Available online at www.sciencedirect.com

Forest Ecology and Management 190 (2004) 119–130

Forest Ecology and Management

www.elsevier.com/locate/foreco

Recruitment of three non-native invasive plants into a fragmented forest in southern Illinois

Emily D. Yates[a], Delphis F. Levia Jr.[b,*], Carol L. Williams[a]

[a]Department of Geography, Southern Illinois University, Carbondale, IL 62901-4514, USA
[b]Department of Geography, Center for Climatic Research, University of Delaware, Newark, DE 19716, USA

Received 28 July 2003; received in revised form 10 September 2003; accepted 11 November 2003

Abstract

Plant invasions are a current threat to biodiversity conservation, second only to habitat loss and fragmentation. Density and heights of three invasive plants, *Rosa multiflora*, *Lonicera japonica*, and *Elaeagnus umbellata*, were examined between edges and adjacent interiors of forest sites in southern Illinois. Density (stems m^{-2}) and heights (cm) of invasive plants were obtained in plots along transects at edge and interior sampling locations within forest sites. The effect of species, sampling location, and site shape index on invasive plant density was investigated, as well as differences in heights of invasive plants in edge vs. interior sampling locations.

Species, sampling location, and fragment shape index were significant factors influencing invasive plant density at study sites. Density for all three species ranged from 0 to 18 stems m^{-2}. All three species invaded interiors of sites, however, *R. multiflora* and *L. japonica* had significantly greater densities in edge as opposed to interior transects. These two species also had significant differences in density among site shape indices. Density of *E. umbellata* was not significantly different between edge and interior sampling locations or among site shape indices. Mean heights of all three invasive plants were higher in edge transects, however, this relationship was only significant for *L. japonica*.

These findings suggest that (1) preservation of tracts of forest with less edge can minimize invasion by these three plants in this area. Interiors of forest sites can be protected from invasion by having less edge habitat for invasive species to enter and become established; and (2) *E. umbellata*, unlike the other two species, is established in both interior and edges of study sites. Therefore, *E. umbellata* may pose the most ecological damage because it is well established throughout sites, and should be given a high priority when implementing management and control efforts in southern Illinois forest ecosystems.

Keywords: Invasive plants; Forest fragmentation; *Rosa multiflora*; *Lonicera japonica*; *Elaeagnus umbellata*; Forest conservation

1. Introduction

Plant invasions are a major threat to biodiversity worldwide. There is growing concern that invasive species, as defined by Richardson et al. (2000), are one of the greatest threats to biodiversity conservation (Wilcove et al., 1986; Coblentz, 1990), second only to habitat loss and fragmentation (Lee and Macdonald, 1997). Both Drake et al. (1989) and Shigesada and Kawasaki (1997) noted that few ecosystems in the world are resistant to invasions by terrestrial plants. In Illinois, non-native potentially invasive plants constitute 811 species or 29% of plant species known to occur (Ebinger, 1983; Harty, 1986).

---

[*] Corresponding author. Tel.: +1-302-831-2295; fax: +1-302-831-6654.
E-mail address: dlevia@udel.edu (D.F. Levia Jr.).

0378-1127/$ – see front matter © 2003 Elsevier B.V. All rights reserved.
doi:10.1016/j.foreco.2003.11.008

Habitat loss by fragmentation of forest ecosystems is a threat to biodiversity as well (Wilcox and Murphy, 1985; Stork, 1997). Changes in microenvironments caused by the creation of forest edges have led to shifts in species composition at forest margins (Ranney et al., 1981; Lovejoy et al., 1986). Such microenvironmental factors include changes in light availability, wind speed, relative humidity, air temperature, and soil temperature and moisture between edges and interiors of forest fragments (Groom and Schumaker, 1993). Although fragmentation and an increase in edge can lead to an increase in total species diversity, this is not necessarily indicative of a rich native community. It is more likely due to the presence of many invasive species with low conservation value (Hobbs and Huenneke, 1992).

Numerous studies have investigated whether microenvironmental changes caused by creation of forest edges due to fragmentation have led to shifts in species composition at forest margins (Ranney et al., 1981; Lovejoy et al., 1986), but this work has emphasized loss of native forest species rather than introduction of invasive species. Changes in available light, wind speed, humidity, temperature, and soil moisture induced by fragmentation of forest ecosystems often add to competitive advantages of invasive species over native species (Brothers and Spingarn, 1992; Groom and Schumaker, 1993). Therefore, a connection exists between the study of forest fragmentation and the study of invasive species colonization.

Much research on forest fragmentation and associated impacts on biodiversity have emphasized tropical forest "hot spot" areas recognized for protection of rich native biodiversity (Douglas, 1998). However, both fragmentation and invasive plants pose a threat to broad scale temperate forest ecosystems, in addition to concentrated hot spots of rich biodiversity (Wilcove et al., 1986). The need to assess the effects of fragmentation and its influence on invasive plants in temperate ecosystems validates the study of non-native invasive plant species in forest fragments in temperate forest ecosystems.

Successful invasion depends not only on the characteristics of the invading species but also on the characteristics, dynamics, and history of the site that is invaded (Hobbs and Humphries, 1995; Meekins and McCarthy, 2001). Since controlling invasive species has high economic and environmental costs (Ebinger, 1983; Huxel, 1999; Mack et al., 2000), it is preferable to prevent establishment rather than relying on post-establishment control and eradication. Land managers and natural preserve managers would benefit from knowledge about which types of areas are more prone to invasion. This information would help eliminate costly invasions before they occur.

