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Restoration of a forest understory following the removal of an invasive shrub, Amur honeysuckle (*Lonicera maackii*)

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ABSTRACT - The recruitment of native seedlings is often reduced in areas where the invasive Amur honeysuckle (*Lonicera maackii*) is abundant. To address this recruitment problem, we evaluated the effectiveness of *L. maackii* eradication methods and restoration efforts using six native tree seedling species planted within eradication and unmanipulated (control) plots. Two eradication methods using glyphosate herbicide were evaluated: cut & paint and stem injection with an EZ-Ject® lance. *Lonicera maackii* density and biomass, as well as microenvironmental characteristics were measured to study their effects on seedling growth and survivorship. Mean biomass of Amur honeysuckle was 360.9 ± 69.1 kg · ha^{-1}, and density was 21380.5 ± 3171.5 plants · ha^{-1}. Both eradiation treatments were effective in killing *L. maackii* (≥ 94%). The injection treatment was most effective on large *L. maackii* individuals (> 3 cm diam) but was 43% faster to apply than cutting and painting, was less fatiguing for the operator, decreased operator exposure to herbicide, and minimized impact to non-target vegetation. Deer browse tree protectors were used on half of the seedlings, but did not affect survivorship or growth. Survival of native seedlings after three years was significantly less where *L. maacki* was left intact (31.9 ± 3.0%) compared to the eradication plots ($P < 0.002$). Survival was significantly different between cut (51.1 ± 3.5%) and injected (45.0 ± 3.4%) plots. Species had different final survival and rates of mortality. Species survival differed greatly by species (in descending order): *Fraxinus pennsylvanica* > *Quercus muehlenbergii* ≥ *Prunus serotina* ≥ *Juglans nigra* > *Cercis canadensis* > *Cornus florida*. Survivorship and growth of native seedlings was affected by a severe first-year drought and by site location. One site exhibited greater spring soil moisture, pH, percent open canopy, which translated into greater survivorship relative to the other site (55.5 ± 2.2% versus 29.9 ± 1.7%). Overall, both *L. maackii* eradication methods were successful, but

restorationists should be aware of the potential for differential survivorship of native seedlings depending on species and microenvironmental conditions.

Key words - succession, recruitment, non-native, understory, Ohio, forest ecology, restoration ecology, tree tubes, seedlings, biodiversity, invasive species

Introduction

Invasions of non-indigenous species are often facilitated by anthropogenic disturbances (Hobbs & Huenneke 1992). The invasion of these species can be problematic in that they can cause further changes in forest attributes including structure, properties, and fundamental ecosystem processes (MacMahon & Holl 2001). Furthermore, if extremely successful species are the invaders, biodiversity can be essentially reduced to a near monoculture resulting in a community that is low in natural diversity and extremely difficult to restore (Olson & Whitson 2002).

The aggressiveness of some invasive species can make wholesale eradication quite difficult; however, their removal at a specific site is a necessary step in the restoration process. Eradication at local spatial scales is especially important because the restoration process, for the most part, proceeds on a site-by-site basis across large areas (Wiens et al. 1993).

After the removal of invasive species occurs, forest restoration practices may include the replanting of sites with native species (Ghersa et al. 2002). This replanting step in restoration is often necessary especially in the deciduous forests of the eastern U.S. where the viability of seeds are relatively short compared to other habitat types (Barnes et al. 1998), and where invasives have occupied a site long enough to result in the degradation of a normally short-lived forest seed bank (Collier et al. 2002).

Along with replanting to restore forest diversity, come several other added benefits. Replanting may inhibit further invasion because native individuals may preempt space and acquire resources which may make it more difficult for non-natives to become reestablished (Shea & Chesson 2002). Natural recovery processes, including succession, would be accelerated by replanting, changing the forest from a non-

indigenous, highly modified ecosystem to one with native species and hopefully self-sustaining processes. The action of planting natives would also circumvent the problem of lack of successful recruitment frequently encountered in the reestablishment of plant communities (Holmes 2001). All of these factors would hopefully result in the restoration of a more structurally representative and functional forest community.

Another important benefit from conducting a restoration experiment would be that practical experience would be gained in developing a protocol for re-establishing natural forest ecosystems. This plan would successfully integrate the abiotic factors (including micro-site variability) as well as the biotic factors (including seedling requirements and effects of deer browse) on the restoration of the forest community. A successful protocol would then incorporate the understanding of site variation and its effects on restoration in terms of structure, function, and recruitment (Reifsnyder & Lull 1965). In addition, a consideration of the relative economic costs of different restoration methods would help guide allocation of sponsorship in future, similar restoration programs (Montalvo et al. 2002).

One of the most challenging plants to forest restorationists is the non-indigenous, invasive shrub, Amur honeysuckle [*Lonicera maackii* (Rupr.) Herder; Caprifoliaceae]. This plant has spread to over 27 eastern U.S. states (Luken & Thieret 1996), the province of Ontario, Canada, and many counties in southwestern Ohio (Hutchinson & Vankat 1997; Trisel 1997) where this experiment was conducted. *Lonicera maackii* is only one member of a genus with many known weedy qualities (Woods 1993; Schierenbeck et al. 1994).

A number of factors make *L. maackii* a threat to native biodiversity and a challenge to restoration practitioners including its ability to resprout following repeated cutting

(Luken 1990), possible allelopathic effects on native vegetation (Trisel 1997), and superior above-ground competitive ability (Trisel 1997). When *L. maackii* invades open sites, these areas are often converted into shrub communities (Luken & Thieret 1995), and in invaded forests, the recruitment and diversity of woody and herbaceous species is often reduced (Hutchinson & Vankat 1997; Gould & Gorchov 2000; Collier et al. 2002). Despite much effort being put forth in eradicating this species, a fully integrated restoration protocol for the eradication and subsequent replacement with natives for this species has not yet been produced. This precipitated the need to investigate the optimal methods for restoring native vegetation following *L. maackii* removal.

The specific goals of this study were to: (1) quantify the biomass of *L. maackii* in the study area, (2) compare methods of *Lonicera maackii* eradication in terms of effectiveness and ease of application, (3) compare differential survival and growth of native tree seedlings planted among *L. maackii* eradication treatments, (4) evaluate the effects of deer tree protectors on native seedling survival, (5) and explain the influence of microenvironmental factors on tree seedling growth and survival.

Methods

Study Site

This experiment was carried out at the Fernald Environmental Management Project (FEMP) Site (39°18'20" N × 84°41'50" W). This is a 425 ha facility located ca. 29 km northwest of Cincinnati, Ohio. The old production facility at FEMP was used for the manufacture of high-grade uranium and thorium to support the U.S. weapons defense program until 1989 at which time remediation and restoration became the primary goal of FEMP (U.S. Dept. of Energy 2002). This research project was conducted in the

"North Woodlot" within the FEMP site, which is 65 ha in size and is located north of the old production facility. Topographically, elevations range from ca. 170 - 190 m (U.S. Geological Survey 1981). The north woodlot area contains four general types of habitats including "old-fields, previously mowed meadows, regenerating forest, and mature forest" (McCarthy 1998).

