from 17 quadrats to 25 of 30 quadrats over the course of our study, with 2017 mean cover of 50–75% for each quadrat where it was present (data not presented). Other notable changes in the presence of individual species included seedlings and saplings of *Acer negundo* L. (Boxelder) and Hackberry, native trees, which increased in frequency from 0 to 18 and from 8 to 22 quadrats, respectively, over the course of the study. The perennial forb *Ageratina altissima* (L.) King & H. Rob. (White Snakeroot), not present in 2012, was present in over 60% of quadrats in 2016 and 2017 (see Table S1 in Supplemental File 1, available at https://www.eaglehill.us/URNAonline/suppl-files/U157-Dolan-s1).

Plant functional traits

Trend analysis showed 3 of 9 functional traits were significantly different between plants present in 2012 and 2017 (Table S2; see Supplemental File 1, available at https://www.eaglehill.us/URNAonline/suppl-files/U157-Dolan-s1). Comparing the flora present at the start of the study in 2012 with that present in 2017, growth form differed, with herbaceous species (forbs and graminoids) comprising a greater percentage of the flora in 2017, while trees had reduced in percentage. This difference was reflected in life form as a reduction in phanerophytes (mostly trees), with a concomitant increase in hemicryptophytes (rosette-producing plants), geophytes (herbaceous perennials), and therophytes (annuals) after 5 years of ABH removal. The 2017 flora consisted of plants more likely to have unassisted or external animal dispersal, with fewer plants dispersed internally by animals, as a percentage, than the 2012 flora.

Plants present along Fall Creek at the start of the study were comprised of functional trait states differing in composition from those present in the flora of the entire county. Trees and woody vines, perennials, non-clonal plants, and

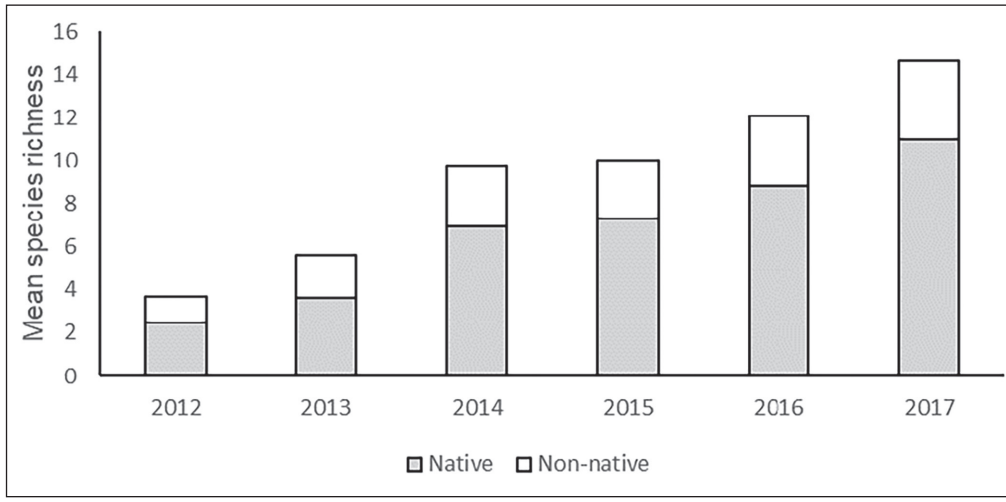

Figure 3. Mean species richness per quadrat (n = 30) along Fall Creek for each sample year.

those dispersed by wind or internally by animals were highly significantly over-represented in the 2012 Fall Creek flora compared with the countywide flora which comprised the local species pool.

New plants appearing in the flora in our quadrats along Fall Creek after ABH removal began (2013–2017) did not reflect a random sampling of the countywide species pool. New plants were significantly more likely to be annuals with dispersal by wind or externally by animals.

Discussion

Quantification of success of treatments to remove Amur Bush Honeysuckle along Fall Creek

