
Forest Restoration in Urbanizing Landscapes: Interactions Between Land Uses and Exotic Shrubs

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Abstract

Preventing and controlling exotic plants remains a key challenge in any ecological restoration, and most efforts are currently aimed at local scales. We combined local- and landscape-scale approaches to identify factors that were most closely associated with invasion of riparian forests by exotic shrubs (Amur honeysuckle [*Lonicera maackii*] and Tatarian honeysuckle [*L. tatarica*]) in Ohio, U.S.A. Twenty sites were selected in mature riparian forests along a rural–urban gradient (<1–47% urban land cover). Within each site, we measured percent cover of *Lonicera* spp. and native trees and shrubs, percent canopy cover, and facing edge aspect. We then developed 10 a priori models based on local- and landscape-level variables that we hypothesized would influence percent cover of *Lonicera* spp. within 25 m of the forest edge. To deter-

mine which of these models best fit the data, we used an information-theoretic approach and Akaike's information criterion. Percent cover of *Lonicera* was best explained by the proportion of urban land cover within 1 km of riparian forests. In particular, percent cover of *Lonicera* was greater in forests within more urban landscapes than in forests within rural landscapes. Results suggest that surrounding land uses influence invasion by exotic shrubs, and explicit consideration of land uses may improve our ability to predict or limit invasion. Moreover, identifying land uses that increase the risk of invasion may inform restoration efforts.

Key words: exotic shrubs, forest restoration, honeysuckle, land use, *Lonicera*, reserve design, urbanization.

Introduction

Exotic plants have encroached into many ecosystems and communities throughout the world, disrupting ecosystem properties, reducing native biodiversity, and altering hydrologic and fire regimes (Vitousek 1990; Vitousek et al. 1997; Mack et al. 2000). Exotic plants also threaten restoration efforts, and considerable monetary resources are spent preventing or controlling invasions (Hobbs & Humphries 1995; Hiebert 1997). Nevertheless, the mechanisms behind the invasion process are poorly understood. Several researchers have identified the characteristics of successful invaders (Baker 1986; Bazzaz 1986; Noble 1989; Rejmanek & Richardson 1996), but these characteristics offer little predictive power because no one trait can predict the success or failure of an invader (Noble 1989; Hobbs & Humphries 1995; Mack et al. 2000). Other researchers have examined the susceptibility of specific communities and ecosystems to invasion; however, their findings fail to consistently predict the vulnerability of specific sites (Baker 1986; Rejmanek 1989; Hobbs &

Humphries 1995; Mack et al. 2000). The difficulty may be, in part, due to the variation in landscape-scale factors that can influence invasion by exotic plants (With 2002). Landscape-scale factors such as surrounding land uses and amount of regional forest cover are known to be an important determinant of bird (Flather & Sauer 1996; Rodewald & Yahner 2001a, 2001b), mammal (Dijak & Thompson 2000), and plant (Hobbs 2000) communities. Fragmentation and changes in land use can alter ecosystems, providing opportunities for exotic species to invade via increased disturbance and increased movement of seeds (Hobbs & Huenneke 1992; Hobbs 2000; Trombulak & Frissell 2000; Luken & Spaeth 2002). Fragmentation also can increase available light near recently created edges, providing suitable microsites for light-demanding exotic plants to establish (Brothers & Spingarn 1992; Matlack 1993). Few studies, however, have specifically investigated the role of surrounding land uses and, in particular, how the degree of urbanization (residential and commercial) affects invasion (Pysek et al. 2002). Instead, investigators have inferred landscape associations based on subjective categorization (Moran 1984), past herbarium collections (Mack 1981), historical records (Luken & Thieret 1996), or aerial photos (Hutchinson & Vankat 1998).