Four main plant species have been identified within our study area as problematic invaders: *Sassafras albidum* (sassafras), *Rosa multiflora* Thunb. (multiflora rose), *Lonicera japonica* Thunb. (Japanese honeysuckle), and *Elaeagnus umbellata* Thunb. (autumn olive) (Davis, 1987; Iverson et al., 1999). Iverson et al. (1999) classifies *S. albidum* as invasive, however, it is native to the area. The remaining three species are all woody invaders introduced from Asia that can dominate the shrub layer and forest floor, and are the focus of the present study. *R. multiflora* is extremely prolific and can form impenetrable thickets that exclude native plant species. When *L. japonica* alters understory forest communities, forest understory bird populations can be affected (Szafoni, 1990). Once *E. umbellata* is established it is highly invasive and difficult to control because plants will resprout vigorously when burned, cut, or mowed (Szafoni, 1990). The 1997 Illinois Exotic Weed Act designating it illegal to buy, sell, or distribute *R. multiflora* and *L. japonica*. Bratton (1982) and Ebinger (1983) described the effects that invasive plants had on natural areas in Illinois: (1) competition with and displacement of native species; (2) change in the biomass and productivity of a site; (3) changes in the nutrient balance of soil; (4) production of allelopathic chemicals; (5) modified disturbance regimes; and (6) changes in plant community structure.

Four hypotheses were tested concerning differences in density and height of the three invasive species at four sites in a fragmented southern Illinois forest. The following discussion of edge effects leads to our first two hypotheses.

Forest islands can be surrounded by a sea of potential invaders. Habitat disturbances, such as fragmentation, are almost always helpful, if not essential, to exotic plant invasions (Westman, 1990). Fragmentation can increase the invasion potential of a community by creating "invasion windows", or points of entry for invasive species, occurring more commonly along habitat edges (Hobbs and Huenneke, 1992;

Meekins and McCarthy, 2001). Fragments with irregular shapes (higher shape index) show more edge effects, and therefore are more likely to have greater occurrences in invasive plants, than circular fragments (Laurance and Yensen, 1991). Therefore, it is thought that invasive plant species will be more dense at forest edges and in sites that have more total edge habitat because of their irregular shapes.

Brothers and Spingarn (1992) observed that 37 of the 58 invasive plants censused in fragments were observed only in the edge transect. Invasive plants at the forest margin moved no more than a meter or two into forest interiors. Both Harrison (1999) and Luken and Goessling (1995) noted that invasive diversity decreased with increasing distance inland into fragments. This study investigated whether: (H_1) densities of the three invasive plant species will be greater at the edge vs. the interior of forest sites; and whether (H_2) sites with a higher shape index, or with irregular shapes (resulting in more edge relative to interior), will have greater densities of invasive plants.

Light availability, among other factors, is important for colonization by invasive plants (Hobbs and Huenneke, 1992). The creation of forest edges can result in increased light availability for understory plants (Groom and Schumaker, 1993; Cadenasso and Pickett, 2001), therefore causing forest edge to be favorable for many invasive plants. Increased availability of light and other essential resources may result in increased height of invasive plants at forest edges. Therefore, this study examines a third hypothesis, (H_3), that heights of the three invasive plants will be greater at the edge vs. the interior of forest sites.

Acceptance of the first three hypotheses support that sites having greater amounts of edge habitat and that are irregularly shaped are more susceptible to invasion because they are less able to sustain species diversity and less able to resist invasion. Such information would support conserving larger, more circular fragments when setting aside land for protection of biodiversity.

A final hypothesis explores possible differences in invasion for the three species in this study: (H_4) degree of invasion will vary among the three invasive plants studied. By investigating whether one species is more prevalent in forest fragments than the other species, useful information can be provided to land managers and natural preserve managers about which species require the most time, attention, and money for eradication, thus allowing the possibility of avoiding costly invasions before they occur. Researchers interested in predicting and explaining the vulnerability of forest communities to non-native species invasion may find the results of this study useful. By examining the relationship between fragment attributes and invasion by plant species, the links between two current human-induced threats to biodiversity can be validated.

2. Study area

2.1. Location, substrate, and climate

Field research was conducted in four forest patches within Touch of Nature Environmental Center (TNEC) (37.47°N, 89.15°W, 118 m above sea level) located within the purchase unit for Shawnee National Forest, approximately 8 miles south of Carbondale, IL, USA (Fig. 1). TNEC is located in the hilly, unglaciated section of southeastern Jackson and southwestern Williamson counties, part of the Greater Shawnee Hills section of the Interior Low Plateau Province, and it rests entirely on Pennsylvanian aged bedrock (Voigt and Mohlenbrock, 1964), covered by Quarternary deposits of clay and sand (Davis, 1987). The majority of the upland forested regions of the study site consists of Hosmer complex soil (Davis, 1987), which has historically been cultivated. Disturbance and fragmentation of original forests and prairie in this area were traditionally for agricultural clearing. Exotic invaders such as *E. umbellata*, and *R. multiflora* are common on the successional fields of abandoned farmlands on Hosmer soils (Davis, 1987).

Mean annual temperature is approximately 13 °C with summer temperatures ranging from 16 to 30 °C and winter temperatures from −4 to 8 °C (National Climatic Data Center, 1961–1990). Fifty-five percent of the 112 cm of mean annual precipitation falls during April–September (Herman et al., 1979). Mean annual snowfall is 25 cm (National Climatic Data Center, 1961–1990).