This area has been subject to considerable anthropogenic-related disturbance (e.g., mowing, roads, grazing), which may have facilitated the influx of *L. maackii*. A floristic study conducted on the site by McCarthy (1999) reported that out of 332 taxa 30.5% were non-indigenous with the most invasive species being *Rosa multiflora*, *Alliaria petiolata*, *Festuca elatior*, and *Polygonum persicaria*; however, he reported that *Lonicera maackii* was the most problematic non-native plant in the forested areas.

All soils within the study site are deep but vary in their drainage and slope. Site A is primarily Xenia silt clay loam (XeB) and lies on a well drained till plain. Site B lies on Ragsdale silt clay loam (Ra) and has poorly drained soils with a flat topography (Lerch et al. 1980).

The climate of Butler County is typical of most of southern Ohio with cold winters and hot summers. The mean temperature for December to February is 0 C, and the mean daily minimum temperature is -5 C. The mean temperature from June to August is 22 C, and the average daily maximum temperature is 29 C (National Climate Data Center / National Oceanic and Atmospheric Administration 2002). The total annual rain precipitation is 53.3 cm, and the total snowfall is 38.1 cm. Eight days in winter have at least 2.5 cm of snow on the ground. Sixty percent of the total precipitation usually falls from April through September, although precipitation is present in every month. The last

frost date of spring is April 26, and the first frost date is October 26. The growing season is 172 days (Lerch et al. 1980).

Experimental setup

This experiment employed a completely randomized block design (Sokal and Rohlf 1995) with eight 5.5 × 13.5 m replicate blocks. Blocks were located in two areas roughly 150 m apart which both contained *L. maackii* stands (sites A and B). Site A contained a total of five replicate blocks, had good drainage, and was dominated by *Carya laciniosa* in the overstory. Site B contained three replicate blocks, had relatively poorer drainage, and was dominated by a mixture of *Acer negundo*, *Cornus florida*, *Fraxinus pennsylvanica*, *Prunus serotina*, and *Ulmus americana* in the overstory. Three rectangular 4.5 × 5.5 m treatment subplots were established within each block. Treatments were randomly assigned to subplots and were designed to test their efficacy of killing *L. maackii* and ultimately their influence on the survival of native tree seedlings. Each block consisted of (1) a control subplot in which no *L. maackii* was removed from plot; (2) a cut subplot in which *L. maackii* stems were cut near ground level, removed from plot, and stumps painted with 50% glyphosate isopropylamine salt solution (Roundup®; Monsanto Company, St. Louis, Missouri); and (3) an injection subplots in which *L. maackii* was killed using an E-Z-Ject® lance (Odum Processing Engineering Consulting, Inc.; Waynesboro, Missouri) but left standing. This lance works by pushing a glyphosate-filled 22-caliber capsule (Bergerud 1988) manually through the bark of the stem or swollen stem base and into the vascular system of the selected woody plant. On large honeysuckle individuals with two or more stems, each stem was injected separately; otherwise, plants were injected individually. Prior to eradiation treatments, all honeysuckle stems in subplots were tagged and measured.

Within each treatment subplot ten individuals each of six species of one-year-old indigenous tree seedlings were randomly planted 0.75 m apart using a dibble bar. Tree seedlings included chinkapin oak (*Quercus muehlenbergii*), black walnut (*Juglans nigra*), black cherry (*Prunus serotina*), green ash (*Fraxinus pennsylvanica*), flowering dogwood (*Cornus florida*), and redbud (*Cercis canadensis*). These species all occur within the county where this experiment took place (Braun 1989) and were found as canopy or sub-canopy species in the area (McCarthy 1999). It was observed that recruitment of tree species was especially poor below *L. maackii* stands (also found by Hutchinson & Vankat 1997 and Collier et al. 2002), which justified replanting after eradication rather than just removal of honeysuckle. Half of the 1440 tree seedlings were enclosed in 122 cm Protex® Pro/Gro Solid Tube Tree Protectors (Forestry Suppliers, Inc; Jackson, Missouri; i.e., tree protectors) to exclude white-tailed deer (*Odocoileus virginianus*) and test the effects of deer browse on tubed versus non-tubed seedlings. Blocks were surrounded by two strands of barbed wire (50 & 100 cm. ht.) with a 1 m buffer to exclude cattle (present for the first year of the study) but not deer. Honeysuckle eradication treatments were applied 24-Mar-99. Seedlings were planted on 24-Mar-99 to 30-Mar-99, and deer tree protectors were applied and staked on the north side from 31-Mar-99 to 11-Apr-99. The experiment ran from 24-Mar-99 to 10-Oct-01.

Sampling

Seedlings were measured for height (cm) and basal diameter (mm) at initial planting. A preliminary analysis of variance (ANOVA) indicated that seedling size was not significantly different ($P > 0.10$) among blocks or treatment subplots at initial time of planting; therefore, an analysis of covariance was not needed. Seedlings were also measured for height and diameter at two other times during 1999, 2000, and 2001, as well

as, evaluation of survivorship and mode of mortality (27-May-99, 10-Oct-99, 21-May-00, 23-Sept-00, 25-May-01, 7-Oct-01).

Environmental data were collected over the growing seasons of 1999 and 2000 on a treatment subplot level. Soil moisture was collected twice in 1999 (25-Jun, 21-Aug) and 2000 (29-July, 29-Aug). Soil moisture was analyzed gravimetrically (McCarthy 1997). Soil pH was measured using a glass electrode method with a Corning 350 pH / ion meter in a 2:1 water-to-soil solution (25-Jun-99, 21-Aug-99, 29-Jul-00, 29-Aug-00). Soil nitrate was measured using absorbent Rexyn® 300 (H - OH) beads (Fisher Scientific; Fair Lawn, New Jersey) buried in the soils' A-horizon in nylon mesh bags for roughly three months (20-May-99 to 21-Jul-99 and 21-May-00 to 22-Jul-00). Nitrate was removed from the Rexyn® beads with 2M KCl solution and analyzed with a cadmium reduction method using NitraVer® 5 Nitrate Reagent (Hach; Loveland, Colorado). Three soil moisture, nitrate, and pH samples were taken at random locations in each subplot each time sampling occurred. Light availability was measured using 35 mm images taken with a 8mm hemispherical fish-eye lens at a height of 0.5 m on 30-Jul-99 and 27-Jul-00. These images were digitized and then analyzed using GLIC software (Canham 1988). A full description of the protocol can be found in Robison and McCarthy (1999). One image was taken per subplot (24 total photographs). Air temperature and humidity were measured for eight random locations inside and outside of deer tree protectors in each subplot on 12-Jul-99 and 13-Jul-00 with a Corning thermohygrometer.