We examined the extent to which invasion of riparian forests by *Lonicera maackii* and *L. tatarica* was associated with local (within site) and landscape (land use within 1 km) factors. Specifically, we expected the vulnerability of

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a forest to *Lonicera* invasion to increase with (1) landscape-scale disturbances created by urbanization and forest fragmentation (Hobbs 2000); (2) small-scale disturbances within forests, such as human trails (Lonsdale 1999); and (3) certain microsite characteristics, such as south- and west-facing edges and open canopies (Brothers & Spingarn 1992; Luken & Goessling 1995). We chose *Lonicera* as a focal group because it (1) successfully invades forests, disrupting native plant establishment (Woods 1993; Trisel & Gorchov 1994; Hutchinson & Vankat 1997; Collier et al. 2002) and (2) is a management concern (Luken & Thieret 1996; Hutchinson & Vankat 1997; Deering & Vankat 1999; Collier et al. 2002), especially in urban landscapes. Although others have suggested that surrounding land uses may influence the proportion of *Lonicera* spp. present within forest patches (Luken & Thieret 1996; Hutchinson & Vankat 1997), few studies have specifically examined the influence of land use on invasion by *Lonicera* spp. Understanding how land uses influence invasion will greatly improve our ability to identify areas where restoration efforts will likely succeed.

Methods

Study Area

We studied invasion of *Lonicera* spp. in riparian forests within the Scioto River Watershed situated in the Till Plains physiographic region of Ohio (lat 40°0'N, long 83°0'W). Sites were located within Franklin and Delaware counties on publicly and privately owned lands. Land cover within these two counties was primarily agriculture (44%) and urban/residential development (41%); only 8.5% of the land cover within these two counties was forested (United States Geological Survey (USGS) Earth Resources Observation and Science (EROS) Data Center 2000). Remnant forests persisted mainly in riparian areas (Laub 1979; Groom & Grubb 2002) and were generally dominated by Boxelder (*Acer negundo*), American elm (*Ulmus americana*), Ohio buckeye (*Aesculus glabra*), Sugar maple (*Acer saccharum*), and Cottonwood (*Populus deltoides*). Dominant shrubs included Amur honeysuckle (*Lonicera maackii*), Tatarian honeysuckle (*L. tatarica*), Multiflora rose (*Rosa multiflora*), Spicebush (*Lindera benzoin*), and Hawthorn (*Crataegus* spp.).

Potential riparian forest study sites (approximately >40 m wide and >250 m long) within the watershed were identified from digital orthophoto quadrangle images (DOQ) (USGS DOQ 1994–1995, 1:24,000) and detailed maps of Franklin and Delaware counties. From these we selected 20 sites where access was allowed and that met the following criteria: (1) mature riparian forest; (2) 40 m or greater in width; (3) 250 m or greater in length; (4) distance of 2 km or more between sites; (5) negligible slope (<5%); and (6) river width of approximately 20–40 m. These sites represented a sample of riparian habitat within a 35-km radius of Columbus, Ohio.

Vegetation Sampling

Percent cover was sampled using a modified line point method (Rotenberry & Wiens 1980; Wiens & Rotenberry 1981; Barbour et al. 1987) along three transects separated by 50 m within each site. Various-length transects were randomly placed perpendicular to the forest edge and extended from the forest edge to the river edge. We defined an edge as the center of the outermost canopy tree (Brothers & Spingarn 1992). We sampled vegetation volume from 0 to 4 m above the ground in 0.5-m intervals using an extendable aluminum pole positioned at successive 5-m points along each transect. At each of these points, we recorded the presence of each species touching the pole within each 0.5-m height interval (Rotenberry & Wiens 1980; Wiens & Rotenberry 1981). All tree and shrub species contacting the pole were identified to species, whereas all other vegetation hits were categorized as forb, grass, or vine. Percent cover was calculated as the total number of hits in each layer at all sampling points (Rotenberry & Wiens 1980; Wiens & Rotenberry 1981). We then averaged estimates over all three transects to obtain percent cover values per site.

Local- and Landscape-Scale Variables

Local-scale variables (within a site) measured included density of human trails, facing edge aspect (North/East [N/E] vs. South/West [S/W]), and percent canopy cover. Assessment of relative trail density within each site was determined by counting the number of times a trail intersected each vegetation transect. Canopy cover was recorded every 5 m along each transect using an ocular tube (James & Shugart 1970). In addition, we recorded the type of adjacent edge (e.g., urban park or agricultural field), noted any unusual site characteristics, and documented each edge with a photograph (35 mm).