2.2. Landcover

According to original land survey records in 1975, this region was completely forested with the

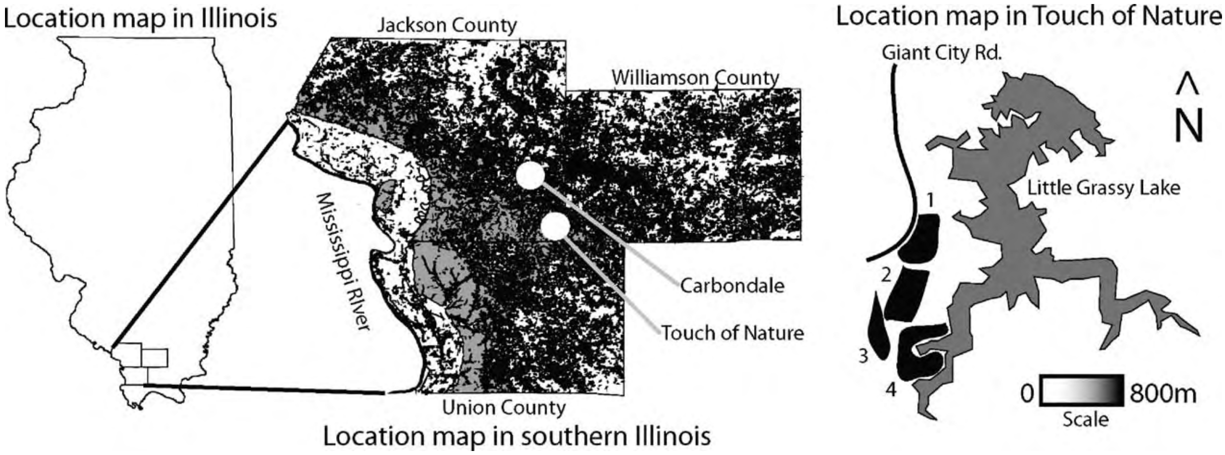

Fig. 1. Touch of Nature Environmental Center (TNEC), Jackson County, Illinois, USA with locations of the four forest sites.

Williamson County portion representing the county's greatest concentration of dense forests (Anderson and Anderson, 1975). Dominant species include *Quercus alba* L. (white oak), *Q. velutina* Lam. (black oak), *Carya* spp. (hickory), and *Q. rubra* L. (northern red oak). Before being preserved for University purposes in 1949, some of the forests within TNEC were cleared for cropland and livestock pasture.

Ashman (1962) documented areas within TNEC pioneered by the secondary successional tree species *S. albidum* Thunb. (sassafras) and *L. japonica* (Japanese honeysuckle). In some areas within TNEC, these species dominate the landscape along with the introduced opportunist *E. umbellata* (Davis, 1987). A preserved tract of land such as TNEC was chosen for this study to illustrate the increasingly apparent tendency of protected land to be invaded by invasive plant species (Usher et al., 1988).

3. Methods

3.1. Sampling

Four forest sites, ranging in area from 26 to 47 ha, were chosen with a distinct southern edge roughly parallel to an east-west transect. Forest sites were at least 100 m from nearby forest communities, with the exception of the adjacent borders of site 2 and 3. Cleared or semi-open grassy pastures, as well as road corridors and power line clearings delineate the majority of site edges in this study, while pastures, clearings and forested areas extend beyond some site edges. Sites were assigned numbers 1–4, from north to south. Areas and perimeters of each site were calculated using ArcView 3.0 (ESRI, 2002) (Table 1).

Two 40 m transects were placed within each site (Fig. 2). Transects were divided into ten 16 m^2 (4 m × 4 m) plots for sampling, which is standard for measuring woody shrub vegetation within a forest canopy (Mueller-Dombois and Ellenberg, 1974; Krebs, 1989). One transect was in the forest edge microenvironment, within 1 m of the site boundary, along a relatively linear edge section of at least 40 m, and as near to the midpoint of the southern boundary as possible. Southern edges were selected to represent edge habitat because they exhibit enhanced edge effects due to greater duration of sun exposure and increased ground level solar radiation (Palik and Murphy, 1990; Murcia, 1995; Williams-Linera et al., 1998). Another transect was located in the geometric center of each site under relatively closed canopy, parallel to the edge transect (Fig. 2). These interior

Table 1
Area, perimeter, and shape index of TNEC sites

Site	Area (ha)	Perimeter (km)	Shape index
1	25.9	1.9	1.05
2	36.3	2.4	1.13
3	28.5	2.7	1.41
4	46.6	4.2	1.90

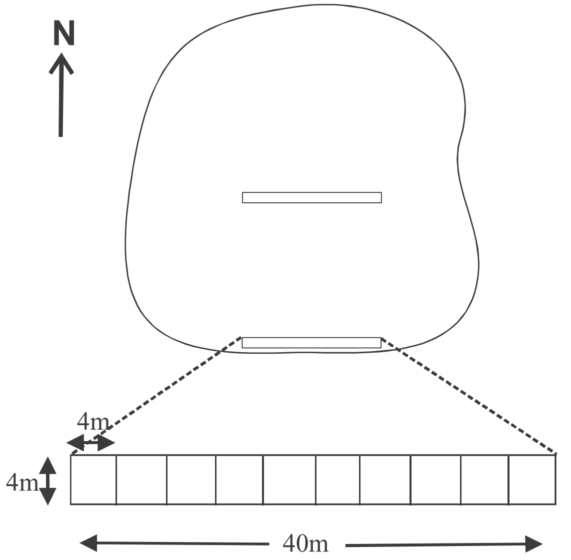

Fig. 2. Hypothetical site showing experimental design. In each of the four study sites a transect was located along the southern edge and the center of the site. The two transects were aligned parallel to each other. The east-west transects consist of ten 16 m^2 contiguous plots for sampling.

transects were well beyond a 50 m threshold to avoid forest edge characteristics (Ranney et al., 1981; Palik and Murphy, 1990). Density (stems m^{-2}) and heights (cm) of *R. multiflora*, *L. japonica*, and *E. umbellata*, were measured from late June to late August 2002, in 16 m^2 plots along each 40 m transect.

3.2. Data analysis

Three-way general linear model (GLM) analysis of variance (ANOVA) was used to assess the variations in species characteristics, sampling location, and site shape on density of the three invasive plants in fragmented forest sites. Since the four sites were approximately equal in size (Table 1), impacts of size were not included.