Statistical analysis

To assess abundance of *L. maackii* within blocks and among treatment plots, a regression model was constructed to estimate honeysuckle biomass (from stem density

and diameter) within blocks and treatment subplots using randomly selected on-site honeysuckle plants (N = 32).

An analysis of variance was used to compare final native seedling survival among treatments after three years. Percent survival was used as the independent variable, and the factor variables were site (random factor), block within site (nested), honeysuckle eradication treatment (fixed), species (fixed), and tree protector (fixed). Normality and equal variance assumptions were satisfied using the D'Agostino omnibus test (D'Agostino et al. 1990) and the F-Max test (Dowdy & Wearden 1991), respectively. When necessary, data transformations ($\log_{10}$ or square root) were conducted in order to meet these assumptions. Untransformed means and standard errors are reported throughout the results.

The shapes of survival curves were analyzed to determine if the native seedlings died at different rates during the three years of sampling. Site, treatment, and species survival distributions were compared using the log rank nonparametric test. This test was utilized because greater than 20% of the seedlings were still alive at the end of the experiment (i.e., censored observations; Pyke & Thompson 1986).

Loglinear modeling was used to analyze the modes of mortality for native seedlings. This uses an n-dimensional contingency table and a stepdown model selection process to determine the most important factors related to whether the seedlings died by drought, fungus, handling, or deer browsing. Computed *Chi*-square values were compared to critical values to determine statistical significance.

MANOVAs were used to examine the microenvironmental patterns as they related to years (1999, 2000), eradication treatments (control, cut, and inject), and sites (A and B), which were all used as predictor variables. All factors were treated as fixed, except site.

Spring soil moisture and pH, nitrate, percent open canopy, ambient air temperature, and relative humidity were analyzed as response variables. Fall soil moisture and fall pH were dropped from the analysis due to multicollinearity, but this did not affect the MANOVA results. The assumptions of multivariate normality and equal variance (Scheiner 1993) were satisfied prior to analysis via data transformations ($\log_{10}$). Wilks' Lambda values were calculated for the overall MANOVA table, and individual ANOVAs were conducted to determine which factors were important in producing the overall MANOVA results.

A repeated measures analysis of variance (RMANOVA) was used to analyze seedling height growth. The data passed the assumptions of residuals following the normal probability distribution and equal within-subject covariances (Von Ende 1993). There was a violation of the sphericity pattern of the covariance matrices (Mauchley's criterion = 144.16, $P = 0.0346$); therefore, a Huynh-Feldt Epsilon correction ($\varepsilon = 0.95$) was used to create a more stringent critical F-value (Crowder and Hand 1990). Seedling diameter was also analyzed with a RMANOVA. Like the height data, diameters suffered from lack of sphericity of the covariance matrices (Mauchley's criterion = 620.07, $P < 0.001$). A Huynh-Feldt correction was used to create a more conservative F-test ($\varepsilon = 0.88$). Site was a random factor, and individual seedling was nested. All other factors such as eradiation treatment, species, and tubing were fixed. To aid in interpretation of interactions, Cicchetti contrasts were computed (Cicchetti 1972). Cicchetti contrasts enable the analysis of every unconfounded comparison by holding all factors constant but one.

All statistical computations of ANOVAs, loglinear modeling, RMANOVAs, and Bonferroni post hoc tests were conducted using NCSS Version 5.0 (Hintze 2000). MANOVA calculations were conducted using SAS Version 8.0 (SAS Institute 2001).

Bonferroni multiple comparison tests were conducted when significant F-tests were found for ANOVAs. Unless otherwise stated, statistical tests were significant when $P < 0.05$.

Results

Honeysuckle parameters

The regression model showed that honeysuckle stem diameter at 5 cm above ground explained the greatest amount of variance; therefore, this was the parameter that was used to estimate honeysuckle biomass ($R^2 = 0.91$ honeysuckle biomass per seedling = basal area in $cm^2 * 0.907 + 0.147$). This equation estimates biomass very similarly to that reported by Luken (1988). Mean biomass of Amur honeysuckle was 360.90 ± 69.10 $kg \cdot ha^{-1}$. Mean stem density was 65959.60 ± 7637.30 stems $\cdot$ ha^{-1}. Mean density of *L. maackii* was 21380.50 ± 3171.50 plants $\cdot$ ha^{-1}. The mean number of stems per plant was 3.65 ± 0.26. A preliminary analysis of variance was conducted on honeysuckle biomass to evaluate pre-treatment differences among blocks and subplots. No significant ($P > 0.10$) differences were found.

Eradication treatment effectiveness on honeysuckle

Honeysuckle mortality was assessed at the end of the 1999 and 2000 growing seasons. At the end of 1999 the mortality was 99% for both of the two eradication treatments. For 2000 it was 98% in the injection plots and 94% in the cut and paint treatment plots. ANOVA results showed that there were no significant differences ($P > 0.10$) in honeysuckle mortality among treatment plots or between years. For both of these eradication treatments honeysuckle mortality was lower during 2000 due to resprouting. Resprouting occurred only for small diameter stems because it was difficult to apply the injection treatment to honeysuckle stems less than 3 cm in diameter, and some operator

error did occur on stems smaller than 3 cm during the initial 1999 injection treatment application. Only the stems in which injections were properly applied were included in the assessments of mortality.

Ease of application of eradication treatments, planting, and tubing

The cut and paint method required more time to apply (7 hours to eradicate 66 m^2 or 1060 person-hours · ha^{-1}) than the injection treatment (3 hours to eradicate 66 m^2 or 454 person-hours · ha^{-1}). In the injection treatment one person operated the E-Z-Ject® lance and another person cleared leaf litter away from honeysuckle stump bases when necessary. Planting of 1440 seedlings required 4 days work by 2 people (or 64 person-hours), and assembly and application of tree protectors on half of the seedlings (720 plants) required 8 days of work by 3 people (or 192 person-hours).

Comparison of costs of restoration

The startup costs for the cut and paint method totaled $253 USD, which included clippers ($30), loppers ($65), and glyphosate herbicide (2.5 gallons at $158). The injection startup costs were $599, which included the E-Z-Ject® lance ($467) and glyphosate-filled capsules (1200 capsules at $132). Protex® Pro/Gro Solid Tube Tree Protectors cost $1.74 each. Seedlings cost roughly $0.33 each for 1440 seedlings totaling $480. Labor costs of eradication treatments, tree seedling planting, and tree protector assembly and installation were not calculated due to the fact that our reports of eradication treatment times are likely overestimated as the application times for persons doing frequent restoration treatments would likely be much less; however, the total cost including the purchase of startup equipment, native seedlings, and tree protectors (but excluding labor) was $831 for the cut treatment and $1177 for the injection treatment.