We calculated the percentage of land area containing urban, forest, and agricultural cover types within a 1-km radius for each site using Thematic Mapper Imagery data and ArcView geographic information software (Environmental Systems Research Institute, ESRI, Redlands, CA, U.S.A.) (Table 1). Land covered by pavement or buildings was classified as urban development, and these urban areas consisted primarily of residential and commercial development. Open (nonforested land lacking wetlands or water) vegetation was classified as agriculture. The amount of forest within the landscape was not confounded with the amount of urban land cover, because the two were unrelated ($r = -0.075$, $p = 0.75$).

Data Analysis

Exotic shrubs in the 20 riparian forest sites sampled consisted mainly of *L. maackii* and *L. tatarica*. Because *L. maackii* and *L. tatarica* (hereafter *Lonicera*) both rely on frugivores for dispersal, exhibit growth in shaded and sunny environments, and tend to dominate edge and intact

Table 1. Percent cover (\pm SE) of *Lonicera* spp. and native trees and shrubs within 25 m of the edge, relative trail density, facing edge aspect, and percent land use covered by urban development, agriculture, and forest within 1 km of 20 riparian forest sites in central Ohio, U.S.A.

Site	<i>Lonicera</i> Cover (%)	Native Tree and Shrub Cover (%)	Relative Trail Density	Canopy Cover (%)	Facing Edge Aspect	Urban (%)	Agriculture (%)	Forest (%)
Big Walnut Park	8.00 \pm 0.00	12.33 \pm 2.19	21.00 \pm 0.01	85.00 \pm 0.08	S/W	46	16	37
Casto Park	9.67 \pm 5.78	4.67 \pm 0.88	18.00 \pm 0.00	83.00 \pm 0.07	S/W	47	27	25
Cherrybottom Park	0.00 \pm 0.00	2.50 \pm 0.50	4.00 \pm 0.01	87.00 \pm 0.03	S/W	31	35	29
Darby Public Hunting	0.00 \pm 0.00	3.67 \pm 0.88	0.00 \pm 0.00	86.00 \pm 0.04	N/E	1	58	40
Elkrun Park	10.50 \pm 7.50	8.33 \pm 1.33	8.00 \pm 0.05	95.00 \pm 0.04	S/W	13	67	18
Galena	0.00 \pm 0.00	8.67 \pm 1.20	0.00 \pm 0.00	94.00 \pm 0.02	S/W	1	42	43
Gardner Road	0.00 \pm 0.00	1.00 \pm 0.00	0.00 \pm 0.00	82.00 \pm 0.03	N/E	1	54	43
Innis Park	0.00 \pm 0.00	11.67 \pm 5.17	18.00 \pm 0.04	96.00 \pm 0.04	N/E	8	40	51
Kilbourne	0.00 \pm 0.00	4.67 \pm 0.88	8.00 \pm 0.01	83.00 \pm 0.05	N/E	<1	63	36
Lockbourne Park	0.00 \pm 0.00	10.33 \pm 5.36	7.00 \pm 0.02	95.00 \pm 0.03	S/W	3	69	24
North Galena	0.00 \pm 0.00	11.67 \pm 4.37	0.00 \pm 0.00	95.00 \pm 0.05	N/E	<1	46	53
North Olentangy	17.00 \pm 6.00	3.00 \pm 0.00	22.00 \pm 0.02	88.00 \pm 0.03	N/E	29	23	45
Prairie Oaks	6.50 \pm 0.50	17.00 \pm 4.73	13.00 \pm 0.05	91.00 \pm 0.05	N/E	1	65	30
Prindle	0.00 \pm 0.00	8.00 \pm 2.00	0.00 \pm 0.00	83.00 \pm 0.04	S/W	<1	84	13
Smith Farm	0.00 \pm 0.00	3.00 \pm 1.00	0.00 \pm 0.00	96.00 \pm 0.02	N/E	15	71	61
South Galena	0.00 \pm 0.00	12.00 \pm 4.04	1.00 \pm 0.01	94.00 \pm 0.04	N/E	<1	50	12
Three Creeks	22.50 \pm 6.50	7.33 \pm 4.91	7.00 \pm 0.03	87.00 \pm 0.03	S/W	8	73	15
Whetstone Park	8.67 \pm 1.67	8.00 \pm 4.58	18.00 \pm 0.04	78.00 \pm 0.10	N/E	31	15	48
Whitehall Park	0.00 \pm 0.00	10.33 \pm 4.33	14.00 \pm 0.01	89.00 \pm 0.02	S/W	35	34	50
Woodside Green Park	0.00 \pm 0.00	9.67 \pm 1.33	10.00 \pm 0.03	97.00 \pm 0.02	N/E	18	48	29