To determine whether each of the three species of invasive plants were individually affected by sampling location and site shape, a two-way ANOVA was performed for each species. *t*-Tests for independent samples were used to compare mean heights of each invasive plant in edge vs. interior transect sampling locations within forest fragments. Plant densities and heights were transformed with natural logarithms to meet assumptions of normality and homogeneity. All statistical analyses were performed using Statistica (StatSoft Inc., 1995) and SPSS (1999) software.

Because shape, as well as size, is important and should be accounted for when accessing edge effects at forest sites (Game, 1980; Schonewald-Cox and Bayless, 1986; Laurance and Yensen, 1991), a shape index (SI) was calculated for each fragment (Table 1). Patton (1975) developed a simple shape index (SI) that describes the deviation of fragment shapes from circularity, and provides a means to compare different fragment shapes. The formula adapted for metric units is: $\mathrm{SI} = P/200[(\pi A)^{1/2}]$, where P is the fragment perimeter and A the area (Laurance and Yensen, 1991). Circular fragments will have an SI value of 1.0, while more complex shapes produce higher SI values (Game, 1980). Fragments with irregular shapes, or higher SI values, tend to show more edge effects than do circular fragments (Laurance and Yensen, 1991).

Interior transects were compared to edge transects to test the hypothesis that edge habitat promotes greater densities of invasive species than interior transects (H_1), as well as the hypothesis that invasive species will have greater heights in edge vs. interior transects (H_2). Site shape indices were compared to test the hypothesis that more irregularly shaped sites (those with greater edge to interior ratios) have greater densities of each invasive species (H_3). Comparisons were also made between species across transects, to address the question of whether one species was more prevalent than another (H_4).

4. Results

4.1. Density

Invasive plant species density ranged from 0 to 18 stems m^{-2} for all three species, with the largest density, 18 stems m^{-2}, for *L. japonica* in the edge transect of site 2. Mean density of all species was 1.84 stems $m^{-2} \pm 3.6$ (1 S.D.). Mean densities of the three invasive species decline from edge to interior (Table 2).

Density of the three invasive plants declined between forest edge and interior habitats (Fig. 3). Mean *R. multiflora* densities decreased from 0.95 stems m^{-2} at edges to 0.14 stems m^{-2} in interiors. Mean

Table 2
Mean densities (stems m^{-2}) of the three invasive plant species in TNEC sites

Site	*R. multiflora* (stems m^{-2})		*L. japonica* (stems m^{-2})		*E. umbellata* (stems m^{-2})		Total invasive plant density (stems m^{-2})		Total
	Edge	Interior	Edge	Interior	Edge	Interior	Edge	Interior	
1	0.13	0.10	0	2.43	0.90	0.38	0.34	0.97	0.65
2	1.65	0	14.10	0	0.95	0.15	5.57	0.05	2.81
3	0.40	0.15	8.68	2.15	0.53	0.40	3.20	0.90	2.05
4	1.63	0.33	5.73	2.60	0.98	0.65	2.78	1.19	1.98
All	0.95	0.14	7.13	1.79	0.84	0.39	2.97	0.78	

L. japonica density decreased from 7.13 stems m^{-2} at edges to 1.79 stems m^{-2} in interiors, and *E. umbellata* density decreased from 0.84 stems m^{-2} at edges to 0.39 stems m^{-2} in interiors.

Both mean species counts and density indicate that site 2 illustrated that edge transects revealed the presence of more invasive species than interior transects (Fig. 4). Sites 3 and 4 have the same, yet less pronounced, relationship. Alternatively, site 1 had more invasive species in the interior transect. In site 2, only *E. umbellata* had a relatively low density within the interior transect (0.15 stems m^{-2}). The other two invasive species were not in any interior site 2 plots; this suggests that these three species have not invaded beyond the edge environment.

The overall mean density of invasive plants per fragment was calculated as X, the mean density of invasive plants in both the edge and interior transects. Overall mean density of the invasive species were 0.66, 2.81, 2.05, and 1.99, for sites 1–4, respectively. It would be expected that the fragment with the greatest area or lowest shape index (SI) would have the least non-native species. Site 1 (lowest SI) and site 4 (greatest area) had the lowest overall mean density of invasive species (0.66 and 1.99 stems m^{-2}, respectively) (Fig. 3). However, site 2, with the second lowest shape index, and the second largest area, had the highest density of invasive species.

A three-way ANOVA revealed that species, sampling location (edge vs. interior), and fragment shape index were significant factors influencing invasive plant density in the forest sites at TNEC (Table 3). Species was a significant ($P = 0.000$) factor in determining invasive species density. Density of invasive species was dependent on which species was present at the site. *L. japonica* had the highest density in both interior and edge transects. *R. multiflora* and *E. umbellata*, had similar, but lower overall densities (Table 2). The three invasive plants studied appear not to be equally prevalent at any given site.

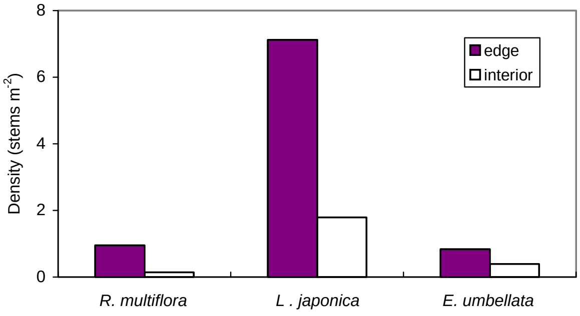

Fig. 3. Mean density of the three invasive plant species in edge and interior transects in TNEC study sites. Mean density was significantly different in edge vs. interior sampling locations for *R. multiflora* ($P = 0.013$) and *L. japonica* ($P = 0.000$), but not for *E. umbellata* ($P = 0.205$).