Invasive species negatively affect ecosystem processes and decrease native biodiversity and species richness (Hejda et al. 2009, Vila et al. 2011). Amur Bush Honeysuckle degrades invaded habitats in multiple ways including through the release of allelopathic compounds that inhibit germination and growth of other species (Dorning and Cipollini 2005, McEwan et al. 2010) and a long leaf-holding period and growing season (Hutchinson and Vankat 1997, McEwan et al. 2009) that deprive co-occurring species of sunlight and exacerbate competition for nutrients and water. The significant reduction in mean density and mean cover class of ABH along Fall Creek over the course of this study has the potential to release the flora from these constraints, if they are operating, supporting the first assumption of our hypothesis on the likely effects of ABH removal. The greatest reduction in number and cover came in the first or second year of treatment. It may not be possible to completely eradicate ABH at this site, even though the shrub is thought not to have a persistent seed bank (Luken and Goessling 1995), due to seed input via the continuing high density of ABH on the other side of the creek and in the adjoining residential areas. Fall Creek also serves as a corridor for continued input of seeds via transport by flood waters and animals. Seeds remain viable when submerged (McNeish and McEwan 2016) and are dispersed primarily by birds (Bartuszevige and Gorchov 2006). Fall Creek is a major migratory corridor for birds. Treatment and removal of ABH will likely be needed on an on-going basis because of this continuing propagule pressure and because ABH growing in open habitats, like those resulting from eradication efforts, produce more seeds than those in areas of dense cover (Luken and Thieret 1996). Flory and Clay (2005) found ABH density and germination success to be greatest in open habitats closer to roads and in early and mid-successional forests in central Indiana.

Removal of ABH has been demonstrated to have positive effects on native plant communities (McNeish and McEwan 2016). Trammel and Carreiro (2011) found ABH was the most important plant species explaining variation in the composition of woody plants in urban tracts in Louisville, KY. Presence of ABH is correlated with reduced species richness, herbaceous cover, and tree seedling density (Collier et al. 2002, Hutchinson and Vankat 1997, Shields et al. 2015). These studies found perennials to be the most resistant physiognomic group to ABH invasion, with reduced fecundity but not survival, attributed to storage of reserves in perenniating

organs (Miller and Gorchov 2004). The most sensitive group of plants was shade-intolerant or early season annuals (Gould and Gorchov 2000). Removal of ABH from invaded sites has been correlated with increased species richness and seedling density (Hartman and McCarty 2004, Runkle et al. 2007). These changes have been attributed to increased light availability (Gorchov and Trisel 2003, Luken et al. 1997, Runkle 2007) and increased germination from the seed bank due to soil disturbance created by removal activities (Gould and Gorchov 2000). O'Donnell et al. (2014) found pioneer herbs and graminoids, early successional species adapted to frequent disturbance, to dominate the seed bank of degraded floodplains in New South Wales, Australia. Likewise, the flora in our quadrats was dominated by woody perennials before treatment, but forbs, especially annuals, increased significantly as percentages of the flora after 5 years of ABH removal.

Functional traits of plants established following Amur Bush Honeysuckle removal

Trend analysis of functional trait states revealed further changes in the flora between 2012 and 2017, with shifts toward more species with unassisted dispersal or transported externally by animals. Humans may be prime dispersal agents of seeds along Fall Creek. Spread of seeds lacking specialized dispersal mechanisms may be enhanced by movement of people within the habitat, and clothing and equipment may serve as vectors for seed dispersal (Drayton and Primack 1996, Pickering et al. 2011)

Over the course of this study, we documented an increase in the frequency and cover of some invasive species already present along Fall Creek before removal targeted at ABH began, as well as the first appearance of other invasive plants. Increases in the presence and cover of non-target invasive species are a common unintended consequence of restoration efforts (Kettenring and Adams 2011). Opening up of the shrub-layer canopy to more sunlight following ABH removal increases vulnerability of the habitat to invasive species (Hutchinson and Vankat 1997, Pavlovic and Leicht-Young 2011). Removal of invasive species creates empty niches and may trigger germination from seed banks that tend to be dominated by non-native species (Giora et al. 2012). This consideration may explain the large increase in cover of Winter-creeper (Swearingen and Bargerson 2016) during the course of our study, but the invasion dynamics of this woody vine are not well documented (Bauer and Reynolds 2016, Mattingly et al. 2016). Winter-creeper is used as an ornamental ground cover. It spreads by vegetative fragmentation, facilitated by flooding, and by seed when allowed to grow up trees and flower (Bender 2007, The Nature Conservancy 2018). Dolan and Moore (2017) found increases in Winter-creeper in a nature preserve upstream on Fall Creek over the course of the last decade and at other locations in Indianapolis (data not presented), paralleling the results reported here.

Invasive species not already present before removal efforts began in 2012 but appearing in quadrats over the course of the study are mostly woody. Dolan et al. (2011) identified woody plants, especially shrubs escaped from cultivation, as the

largest physiognomic class of invasives added to the flora of Indianapolis over the last 70 years. Of particular concern along Fall Creek is the addition to the flora of *Euonymus alatus* (Thunb.) Siebold (Winged Euonymus) and *Pyrus calleryana* Decne. (Callery Pear). These taxa were recently identified as among the top 5 emerging invasive species of concern in Indianapolis (Dolan 2016).