forest patches, we grouped these two species in our analysis (Woods 1993; Luken et al. 1995). We developed a set of a priori models based on previous studies using percent cover of *Lonicera* as a dependent variable. These models incorporated landscape- and local-level independent variables as well as interactions between variables. Initial models were developed to include variables that we hypothesized would influence *Lonicera* cover. We hypothesized that edge characteristics likely play a role in the proportion of *Lonicera* present within a site and therefore only analyzed data collected 0–25 m from the forest edge. To reduce the number of initial model statements, we removed variables that were redundant or strongly correlated ($r > 0.70$) with other terms and those that lacked adequate variation across sites. To examine the nature of the relationship between *Lonicera* and native trees and shrubs, we conducted a post hoc correlation analysis of the two variables within 25 m of the edge.

We used an information-theoretic approach (Akaike's information criterion [AIC]) (Burnham & Anderson 1998) to determine which candidate model(s) was best supported by the data. The information-theoretic approach provides a strength of evidence for a set of a priori models based on AIC values that are generated from log-likelihood values and do not rely on null hypothesis testing (Burnham & Anderson 1998; Anderson et al. 2000). We calculated AIC and bias-corrected AIC_c (Burnham & Anderson 1998) from log-likelihood values generated using the GENMOD procedure (SAS Institute 1990). Bias-corrected AIC_c values adjust for small sample sizes relative to the number of parameters within model statements. Because the distribution of *Lonicera* cover fit a negative binomial distribution

(i.e., the variance was greater than the mean; Fowler & Cohen 1990), we specified the negative binomial distribution in GENMOD. Differences in AIC_c values, delta AIC (Δ_i), and Akaike weights (w_i) among models were used to identify the model that was best supported by our data. Models with large Δ_i values are less plausible given the data, and w_i provide an additional measure of strength of evidence for a model (Burnham & Anderson 1998).

Results

Relative trail density at each site was positively correlated with the proportion of urbanization within 1 km ($r = 0.70$, $p < 0.001$), and therefore, we removed trail density from the analysis. The proportion of agriculture in the landscape was negatively correlated with the proportion of urban land cover ($r = -0.75$, $p < 0.001$) and also was removed from subsequent analyses because our focus was on urbanization. Because canopy cover showed little variation among sites (range = 78–97%, mean = 89.32 \pm 5.78 SD), it was not included in model statements. In total, 10 candidate models were created for *Lonicera* vegetation using the variables proportion of urban land cover within 1 km, proportion of forest cover within 1 km, edge aspect (N/E vs. S/W), and interactions between variables (Tables 1 & 2). Of these 10 candidate models, Δ_i and w_i identified the model containing percent urban land cover within 1 km as the model best describing *Lonicera* cover within riparian forests (Table 2). Percent cover of *Lonicera* was positively associated with the proportion of urban land cover within the landscape. Although four models were

Table 2. Model statements describing percent cover of *Lonicera* in 20 riparian forests in central Ohio.