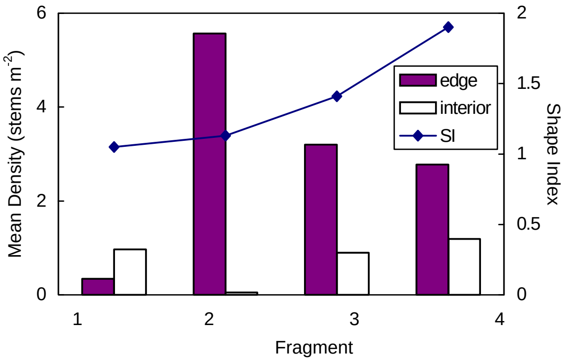

Fig. 4. Mean density of all invasive plant species in edge and interior transects of TNEC study sites as a function of site and SI.

Table 3
Results of ANOVA for affects of species, sampling location, and shape index on invasive plant density

	SS	d.f.	MS	F	P-value
Corrected model	133.64	20	6.68	14.28	0.000
Intercept	28.97	1	28.97	61.92	0.000
Species	98.75	2	49.38	105.54	0.000*
E vs. I	16.80	1	16.80	35.92	0.000*
SI	4.53	3	1.51	3.23	0.025*
Species × E vs. I	4.66	2	2.33	4.98	0.008*
Species × SI	7.84	6	1.31	2.79	0.014*
E vs. I × SI	0.24	3	8.1×10^{-2}	0.17	0.914
Species × E vs. I × SI	0.82	3	0.27	0.58	0.628
Error	54.73	117	0.47		
Total	217.34	138			
Corrected total	188.38	137			

* Denotes significant P-values.

Invasive plant sampling location, edge transect opposed to interior transect ($P = 0.000$) as well as shape index ($P = 0.025$), significantly affected density of invasive plants (Table 3). There were significant differences in overall invasive plant densities between edge and interior transects (H_1). Invasive plant densities were also significantly different at all sites due to differences in shape index (H_3).

There were significant interactions between both species and sampling location (edge vs. interior) ($P = 0.008$), and species and shape index ($P = 0.014$) (Table 3). However, there were no interactions between sampling location (edge vs. interior) and shape index ($P = 0.914$), or the three factors together ($P = 0.628$). Fig. 5 shows the interaction between species and sampling location. In edge transects all species were more dense than in interior transects. *L. japonica* was much more dense than the other two species, *R. multiflora* and *E. umbellata*, which are similar. *R. multiflora* was more dense than *E. umbellata* in the edge, however the opposite was true in the interior where *E. umbellata* was more dense than *R. multiflora*. This suggests that *E. umbellata* is more likely to invade forest interiors than is *R. multiflora*.

t-Tests indicated that density between *R. multiflora* and *E. umbellata* were not significantly different in edge transects ($P = 0.60$), but the densities were significant in interior transects (0.007). This relationship can be visualized in Fig. 5 and suggests that these two species were equally prevalent in edges, however, *E. umbellata* was significantly more prevalent in the interior of fragments. Mean densities between *R. multiflora* and *L. japonica* ($P = 2.5 \times 10^{-7}$) and *L. japonica* and *E. umbellata* ($P = 1.6 \times 10^{-7}$) in edge transects, and *R. multiflora* and *L. japonica* ($P = 1.5 \times 10^{-5}$) and *L. japonica* and *E. umbellata* ($P = 0.0002$) in interior transects were not significant.

A two-way ANOVA on the effect of sampling location (edge vs. interior) and fragment shape index on the three species of invasive plants revealed that *L. japonica* ($P = 0.000$) and *R. multiflora* ($P = 0.013$) had significantly different densities between edge and

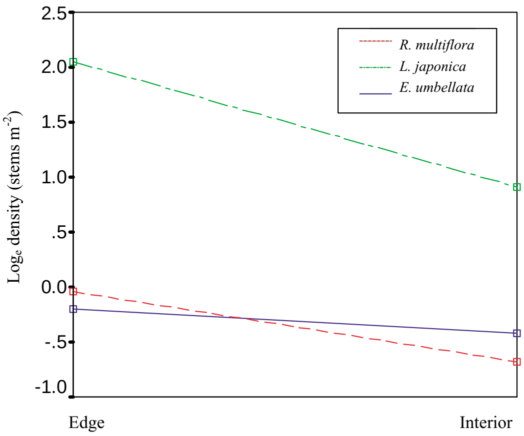

Fig. 5. Interaction of species and sampling location on invasive plant density in the three-way GLM analysis of variance.

interior transects. *E. umbellata* ($P = 0.205$) was not significantly different between the two sampling locations (Fig. 3).

Shape index was a significant factor for density of *R. multiflora* ($P = 0.036$) and *L. japonica* ($P = 0.012$), but not *E. umbellata* ($P = 0.326$). There was no relationship between sampling location and fragment shape index for any of the three invasive species.

4.2. Mean species heights

Mean heights of all three invasive plants were slightly higher in edge transects vs. interior transects. All species had greater mean heights in edge vs. interior transects, but only *L. japonica* had significantly greater heights in the edge transect when compared with interior transects ($P = 0.01$).

5. Discussion

All three invasive species are present in the study sites and have all invaded the sites' forested interiors. This is consistent with Meiners and Pickett (1999) who found that non-natives were all within 20 m of forest edge. All three invasive plants were in interior transect sampling locations, ranging from 195 to 495 m into interior plots. Species, sampling location and fragment shape index were found to be significant factors influencing invasive plant density in TNEC forest fragments. Heights of all species were slightly higher in edge vs. interior sampling locations, however, this relationship was significant only for *L. japonica*.

5.1. Sampling location

The decline in the density of invasive plants in interior transects suggests that edge effects decline towards the interiors of these forested fragments (H_1). Ranney et al. (1981) stated that vegetation beyond 10–15 m from the edge of forest fragments retained characteristics of interior forests. Interior transects in this study were all placed well beyond 10–15 m from the edge transects (195–495 m). Murcia (1995) stated that edge effects usually do not extend beyond 50 m into forest interiors. Therefore, this study revealed that the three invasive species penetrated into site interiors up to 495 m. The prevalence of all three invasive plant species within interiors of the forest fragments could be a result of their particularly aggressive nature (Szafoni, 1990) or the small size of the study fragments. Fragments were relatively small, therefore the interior sampling locations do not ecologically function as interior habitat with the characteristic microenvironmental differences between forest fragment edges.