Changes in habitat quality based on floristic quality assessment and Shannon diversity

Sites with high natural area quality in Indiana are expected to have FQI values of 35 or greater (Rothrock and Homoya 2004). Locations like Fall Creek that are actively undergoing restoration would be expected to have much lower values. The absolute value, however, is not as important as how the number changes through time, with an increase indicating better site quality from an ecological perspective. Floristic quality index values for Fall Creek changed little following ABH removal, even though Shannon diversity increased and the total number of species in quadrats more than doubled. Habitats with more species are generally expected to be more stable and resilient to environmental perturbations like invasion than habitats with fewer species (Knops et al. 1999).

Although at the quadrat-level the percentage of native species increased twice as much as non-natives, the percentage of species overall at Fall Creek that are native did not change appreciably, and most new species are natives with low C-value. It may be that natural recolonization of the site by native plants with higher C-values will take more years to occur, perhaps due to depauperate native seed banks and limited dispersal into the site. Natural regeneration from the seed bank alone is thought to be insufficient for ecological restoration of a variety of plant community types in Europe (Bossuyt and Honnay 2008). Dispersal, not in situ germination, is needed for the reestablishment of later successional species in these communities. However, our functional trait analysis showed new plants dispersing into our Fall Creek sites were more likely to be wind dispersed than a random sample of the countywide flora would predict, and wind-dispersed species are likely more characteristic of early successional habitat (Walker and Chapin 1987). In addition, recent surveys of vegetation along Fall Creek and 2 other urban creeks in Indianapolis (Holland et al. 2017), identified 275 vascular plant species, 74% native, but these species had mean C-values of 2–3, with few high C-value species, suggesting local dispersal will be unlikely to establish higher C-value species in our sites.

It may take more time to detect vegetation quality changes following ABH removal along Fall Creek. The expected timeframe for vegetation recovery is not well documented. Runkle et al. (2007) used paired plots (removal and control) to study forest vegetation response to ABH removal in west-central Ohio. No differences were found after 1 year. Increases in richness, cover, and tree seedling density were detected when the same sites were sampled 7 to 8 years later, evidence of a multi-year lag period for vegetation response, as also reported by Luken et al. (1997). This lag period may be due to depauperate seed banks in areas long impacted by ABH, since the impact of invasive plants on seed banks generally increases with residence time (Giora et al. 2012). Alternatively, our assumption that we will be

able to detect improved habitat quality as indicated by FQA following removal of an invasive plant in this urban setting may be flawed. FQA is likely more suitable for tracking change in habitats that are less dynamic such as wetlands, prairie, and upland forest. It has been shown to be useful in longitudinal studies of prairie restorations (McIndoe et al. 2008) but less helpful in assessing restoration progress in natural old-field succession (Rothrock et al. 2011).

The riparian border of Fall Creek has been highly modified. It has been impacted by factors common to urban streams: land-use conversion and increased frequency and magnitude of flooding, resulting in increased downcutting due to high peak-flow rates (National Research Council 2002). Fall Creek has been further altered, possibly in more unique ways, by vegetation management related to parks and greenway plans in the early 1920s that included narrowing of its wooded border and planting of non-native species. It is not entirely clear what improved habitat quality would look like in this system or which reference site conditions should be targeted as restoration goals. The perpetual disequilibrium of urban ecosystems has led to their being dubbed novel ecosystems (Hobbs et al. 2009), with no cognates in natural areas. River and stream systems are naturally dynamic and non-equilibrium ecosystems (Richardson et al. 2007, Vosse et al. 2008). The combination of natural flood events and human-caused disturbance on the narrow linear habitat of urban creeks and streams—along with other negative effects of being urban, spatial isolation with lots of edge to area, and enhanced opportunity for contact with non-native species which act as agents of perturbation—promote early successional plant species. Establishment of species more characteristic of stable riparian borders here will likely require additional active restoration, as is now recognized to be necessary in riparian zones across the globe, along with continued monitoring and removal of invasive species (Giora et al. 2012, Vosse et al. 2008). A better understanding of the dynamics of riparian seed banks (Vosse et al. 2008), especially those related to long-term impacts of seed-bank impoverishment on vegetation dynamics and ecosystem function in sites formerly impacted by invasive species (Giora et al. 2012), would help inform this effort.

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