Comparison of survival for native seedlings

The survival of planted seedlings was 51.28% ± 5.04 for 1999, 43.78% ± 5.13 for 2000, and 40.02% ± 4.98 for 2001. An analysis of variance of final survival of seedlings (after three years) found that two factors, block and tube, were non-significant ($P > 0.05$) and were not included in subsequent analyses. Site, treatment, species, and site × species were all significant (Table 1). Overall survival at site B (55.54% ± 2.21) was significantly greater ($P < 0.001$) than at site A (29.89% ± 1.71). The site × species interaction was significant ($P < 0.001$) indicating that species survived differently between the two sites (Table 1, Fig. 1). Survival between control and honeysuckle treatments was also significantly different ($P = 0.002$; control survival = 31.92% ± 2.99, cut survival = 51.16% ± 3.49, injection survival = 45.07% ± 3.43). Survival between cut and paint and injection eradication treatments were significantly different from each other (Fig. 2). No other two or three-way interactions were found to be significant (Table 1).

Because of significantly different final survival between sites, among years, and among treatments, survivorship was modeled to determine if seedlings died at different rates within these groups. Weibull distribution curves (Dodson 1994) were found to produce the best estimates of species survivorship. Seedlings within control treatments died at different rates than both the cut ($\chi^2 = 40.67$, $df = 1$, $P < 0.001$) and injection treatment ($\chi^2 = 23.68$, $df = 1$, $P < 0.001$); however the cut was not different than the injection treatment ($\chi^2 = 2.38$, $df = 1$, $P = 0.123$; Fig. 3). Because there was a site × species interaction for survival at the end of two years, species survival rates were analyzed separately within sites. Several species (*F. pennsylvanica*, *P. serotina*, and *J. nigra*) had fairly constant mortality rates over the length of the experiment at both sites. Other species (*Q. muehlenbergii* at site A and *C. canadensis* and *C. florida* at site B) had

high rates of early death followed by constant mortality. *Cercis candadensis* and *C. florida* had fairly constant but high mortality for only the first two years of the experiment with stabilized survival during the third year (Fig. 4).

Modes of mortality for native seedlings

The modes of mortality in different treatments were assessed to determine if seedlings died by different means among sites, honeysuckle treatments, and species. Overall the most common mode of mortality was drought (39.4%) followed by handling (6.31%), browsing (3.1%), and fungus (1.2%).

Logistic regression results indicated that the factors associated with mortality due to drought were site, treatment, and species ($\chi^2 = 34.5$, $df = 10$, $P < 0.001$). Drought was a greater factor of mortality at site A (29.7%) than at site B (9.7%), and drought was a greater factor of mortality in the control plots (16.5%) than in the injection (11.8%) or cut plots (11.1%). *Fraxinus pennsylvanica* had less mortality due to drought (2.4%) than the other five species (6.1 – 8.2%). Factors associated with mortality due to browsing were site and species ($\chi^2 = 7.6$, $df = 2$, $P = 0.022$). Overall, mortality due to browsing was 1.9% with site A having a greater mortality due to browsing (2.8%) than site B (0.3%). Species' mortality due to browsing were as follows: *C. florida* (1.5%), *C. canadensis* (1.1%), and *Q. muhlenbergii* and *P. serotina* (0.2%). Both *F. pennsylvanica* and *J. nigra* had no incidences of browsing. Mortality due to fungus was low but best predicted by the factors of site and species ($\chi^2 = 5.4$, $df = 2$, $P = 0.067$). Site A had a slightly greater incidence of fungus (0.8%) than site B (0.3%). The incidence of fungus mortality per species were as follows: *C. florida* (0.5%), *C. canadensis* and *Q. muhlenbergii* (0.14%), *F. pennsylvanica* and *J. nigra* (0.07%). Mortality due to handling was only significant for species ($\chi^2 = 441.31$, $df = 5$, $P < 0.001$). The handling mortality

incidences were mostly associated with *C. florida* (4.03%) followed by *C. candadensis* (2.14%) and *J. nigra* (0.14%), which is evident as early mortality (Fig. 4).

Seedling growth

Seedling height growth was analyzed by calculating total seedling height growth for each year, 1999 and 2001. Growth was not significantly different between sites (P = 0.40) and was subsequently removed from the analysis. Seedlings grew equally well in tubed and non-tubed conditions (P = 0.79) and among eradication treatments (P = 0.99). There were significant year, species, and species × year effects (all $P < 0.001$; Fig. 5).

Analysis of seedling diameters used the same factors as those in the height growth RMANOVA. Site was nonsignificant (P = 0.83) and was dropped from the analysis. Diameter growth was not significantly different between tubed and non-tubed seedlings and among treatments. There was a significant year ($P < 0.001$) and species × year interaction (P = 0.01). There were no significant differences among species' diameter growth for 1999 and 2000; however, there were significant species differences for 2001 (Fig. 6).

Analysis of environmental parameters

Only two factors, site and year, were significant ($P < 0.001$) in predicting microenvironmental response. There were no environmental differences among eradication treatments. Also, no two- or three-way interactions were significant. Follow-up ANOVAs indicated that spring moisture, pH, nitrate, and temperature were significantly different ($P < 0.001$) between years. Site A had significantly lower environmental measurements (spring moisture, spring pH, and percent open canopy) than site B (ANOVA results). For all of these environmental parameters, 1999 had lower values than 2000. Spring moisture and pH, and percent open canopy had significant

ANOVAs for site; site A was lower than site B for these environmental measurements ($P < 0.05$).

The Palmer Drought Severity Index (PDSI) values were substantially different for the three years of the study. For 1999, the PDSI values ranged from a mild drought (May, June) to moderate and severe drought (Aug to October) to extreme drought (November, December). The 2000 PDSI values ranged from normal to slightly wet for the entire year, and the 2001 PDSI values ranged from a mild drought (March, April) to very wet (October; National Climate Data Center / National Oceanic and Atmospheric Administration 2002).

Discussion

The successful restoration of a forest following the eradication of non-native plants includes the restoration of the overall diversity of the site (Sinclair et al. 1995) as well as restoring the composition to a close approximation of the original habitat (Harrington et al. 1999). Also important is the restoration of the community structure (Holmes 2001) and ecosystem level processes (Vitousek 1990). Not all of these attributes are completely restorable within a single, relatively short term restoration project, but hopefully activities will set the stage for appropriate restorative and natural trajectories.