Luken and Goessling (1995) argued that seed inputs are greater at forest fragment edges than interiors. Forest edges often attract birds because of ample nesting sites and food sources provided in edges (Hunter, 1999). All three of the invasive plants investigated in this study are dispersed by birds. This may be a factor in explaining why these three species were more dense in edge transects at TNEC.

Light availability is also an important factor in habitat colonization by invasive plants (Hobbs and Huenneke, 1992). The creation of forest edges can result in increased light availability for understory plants (Groom and Schumaker, 1993; Cadenasso and Pickett, 2001), therefore causing the edge environment to be favorable for many invasive species. A decrease in the quality of light in the interior of forest fragments may suppress seed germination even if the seeds are able to penetrate into the forest interior (Luken and Goessling, 1995). Ranney (1977) found that increased solar radiation may augment plant growth along fragment edges, and understory cover was observed to increase from 15% at forest interiors to 40% at forest edges. The present study confirms Ranney's findings that both heights of invasive species, which necessarily correlate with plant growth, and density of invasive species, were greater at edges than interiors.

Brothers and Spingarn (1992) noted that although invasive species frequency decreased from forest fragment edges to interiors, invasive species still permeated forest interiors. However, plants within interior plots were generally smaller, more isolated, non-reproducing individuals (Brothers and Spingarn, 1992). Although isolation or reproduction capabilities of the three invasive plant species was not considered, invasive plants in the interior of forest sites were smaller than those at edges.

Although the three species were present in all fragments, *R. multiflora* and *L. japonica* had significantly greater densities in edge transects as opposed to interior transects. These two species also had significant differences in density among fragments of different shape indices. *E. umbellata* did not have significant differences in density between edge and interior sampling locations or among fragments with different shape indices. This suggests that both location within a fragment and fragment shape are primary factors affecting prevalence of *L. japonica* and *R. multiflora* in the study area.

The significant differences between invasive plant density between fragment edge and interior sampling locations may have been observed because southern edges of fragments were used to represent edge habitat when sampling invasive species. Edge effects are thought to be more prominent in southern aspects (Palik and Murphy, 1990; Murcia, 1995; Williams-Linera et al., 1998). In future studies it would be interesting to compare invasive plant densities in northern, eastern, and western edge transects to determine if the same significant differences between invasive plant densities occur between interior transects and edge transects of different aspect.

5.2. Site shape index

As indicated previously in the description of the four TNEC study sites, site 1 had experienced more disturbance and perhaps the interior sampling transect did not represent true ‘‘interior’’ habitat as could be found in the other three sites. Sites 2–4 are old-growth forest (Carter and Wallace, personal comm.), dominated by mature forest species like *Quercus* spp., *Acer* spp., and *Carya* spp., which provide greater canopy cover and less light availability to understory shrubs like *R. multiflora*, *L. japonica*, and *E. umbellata* in the interior of the sites. All sites, with the exception of site 1, have interior transects in which the forest floor is shaded. This difference in available light may account for the presence of more invasive species in the interior transect vs. the edge transect observed only in site 1.

This research generally supports the preservation of forest with less edge (lower shape index values) to reduce invasion by non-native plants. However, when the overall mean invasive species density in fragments is considered, with the exception of site 1, density decreased with increasing shape index. This is opposite of what would be expected. The same relationship occurred for mean invasive species density in only edge transects. However, when mean invasive species density of only interior transects is considered, the expected density increase with increasing shape index does occur. Therefore interiors of forest fragments can be protected from invasion by having less edge in fragments for invasive species to enter and become established.

The SLOSS (single large or several small) reserves debate on the spatial attributes of tracts of land for conservation (Wilcox and Murphy, 1985) has been used when investigating the effects of forest fragmentation on native species diversity. Higher species diversity found in larger tracts of land has been linked with greater resistance to invasion by plants (Fragmentation, 2000; Hobbs, 2000). This study was unable to assess how site area affected invasive plant density. However, by using shape index, a measure of an area’s departure from circularity, it appeared that invasive plant density decreased with increasing shape index, which is not what one would generally expect (H_2). Theoretically, a forest site with low shape index (more circular) should have fewer invasive species because of its lower edge to interior ratio. However, we provide evidence that sites with high shape indices (greater amount of edge vs. interior) are not necessarily more readily invaded. Stohlgren and Banowetz (1999) and Cadotte and Lovett-Doust (2001) also found that invasive plants are slightly more likely to be found in diverse communities (lower shape index), mainly because invasibility depends more on the presence of resources than on absence of competitors.

5.3. Invasive plant height

All sites have greater plant heights in the edge than the interior, which could be interpreted as an indicator of greater biomass, in support of H_3. It might be hypothesized that heights of individuals in edge transects will be greater than heights of individuals in interior transects. This could be the result of differences in microenvironmental variables such as light availability, air and soil temperature and moisture at the edges of forested fragments that could promote growth and development of invaders. Increased microenvironmental resources at forest fragment edges could result in invasive species becoming more "established" at forest edges compared with forest interiors. This relationship could be reflected in higher invasive species heights at forest fragment edges vs. interiors. Especially in the case of *R. multiflora* and *E. umbellata* increased heights at forest edges may allow these plants to occupy niches above the understory layer where they may shade out native herbaceous understory plants.