Model Statements	Log Likelihood	K	AIC _c	Δ _i	w _i
Percent <i>Lonicera</i> cover = urban ^a	104.80	3	-202.09	0.00	0.32
Percent <i>Lonicera</i> cover = forest ^b	104.46	3	-201.43	0.68	0.23
Percent <i>Lonicera</i> cover = urban forest	107.82	5	-201.35	0.75	0.22
Percent <i>Lonicera</i> cover = urban and forest	105.74	4	-200.82	1.28	0.17
Percent <i>Lonicera</i> cover = aspect ^c	104.48	4	-198.30	3.79	0.05
Percent <i>Lonicera</i> cover = urban and aspect	105.03	5	-195.78	6.32	0.01
Percent <i>Lonicera</i> cover = forest and aspect	104.49	5	-194.69	7.40	<0.01
Percent <i>Lonicera</i> cover = urban and forest and aspect	106.42	7	-189.51	12.59	<0.01
Percent <i>Lonicera</i> cover = aspect urban	105.50	7	-187.67	14.42	<0.01
Percent <i>Lonicera</i> cover = aspect forest	104.84	7	-186.34	15.75	<0.01

Models are ranked according to AIC_c and delta AIC (Δ_i) scores. Best-supported models have smaller delta AIC (Δ_i) scores and larger Akaike weights (w_i). Vertical bars represent both main effects and interactions between variables. Model statements with two or more variables combined with "and" represent main effects only.

^aPercent urban land cover within 1 km of each site.

^bPercent forest cover within 1 km of each site.

^cFacing edge orientation (N/E or S/W).

closely ranked with Δ_i less than 2 (Table 2), the sums of the w_i for all models containing the urban variable suggest that the most important variable among those tested was the percentage of urban land cover (summed w_i for urban = 0.72). The remaining closely ranked models suggest that the proportion of forest cover within 1 km and the interactions between forest cover and urban land cover may play a role in determining the proportion of *Lonicera* present at a site. Forest patches with reduced forest cover and higher levels of urbanization contained more *Lonicera* relative to sites in more forested and less urbanized landscapes.

We did not find a relationship between *Lonicera* cover and native tree and shrub cover. Percent cover of native trees and shrubs less than 4 m in height was not correlated with the percentage of *Lonicera* cover at a site ($r = -0.02$, $n = 20$, $p = 0.92$).

Discussion

We found that the proportion of *Lonicera* present along forest edges was best explained by landscape-scale factors. Specifically, the proportion of urban land cover within 1 km of riparian forests was positively associated with the percentage of *Lonicera* cover. Several other studies also suggest that the risk of invasion increases for forests near urban areas (Moran 1984; Nilsson et al. 1989; Timmins & Williams 1991; Luken & Thieret 1996; Planty-Tabacchi et al. 1996; Hutchinson & Vankat 1997; Rottenborn 1997; Pysek et al. 2002). For example, Hutchinson and Vankat (1997) found that *Lonicera maackii* cover was negatively correlated to the distance from Oxford, Ohio. Luken and Thieret (1996) also suggested that *Lonicera* populations occur most often in urban landscapes. Although these studies suggested that urban areas contain higher densities of exotic plants, composition of the matrix or surrounding land uses were not explicitly tested. An important distinction between our study and others is that we

quantified the degree of urbanization along a gradient from areas with less than 1% to greater than 40% development (residential and commercial). Examining the extent of invasion along a rural to urban gradient provides useful information to help guide the decision-making process, which may ultimately help ecologists predict the success of restoration efforts based on landscape-scale factors (McDonnell & Pickett 1990; Marzluff et al. 2001; With 2004).

Highly disturbed landscapes are at a higher risk of invasion because fragmentation and urban development increase disturbance which, in turn, provides ideal microsites for the establishment of exotic plants (Hobbs 1989; Hobbs & Huenneke 1992). Landscape configuration and proximity of a forest patch to a seed source also greatly increases dispersal probabilities and consequently influence the risk of invasion (Rejmanek 1989; Matlack 1994; Richardson et al. 2000; With 2002, 2004). For example, forests adjacent to urban areas have a higher risk of invasion because developed lands often maintain high densities of exotic ornamental shrubs that provide ample seed sources. Forest patches situated in fragmented landscapes are also at risk of invasion, and the degree of invasion may be linked to the proportion of the landscape that is disturbed (With 2004). In fact, we found that *Lonicera* cover was negatively related to the proportion of forest cover within the landscape. Fragmentation of forest patches also can increase invasion risk, particularly if dispersal of exotic plants is enhanced along forest edges (With 2004). Dispersal may be enhanced along edges because similar landscape-scale factors (e.g., edges) also influence the distribution of avian seed dispersers and subsequent dispersal patterns (Debussche & Isenmann 1990; Rose 1997; Renne et al. 2002). In our study area, Northern Cardinals (*Cardinalis cardinalis*) and American Robins (*Turdus migratorius*), which are often associated with edge habitats (Halkin & Linville 1999; Sallabanks & James 1999), were observed foraging on the fruits of both *L. tatarica* and *L. maackii* (K. Borgmann 2001, 2002, Ohio