The main goal within our experiment was one of accelerating succession (MacDonald 1993). In our case, after eradicating *L. maackii*, we desired to increase the rate of succession by overcoming the problem of limited dispersal of propagules and relatively unsuccessful recruitment, which is a frequently encountered problem in restoration efforts (Robinson and Handel 2000). Planting native species accomplished the restoration of the forest composition (and likely mid-story and canopy structure as well as recruitment processes in the future), and hopefully this will facilitate the restoration of higher-level processes; Cairns (1986) states that

most of the functional attributes of a restored system are correlated with the replacement of its vegetative structure and composition.

We also wanted to explain some of the results of the experiment to aid in future restoration efforts (for example the differential survival of the native seedlings). Seedling establishment is most importantly influenced by the existing environmental conditions during the early stages of life (Walters & Reich 2000). The differential survival of native seedlings in our experiment was largely due to individual species responses to microenvironmental conditions, which varied according to site location, *L. maackii* eradication treatment, year, and other perturbations such as fungal infection, browsing, and handling mortality.

The early mortality of *C. Canadensis* and *C. florida* was typical of a Type III survivorship pattern (Deevey 1947), which is common for establishing seedlings (De Steven 1991). Alternatively, the survivorship exhibited by *Fraxinus pennsylvanica*, *P. serotina*, and *J. nigra* at both sites and *Q. muhlenbergii* at site B was more typical of a Type II survivorship with constant mortality (Deevey 1947). No species showed late stage, Type I, mortality. Actually species-specific responses to site conditions are quite common (Sipe & Bazzaz 2001; Veenendaal et al. 1995) as is year-to-year variance in survival (Van der Meer et al. 1999). These differences in seedling survival will likely result in a forest composition which will be different than the one originally intended during initial planting; therefore, follow-up procedures such as replanting may be necessary in restorations where survival of native seedlings is relatively unpredictable and a particular forest composition is desired. Our results clearly indicate the need to restore forest systems using a diversity of species to account for environmental heterogeneity and the vagaries of climate that may dictate differential species response.

Another factor influencing the survival of seedlings was the presence or absence (through eradication) of *L. maackii*. It is not surprising that reducing *L. maackii* abundance

was associated with increased survival of native seedlings. Other restoration projects have also found greater survival following clearing of non-natives using herbicide treatments (Sweeney et al. 2002). Survival of native seedlings was clearly greater when *L. maackii* was killed by either eradication method rather than being left intact.

The mechanism for the greater native seedling survival in eradication plots is not evident because the microenvironmental measurements that we took were not statistically different between the eradication and control treatments. Significantly greater temperature, light, soil moisture, and pH levels were found (C. Keiffer 2002, personal communication) where *L. maackii* was killed versus left intact. Greater light intensity, although not significant, may account for the increased survival that we found in eradication plots. Work by Trisel (1997) may offer two possible explanations for lower seedling survival in *L. maackii* control versus removal plots. He found that native seedlings had increased mortality when grown with *L. maackii* due mostly to shoot competition, but root competition may also play a role. He suspected this aboveground competition would be mostly competition for light. He also found that *L. maackii* plants may be allelopathic in that watering with *L. maackii* leaf extract had effects similar to those of a 10^{-4} solution of juglone, a known allelopathic chemical. Generally, seedlings in *L. maackii* control subplots may have been chemically inhibited or had fewer resources than in areas where *L. maackii* was eradicated. Clearly more work needs to be conducted to elucidate the exact nature of *L. maackii*'s interference mechanism.

In addition to the two eradication treatments that we used, others have employed several other methods to control *L. maackii* with varying levels of success. A 1% foliar glyphosate spray has been used to control seedlings as well as to control adult *L. maackii* along heavily invaded edge habitats. The advantage of foliar spraying is that it is relatively

easy to apply (Conover and Geiger 1993); however, it can result in up to 100% mortality to native vegetation in the underlying herb layer (Trisel 1997). Others have argued similarly against the use of herbicides in restoration areas because of negative effects on non-target plants (Brockway et al. 1998).

Cutting alone has been found to be a less than adequate control method for *L. maackii* because resprouting occurs from a meristematic burl located at the base of stems (Luken 1988). Although resprouting following clipping occurs less frequently in forested versus open habitats (30% v. 70%, respectively), clipping alone has been found to actually increase stem numbers (Luken 1990). We do not recommend clipping alone without an application of herbicide in any type of habitat.

Cutting and herbiciding was found to be effective in our experiment. Cutting and herbiciding (i.e., cutting and painting) is one of the most widely used eradication procedures for woody invasive plants (Reinartz 1997; Olson & Whitson 2002). A 20% glyphosate solution has been found to be effective in controlling *L. maackii* in forest interiors, and a 50% solution is more appropriate in open habitats where the plant seems more resistant (T. Borgman 2002, personal communication). Our experiment successfully controlled Amur honeysuckle using a 50% solution for the cut and paint treatment in a young closed-canopy forest.

Cutting alone and cutting and herbicide methods leave behind stump bases, which may take a number of years to decay. However, cutting (and herbiciding) has the advantage of creating wildlife habitat if the stems are placed in piles rather than being chipped or removed (J. Klein 2002, personal communication). The biggest disadvantage of eradication involving *L. maackii* cutting is that it is very labor intensive.

A number of other *L. maackii* eradication methods have been reported, but each has its own drawbacks. Hand pulling can be effective in areas with moist ground, but the plants will likely resprout if root portions remain (Gayek1990). Generally hand pulling is not difficult if individuals are less than three years old or are growing on moist soils, but the spread of *L. maackii* from remaining roots may not make this method effective (Gayek 2000). Other methods for controlling *L. maackii* include using a "weed wrench" to remove whole crowns (100% mortality) or a polaski axe (98% mortality); however, these methods are very labor intensive (Trisel 1997).

We found that the injection system may be the best overall method for the eradication of *L. maackii*. There are several advantages to its use. Injecting produced very little operator fatigue and limits exposure of the operator to herbicide. Also less overall herbicide is used relative to other methods, and the herbicide that is used is restricted entirely to target plants. We also found that injecting was 43% faster than cutting and herbiciding. This may be of considerable importance in larger restoration efforts. Herbicide injections have successfully been used by others to kill woody plants and prevent regrowth (Johansson 1985). Franz and Keiffer (2000) successfully used the injection system to eradicate *L. maackii* in a stand in southwest Ohio. They found that fall injections are more effective than spring, and that it is more effective to inject all stems, rather than a single stem, of a *L. maackii* plant.