5.4. Species

The data confirm that there are differences in density between the three species at the study sites (H_4). Species as a significant factor influencing invasive plant density is an expected finding because three different invasive species, which would likely have three different density distributions or patterns, were investigated in this study. *R. multiflora* and *E. umbellata* would likely be closer to the same density because they both have the same woody shrub growth form. *L. japonica* can take the form of a woody vine, however, the majority of *L. japonica* within the plots were not well established and had not reached this more mature state of woody growth form. Disproportionately, many pioneer plants are non-native invasive species. These three species are aggressive invasive plants, which exhibit pioneer species characteristics: (1) the ability to germinate under a variety of conditions, (2) preference for high light environments (shade intolerance), and (3) ability to withstand disturbance (Iverson et al., 1999; Szafoni, 1990).

Ebinger (1983) reports *E. umbellata* densities in east-central Illinois from 0.5 to 3.4 stems m^{-2}. Mean density of *E. umbellata* for both edge (0.90) and interior transects (0.38) is comparable to Ebinger's lower densities. Although *E. umbellata* had greater density in edge vs. interior sampling locations, this relationship was not significant. When comparing *R. multiflora* and *E. umbellata* densities, they are equally prevalent on edges, but *E. umbellata* is significantly more prevalent in interiors. This supports findings by Nestleroad et al. (1984, unpublished data), that mature *E. umbellata* will tolerate light shade, although they produce more fruits in full sun. This may explain why *E. umbellata*, and not the other two shade intolerant species, was more likely to invade forest interiors.

6. Conclusions

This study illustrates the presence and relative prevalence of three invasive species at TNEC. These results can lend support to further studies of invasive species in the southern Illinois region, especially those that propose to develop invasive species management techniques. Westman (1990) states that in the absence of guidelines for prioritizing control of invasive species, "often the easiest plant species to control are given the most attention, regardless of the relative ecological damage they pose". This study suggests that *E. umbellata*, unlike the other two species, is just as well established in the interior of fragments as in fragment edges. Although, *E. umbellata* may not be the easiest to remove, it may pose the most ecological damage once it has become well established as it had been in all study sites. It should be given a high priority when implementing management and control efforts in southern Illinois forest ecosystems.

Future studies might quantify the microenvironmental variables that distinguish edges and interior environments, examine herbaceous invasive species as well as woody species examined here, or compare densities in these sites to sites of much greater or smaller area. Land managers are interested in knowing if invasive species are just present or if they actually are replacing native species. Future researchers could survey all vegetation within the sites to assess how native plant diversity affects invasive plant densities.

Acknowledgements

We acknowledge the staff at Touch of Nature Environmental Center, in particular C. Carter and J. Wallace, for their cooperation and assistance with site location and description in the initial stages of this project. We also thank the Geography Department at SIU Carbondale for the use of equipment and facilities in support of this research. We are grateful to two anonymous reviewers for valuable comments on a previous version of the manuscript.

References

Anderson, R.C., Anderson, M.R., 1975. The presettlement vegetation of Williamson County, Illinois. Castanea 40, 345–363.

Ashman, R.I., 1962. A plan for the management of the Southern Illinois University forest. Unpublished Internal Document. Southern Illinois University, Carbondale.

Bratton, S.P., 1982. The effect of exotic plant and animal species on nature preserves. Natural Areas J. 2 (3), 3–13.

Brothers, T.S., Spingarn, A., 1992. Forest fragmentation and alien plant invasion of central Indiana old-growth forests. Conserv. Biol. 6, 91–100.

Cadenasso, M.L., Pickett, S.T.A., 2001. Effect of edge structure on the flux of species into forest interiors. Conserv. Biol. 15, 91–97.

Cadotte, M.W., Lovett-Doust, J., 2001. Ecological and taxonomic differences between native and introduced plants of southwestern Ontario. Ecoscience 8, 230–238.

Coblentz, B.E., 1990. Exotic organisms: a dilemma for conservation biology. Conserv. Biol. 4, 261–265.

Davis, G.T., 1987. A floristic study of the Touch of Nature Environmental Center. M.S. Thesis. Department of Botany, Southern Illinois University, Carbondale.

Douglas, K., 1998. Hot spots: why are there so many species in the tropics? New Sci. 158, 32–36.

Drake, J.A., Mooney, H.A., di Castri, F., Groves, R.H., Kruger, F.J., Williamson, M., 1989. Biological Invasions: A Global Perspective. Wiley, New York.

Ebinger, J.E., 1983. Exotic shrubs: a potential problem in natural area management in Illinois. Nat. Areas J. 3, 3–6.

ESRI, 2002. ArcView 3.0. Environmental Systems Research Institute, Redlands, CA.

Fragmentation, 2000. A Conference on Sustaining Private Forests in the 21st Century, Annapolis, MD, September 17–20, 2000.

Game, M., 1980. Best shape for nature reserves. Nature 287, 630–631.

Groom, M.J., Schumaker, N., 1993. Evaluating landscape change: patterns of worldwide deforestation and local fragmentation. In: Kareiva, P.M., Kingsolver, J.G., Huey, R.B. (Eds.), Biotic Interactions and Global Change. Sinauer Associates, Sunderland, MA.

Harrison, S., 1999. Local and regional diversity in a patch landscape: native, alien, and endemic herbs on serpentine. Ecology 80 (1), 70–80.

Harty, F.M., 1986. Exotics and their ecological ramifications. Nat. Areas J. 6, 20–26.

Herman, R.J., Miles, C.C., Dungan, L.A., Currie, B.E., Ice, P.W., 1979. Soil Survey of Jackson County, Illinois. United States Department of Agriculture, Washington, DC.

Hobbs, R.J., Humphries, S.E., 1995. An integrated approach to the ecology and management of plant invasions. Conserv. Biol. 9, 761–770.

Hobbs, R.J., Huenneke, L.F., 1992. Disturbance, diversity, and invasion: implications for conservation. Conserv. Biol. 6, 324–337.

Hobbs, R.J., 2000. Land use changes and invasions. In: Mooney, H.A., Hobbs, R.J. (Eds.), Invasive Species in a Changing World. Island Press, Washington, DC.