State University, personal observation). Thus, forest patches situated in urban landscapes with reduced forest cover may be at a particularly high risk of invasion.

Unlike other studies (Brothers & Spingarn 1992; Matlack 1993; Luken & Goessling 1995; Luken et al. 1995), we found no evidence that several local-scale factors, such as edge aspect, influenced invasion of riparian forests by *Lonicera*. Light availability is often a factor limiting exotic plant establishment (but see Harrington et al. 1989), and edge aspect and canopy cover are often used as indicators of light availability. Light penetration into forests is typically higher along S/W edges than along N/E edges; however, edge aspect did not influence the proportion of *Lonicera* present along forest margins. Although we did not quantify light availability (e.g., Luken & Goessling 1995) or include canopy cover in analysis, light may not be a major factor determining the presence of exotic plants at our sites because (1) extended leaf phenology of *L. maackii* and *L. tatarica* allows the plants to increase photosynthetic activity during spring and fall when canopy cover is reduced (Harrington et al. 1989; Woods 1993; Trisel & Gorchoy 1994); (2) *L. maackii* exhibits plasticity in response to light (Luken et al. 1995); (3) *L. tatarica* is shade tolerant (Woods 1993); and (4) both species are capable of invading intact forest patches (Woods 1993; Luken & Goessling 1995).

As a whole, our study illustrates the importance of considering the landscape in ecological restorations. Many previous studies have focused solely on local-scale mechanisms that affect exotic plant invasion and persistence (Luken 1988; Brothers & Spingarn 1992; Matlack 1993; Luken & Goessling 1995), and oftentimes restorations are guided primarily by these local environmental constraints, overlooking landscape influences. Our results, however, suggest that land uses surrounding sites may affect the success of restoration efforts by determining, to some extent, the vulnerability to invasion by exotic plants and the availability of native seed sources. In our study system, urban development and reduced forest cover seemed to promote the invasion of *Lonicera* within riparian forests, although the proximate mechanism was not identified. Although *Lonicera* is often negatively related to native vegetation at microsite scales (e.g., Collier et al. 2002; Gorchoy & Trisel 2003; Miller & Gorchoy 2004), we found no relationship between *Lonicera* and native vegetation at the stand level, which further suggests that restoration potential may be tightly linked to landscape-mediated factors.

Incorporating landscape-scale features is a critical component in understanding the invasion process, which may increase our ability to predict or prevent future invasions (Hobbs & Humphries 1995; Ewel et al. 1999; Marvier et al. 2004; With 2004). Moreover, because restoration efforts that rely on eradication of exotic species have proven to be both laborious and economically challenging, efforts may be better directed toward areas where restoration will likely succeed. Once land uses that facilitate invasion are identified, biologists and managers may be

able to minimize the impacts of exotic plants via local restoration efforts and/or strategic reserve design and planning (Luken 1997; Byers et al. 2002; With 2002). For example, strategically placed reserves (e.g., near old-growth or older second-growth stands) may increase the potential for native plant recolonization and survival (Matlack 1994; Pysek et al. 2002; Jacquemyn et al. 2003; Marvier et al. 2004; With 2004). Furthermore, promoting low-development buffer zones in areas surrounding reserves may reduce the risk of invasion by ornamental exotic plants (Hobbs & Humphries 1995; Pysek et al. 2002). We suggest that explicit consideration of landscape-scale factors in both restoration and reserve-design plans may substantially improve conservation outcomes, particularly as related to the risk of *Lonicera* invasion.

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