Other considerations in restorations of this type include the use of tree protectors, particularly where large mammal browsing is a problem. Sweeney et al. (2002) found tree seedlings survived best when a combination of herbiciding and tubing treatments were applied; however, we did not find that tree protectors increased survival or growth of native seedlings. We found that despite there being environmental differences in tubed and non-

tubed conditions, there were no significant survival or growth differences for native seedlings. Mortality due to browsing was only 3.1% at this site; however, it is widely known that deer as a keystone herbivore can have profound impacts on forest composition (Rooney 2001). Perhaps if the deer populations are a problem locally, then tree protectors would be justified.

It is best to target control of *L. maackii* populations when they are small. Deering and Vankat (1999) recommend that early control is best because its done before plants reach reproductive maturity, which is roughly 4 or 5 years. This approach does have merit as the early control of invasives is often less costly and has a greater chance of success than later control (Chippendale 1991). Reinvasion of non-natives is quite common and has been reported in a number of restoration studies (Hurst & John 1999). Non-native plant control is done most effectively when all plants are eradicated from an area, and this is the case with *L. maackii*. When total eradication is accomplished, *L. maackii* must repopulate from another site, and reinvasion is likely to occur in small, manageable amounts (Deering & Vankat 1999).

Although restoration costs are rarely reported in academic articles, the economics of a restoration project are very important (Holl and Howarth 2000). While often seemingly expensive, the cost of eradicating and restoring native communities is often recovered in a short time (Zavaleta 2000). Various restoration studies have shown that the cost of the removal of *L. maackii* can be quite variable. Gayek (2000) reported the cost of a *L. maackii* restoration to be $8200 for a single two hectare project which included labor for a crew of 30 as well as safety equipment, 28.5 liters of herbicide, and the rental of a brush chipper. Gayek reported $2000 for the cost of another removal project on four hectares of land, which was less expensive because volunteers were

used and no brush chipper was employed. Trisel (1997) indicated lower costs of *L. maackii* eradication. He gave the startup costs for three methods of *L. maackii* control including crown removal using a polaski axe and hand saw ($42), foliar spray using herbicide and backpack sprayer ($165), and the stem cut and paint method using clippers, loppers, and herbicide ($172). However, he did not include the cost of labor in his estimates or account for per hectare area costs. Additionally, he states that these methods were either labor intensive or damaging to native plants.

For our project, cutting and painting was less expensive ($253) in terms of startup costs than the injection method ($599); however, the most expensive part of restoration would most likely be labor for the implementation of eradication treatments and planting of native seedlings. The amount of time required for our experiment to inject an area was roughly 43% less than the cut and paint treatment. Clearly labor is a major cost to restorations of this type and should be considered in any restorative effort.

Restoring human-damaged ecosystems without intervention is often difficult (Ghersa et al. 2002). Revegetating sites following the removal of dominant non-native species is particularly complex. When existing seed banks and dispersal opportunities are limited, restoration involving seed or seedling reintroduction is one way to accelerate succession (Zedler 2000). We found that limited recruitment below *L. maackii* stands necessitated eradicating *L. maackii* and replacing it with native tree seedlings. This was successfully done at Fernald, and the restoration effort yielded increased woody plant composition, structure, and diversity. Likely, this will facilitate larger-scale processes in the future as these species mature.

Lonicera maackii highlights a larger problem. The best situation would be to predict successful invaders prior to introduction and prevent their establishment at all.

However with few restrictions on introductions, invaders frequently do enter communities and become problematic (Reichard & Hamilton 1997). It then becomes the task of land managers and ecologists to work cooperatively to gather information on the impacts of invasives, invasion patterns, traits of invaded communities, how to most effectively control non-natives, and at the same time, protect native biodiversity (Byers et al. 2002). We believe that by combining the findings of our research with prior studies, many of these notions would come to light which could be applied to bring about successful ecological restorations, particularly in those regions of the country affected by *Lonicera maackii*.

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References

Barnes, B. V., D. R. Zak, S. R. Denton, and S. H. Spurr. 1998. *Forest* ecology, 4th ed. Wiley Press, New York.

Bergerud, W. A. 1988. Dose-response models for stand thinning with the "Ezject" herbicide injection system. Research Note No. 102. British Columbia Ministry of Forests and Lands. Victoria, British Columbia.

Braun, E. L. 1989. The woody plants of Ohio: trees, shrubs and woody climbers, native, naturalized, and escaped. Ohio State University Press, Columbus, Ohio.

Brockway, D. G., K. W. Outcalt, and R. N. Wilkins. 1998. Restoring longleaf pine wiregrass ecosystems: plant cover, diversity and biomass following low-rate hexazinone application on Florida sandhills. Forest Ecology and Management **103**: 159-175.

Byers, J. E., S. Reichard, J. M. Randall, I. M. Parker, C. S. Smith, W. M. Lonsdale, I.A. E. Atkinson, T. R. Seastedt, M. Williamson, E. Chornesky, D. Hayes. 2002. Directing research to reduce the impacts of nonindigenous species. Conservation Biology **16**: 630-640.

Cairns, J., Jr. 1986. Restoration, reclamation, and regeneration of degraded or destroyed ecosystems. Pages 465-484 in M. E. Soule, editor. Conservation biology. Sinauer Publishers, Ann Arbor, Michigan.

Canham, C. D. 1988. An index for understory light levels in and around canopy gaps. Ecology **69**: 1634-1638.

Chippendale, J. F. 1991. Potential returns to research on rubber vine (*Cryptostegia grandiflora*). Master's Thesis, University of Queensland, Brisbane, Australia.

Cicchetti D. V. 1972. Extension of multiple-range tests to interaction tables in the

analysis of variance: A rapid approximate solution. Psychological Bulletin **77**: 405-408.

Collier, M. H., J. L. Vankat, and M. R. Hughes. 2002. Diminished plant richness and abundance below *Lonicera maackii*, an invasive shrub. American Midland Naturalist **147**: 60-71.

Conover, D. G., and D. R. Geiger 1993. Glyphosate controls Amur honeysuckle in native woodland restoration (Ohio). Restoration and Management Notes **11**: 168-169.

Crowder, M. J., and D. J. Hand. 1990. Analysis of repeated measures. Chapman and Hall, London.

D'Agostino R. B., A. Balanger, and R. B. D'Agostino, Jr. 1990. A suggestion for using powerful and informative tests of normality. American Statistitian **44**: 316-321.

De Steven, D. 1991. Experiments on mechanisms of tree establishment in old-field succession: Seedling survival and growth. Ecology **72**: 1076-1088.

Deering, R. H., and J. L. Vankat. 1999. Forest colonization and developmental growth of the invasive shrub *Lonicera maackii*. American Midland Naturalist **141**: 43-40.

Deevey, E. S., Jr. 1947. Life tables for natural populations. The Quarterly Review of Biology **22**: 283-314.

Dodson, B. 1994. Weibull Analysis. ASQC Quality Press. Milwaukee, Wisconsin.