Hunter, M.L., 1999. Maintaining Biodiversity in Forest Ecosystems. Cambridge University Press, Cambridge, UK.

Huxel, G.R., 1999. Rapid displacement of native species by invasive species: effects of hybridization. Biol. Conserv. 89, 143–152.

Iverson, L.R., Ketzner, D., Karnes, J., 1999. Illinois Plant Information Network. Illinois Natural History Survey and USDA Forest Service. http://www.fs.fed.us/ne/delaware/il-pin.html.

Krebs, C.J., 1989. Ecological Methodology. Harper Collins Publishers, New York.

Laurance, W.F., Yensen, E., 1991. Predicting the impact of edge effects in fragmented habitats. Biol. Conserv. 55, 77–92.

Lee, G., Macdonald, I.A.W., 1997. Foreword. In: Proceedings of the World Conservation Congress Workshop on Alien Invasive Species, Montreal, Canada, October 13–23, 1996.

Lovejoy, T.E., Bierregaard, R.O., Rylands, A.B., Malcom, J.R., Harper, C.E., Brown, K.S., Powell, A.H., Powell, G.V.N., Shubart, H.O.R., Hays, M.B., 1986. Edge and other effects of isolation on Amazon forest fragments. In: Soule, M.B. (Ed.), Conservation Biology: The Science of Scarcity and Diversity. Sinauer Associates, Sunderland, MA.

Luken, J.O., Goessling, N., 1995. Seedling distribution and potential persistence of the exotic shrub *Lonicera maackii* in fragmented forests. Am. Midl. Nat. 133, 124–130.

Mack, R.N., Simberloff, D., Lonsdale, W.M., Evans, H., Clout, M., Bazzaz, F., 2000. Biotic invasions: causes, epidemiology, global consequences, and control. Issues Ecol. 5.

Meekins, J.F., McCarthy, B.C., 2001. Effect of environmental variation on the invasive success of a nonindigenous forest herb. Ecol. Appl. 11, 1336–1348.

Meiners, S.J., Pickett, S.T.A., 1999. Changes in community structure and population responses across a forest-field gradient. Ecography 22, 261–267.

Mueller-Dombois, D., Ellenberg, D.H., 1974. Aims and Methods in Vegetation Ecology. Wiley, New York.

Murcia, C., 1995. Edge effects in fragmented forests: implications for conservation. TREE 2, 58–62.

National Climatic Data Center TD 9641 Clim 81, 1961–1990. Normals. http://www.worldclimate.com/cgi-bin/grid.pl?gr=N37W089.

Palik, B.J., Murphy, P.G., 1990. Disturbance verses edge effects in sugar-maple/beech forest fragments. For. Ecol. Manage. 32, 187–202.

Patton, D.R., 1975. A diversity index for quantifying habitat edge. Wildlife Soc. Bull. 3, 171–173.

Ranney, J.W., 1977. Forest island edges: there structure, development, and importance to regional forest ecosystem dynamics. Oak Ridge National Laboratory, Oak Ridge, TN.

Ranney, J.W., Bruner, M.C., Levenson, J.B., 1981. The importance of edge in the structure and dynamics of forest islands. In: Burgess, R.L., Sharpe, D.M. (Eds.), Forest Island Dynamics in Man-Dominated Landscapes. Springer-Verlag, New York.

Richardson, D.M., Pysek, P., Rejmanek, M., Barbour, M.G., Panetta, F.D., West, C.J., 2000. Naturalization and invasion of alien plants: concepts and definitions. Divers. Distrib. 6, 93–107.

Schonewald-Cox, C.M., Bayless, J.W., 1986. The boundary model: a geographical analysis of design and conservation of nature reserves. Biol. Conserv. 38, 305–322.

Shigesada, N., Kawasaki, K., 1997. Biological Invasions: Theory and Practice. Oxford University Press, Oxford, UK.

SPSS, 1999. SYSTAT 9 Statistics I. SPSS Inc., Chicago, IL.

StatSoft Inc., 1995. STATISTICA for Windows. StatSoft Inc., Tulsa, Oklahoma, USA.

Stohlgren, T., Banowetz, M., 1999. USGS research upsets conventional wisdom on invasive species invasions. USGS, Midcontinent Ecological Science Center. http://159.189.161.2/news/press_releases/conventional_wisdom.html.

Stork, N.E., 1997. Measuring global biodiversity and its decline. In: Reaka-Kudla, M.L., Wilson, D.E., Wilson, E.O. (Eds.), Biodiversity II: Understanding and Protecting Our Biological Resources. Joseph Henry Press, Washington, DC.

Szafoni, B., 1990. Vegetation Management Manual. Illinois Nature Preserves Commission, Springfield, IL.

Usher, M.B., Kruger, F.J., Macdonald, I.A.W., Loope, L.L., Brockie, R.E., 1988. The ecology of biological invasions into nature reserves: an introduction. Biol. Conserv. 44, 1–8.

Voigt, J.W., Mohlenbrock, R.H., 1964. Plant Communities of Southern Illinois. Southern Illinois University Press, Carbondale.

Westman, W.E., 1990. Managing for biodiversity: unresolved science and policy issues. BioScience 40, 26–33.

Wilcove, D.S., McLellan, C.H., Dobson, A.P., 1986. Habitat fragmentation in the temperate zone. In: Soule, M.E. (Ed.), Conservation Biology: The Science of Scarcity and Diversity. Sinauer Associates, Sunderland, MA.

Wilcox, B.A., Murphy, D.D., 1985. Conservation strategy: the effects of fragmentation on extinction. Am. Nat. 125, 879–887.

Williams-Linera, G., Dominquez-Gastelu, V., Garcia-Zurita, M.E., 1998. Microenvironment and floristics of different edges in a fragmented tropical rainforest. Conserv. Biol. 12, 1091–1101.