Dowdy, S. and S. Wearden. 1991. Statistics for research, 2nd ed. John Wiley and Sons, New York.

Franz, C. M. And C. H. Keiffer. 2000. Effectiveness of the EZJECT capsule injection system against the invasive shrub, *Amur Honeysuckle*. The Ohio Woodland Journal **7**:19-20.

Gayek, A. 2000. Effect of habitat variation on rate and success of colonization of two

invasive shrubs, *Lonicera maackii* and *Ligustrum vulgare*, in a forested glen. Master's Thesis, Department of Horticulture and Crop Science. Ohio State University, Columbus, Ohio.

Ghersa, C. M., E. de la Fuenta, S. Suarez, R. J. C. Leon. 2002. Woody species invasion in the Rolling Pampa grasslands, Argentina. Agriculture, Ecosystems, and Environment **88**: 271-278.

Gould, A. M. A., and D. L. Gorchov. 2000. Effects of the exotic invasive shrub *Lonicera maackii* on the survival and fecundity of three species of native annuals. American Midland Naturalist **144**: 36-50.

Harrington, C. A. 1999. Forests planted for ecosystem restoration or conservation. New Forests **17**: 175-190.

Hintze, J. L. 2000. Number Cruncher Statistical Systems (NCSS). 2000 User's Guide. Kaysville, Utah.

Hobbs, R. J., and L. F. Huenneke. 1992. Disturbance, diversity, and invasion: Implications for conservation. Conservation Biology **6**: 324-337.

Holl, K. D., and R. B. Howarth. 2000. Paying for restoration. Restoration Ecology **8**: 260-267.

Holmes, P. M. 2001. Shrubland restoration following woody alien invasion and mining: effects of topsoil depth, seed source, and fertilizer addition. Restoration Ecology **9**: 71-83.

Hurst, A., E. John. 1999. The effectiveness of glyphosate for controlling *Brachypodium pinnatum* in chalk grassland. Biological Conservation **89**: 261-265.

Hutchinson, T. F., and J. L. Vankat. 1997. Invasibility and effects of Amur honeysuckle in southwestern Ohio forests. Conservation Biology **11**: 1117-1124.

Johansson, T. 1985. Herbicide injections into stumps of aspen and birch to prevent regrowth. Weed Research **25**: 39-45.

Lerch, N. K., W. F. Hale, and D. D. Lemaster. 1980. Soil survey of Butler County, Ohio. United States Department of Agriculture, Soil Conservation Service. Washington, D.C.

Luken, J. O. 1988. Population structure and biomass allocation of the naturalized shrub *Lonicera maackii* (Rupr.) Maxim. in forest and open habitats. American Midland Naturalist **119**: 258-267.

Luken, J. O. 1990. Forest and pasture communities respond differently to cutting of exotic Amur honeysuckle (Kentucky). Restoration and Management Notes **8**: 122-123.

Luken, J. O., and J. W. Thieret. 1995. Amur honeysuckle (*Lonicera maackii*; Caprifoliaceae): its ascent, decline, and fall. Sida **16**: 479-503.

Luken, J. O. and J. W. Thieret. 1996. Amur honeysuckle: its fall from grace. BioScience **46**: 18-24.

MacDonald, A. W. 1993. The role of seedbank and sown seeds in the restoration of an English flood-meadow. Journal of Vegetation Science **4**: 395-400.

McCarthy, B. C. 1997. Lab protocols for the testing of eastern deciduous forest soils. http://www.plantbio.ohiou.edu/epb/soils/soils.htm

McCarthy, B. C. 1999. Identification and Control of Invasive Species. Annual report to Fernald Environmental Management Project. Cincinnati, Ohio.

Montalvo, A. M., P. A. McMillan, and E. B. Allen. 2002. The relative importance of seeding method, soil ripping, and soil variables on seeding success. Restoration Ecology **10**: 52-67.

National Climate Data Center and National Oceanic and Atmospheric Administration. 2002. CLIMVIS Information: Selection criteria for displaying period of record. http://www5.ncdc.noaa.gov/cgi-bin/climatenormals/climatenormals.pl

Nyboer, R. 1992. Vegetation management guideline: bush honeysuckles - Tatarian, Morrow's Belle, and Amur honeysuckle (*Lonicera tatarica* L., *L. morrowii* Gray, *L.* x *bella* Zabel, and *L. maackii* (rupr.) Maxim.). Natural Areas Journal **12**: 218-219.

Olson, R. A., and T. D. Whitson. 2002. Restoring structure in late-successional sagebrush communities by thinning with tebuthiuron. Restoration Ecology **10**: 146-155.

Pyke, D. A., and J. N. Thompson. 1986. Statistical analysis of survival and removal rate experiments. Ecology **67**: 240-245.

Reichard, S. H. and C. W. Hamilton. 1997. Predicting invasions of woody plants introduced into North America. Conservation Biology **11**: 193-203.

Reifsnyder, W. E., and H. W. Lull. 1965. Radiant energy in relation to forests. United States Department of Agriculture, Forest Service Technical Bulletin. No. 1344.

Reinartz, J. A. 1997. Controlling glossy buckthorn (*Rhamnus frangula* L.) with winter herbicide treatments of cut stumps. Natural Areas Journal **17**: 38-41.

Robison, S., and B. C. McCarthy. 1999. Potential factors affecting the estimation of light availability using hemispherical photography in oak forest understories. Journal of the Torrey Botanical Society **126**: 344-349.

Robinson, G. R. and S. N. Handel. 2000. Directing spatial patterns of recruitment during an experimental urban woodland reclamation. Ecological Applications **10**: 174-188.

Rooney, T. P. 2001. Deer impacts on forest ecosystems: A North American perspective. Forestry **74**: 201-208.

SAS Institute. 2001. SAS / STAT User's Guide, Version 5.0. SAS Institute, Cary, North Carolina.

Scheiner, S. M. 1993. MANOVA: Multiple response variables and multispecies interactions. Pages 94-112 in S. M. Scheiner and J. Gurevitch, editors. Design and analysis of ecological experiments. Chapman and Hall, New York.

Schierenbeck, K. A., R. N. Mack, and R. R. Sharitz. 1994. Effects of herbivory on growth and biomass allocation in native and introduced species of *Lonicera*. Ecology **75**: 1661-1672.

Sinclair, A. R. E., and D. S. Hik, O. J. Schmitz, G. G. E. Scudder, D. H. Turpin, and N.C. Larter. 1995. Biodiversity and the need for habitat renewal. Ecological Applications **5**: 579-587.

Sipe, T. W. and F. A. Bazzaz. 2001. Shoot damage effects on regeneration of maples (*Acer*) across an understory-gap microenvironmental gradient. Journal of Ecology **89**: 761-773.

Shea, K. and P. Chesson. 2002. Community ecology theory as a framework for biological invasions. Trends in Ecology and Evolution **17**: 170-176.

Sokal, R. R., and F. J. Rohlf. 1995. Biometry: The principles and practice of statistics in biological research. W. H. Freeman, New York.

Sweeney, B. W., S. J. Czapka, and T. Yerkes. Riparian forest restoration: increasing success by reducing plant competition and herbivory. Restoration Ecology **10**: 392-400.

Trisel, D. E. 1997. The invasive shrub, *Lonicera maackii* (Rupr.) Herder (Caprifoliaceae) factors leading to its success and its effect on native species. Ph.D. Dissertation. Department of Botany. Miami University, Oxford, Ohio.

U.S. Department of Energy. 2002. 2001 Site Environmental Report. Fernald Environmental Management Project, U.S. Department of Energy, Fernald Area Office, Cincinnati, Ohio.

United States Geologic Survey. 1981. Shandon (OH) quadrangle, 7.5 minute series (topographic) map, 1: 24000 scale. USGS, Reston, Virginia.

Van Der Meer, P. J., P. Dignan, and A. G. Saveneh. 1999. Effect of gap size on seedling establishment, growth and survival at three years in mountain ash (*Eucalyptus regnans* F. Muell.) forest in Victoria, Australia. Forest Ecology and Management **117**: 33-42.

Von Ende, C. N. 1993. Repeated-measures analysis: Growth and other time-dependent measures. Pages 113-137 in S. M. Scheiner and J. Gurevitch, editors. Design and analysis of ecological experiments. Chapman and Hall, New York.

Walters, M. B., and P. B. Reich. 2000. Seed size, nitrogen supply, and growth rate affect tree seedling survival in deep shade. Ecology **81**: 1887-1901.

Wiens, J. A., N. C. Stenseth, B. Van Horne, and R. A. Ims. 1993. Ecological mechanisms and landscape ecology. Oikos **66**: 369-380.

Woods, K. D. 1993. Effects of invasion by *Lonicera tatarica* L. on herbs and trees in four New England forests. American Midland Naturalist **130**: 62-74.

Zavaleta, E. 2000. The economic value of controlling an invasive shrub. Ambio **29**: 462-467.

Zedler, J. B. 2000. Progress in restoration ecology. Trends in Ecology and Evolution **15**: 402-407.

Table 1. Analysis of variance of final survival for native tree seedlings.

Source Term	DF	SS	MS	F-ratio	P	Power (alpha=0.05)
Site	1	4.438	4.438	84.07	< 0.001	
Treatment	2	1.741	0.704	331.04	0.003	1.000
Control vs Treatments *	1			25.27	0.002	
Cut vs Inject *	1			8.23	0.014	
Site x Treatment	2	0.005	0.003	0.05	0.951	
Species	5	8.339	0.668	6.65	0.029	0.771
Site x Species	5	1.253	0.251	4.75	< 0.001	
Treatment x Species	10	0.403	0.040	0.43	0.902	0.129
Site x Treatment x Species	10	0.941	0.094	1.78	0.064	
S	52	13.304	0.053			
Total (Adjusted)	28	31.917				
Total	288					

*orthogonal contrasts

Figure 1. Final survival of planted seedlings comparing sites across treatments (panels A - C). Species abbreviations are as follows: *Cercis canadensis* (CECA), *Cornus florida* (COFL), *Fraxinus pennsylvanica* (FRPE), *Juglans nigra* (JUNI), *Quercus muhlenbergii* (QUMU), *Prunus serotina* (PRSE). Note: all species had greater survival at site B relative to site A, except *F. pennsylvanica*, which had greater survival in the two eradication treatments of Site A (panels B and C).

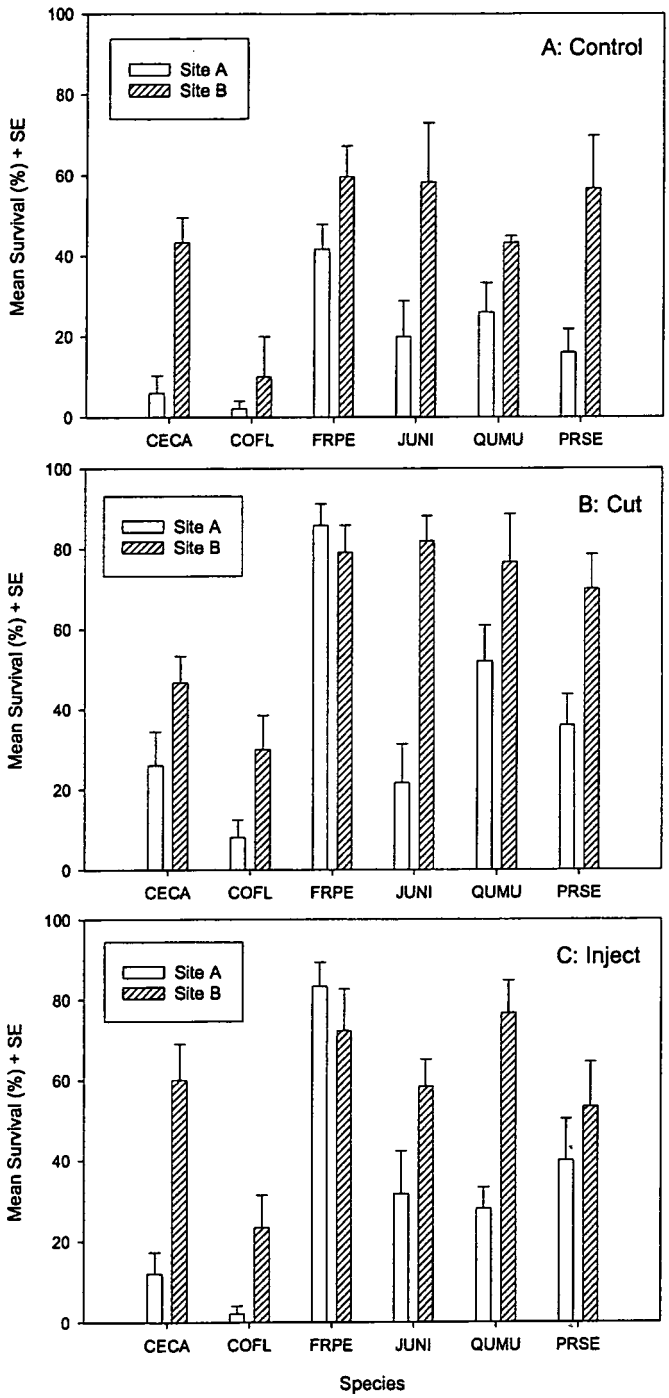
Figure 2. Final survival (after three years) among eradication treatments. Lowercase letters indicate significant differences ($P < 0.05$).

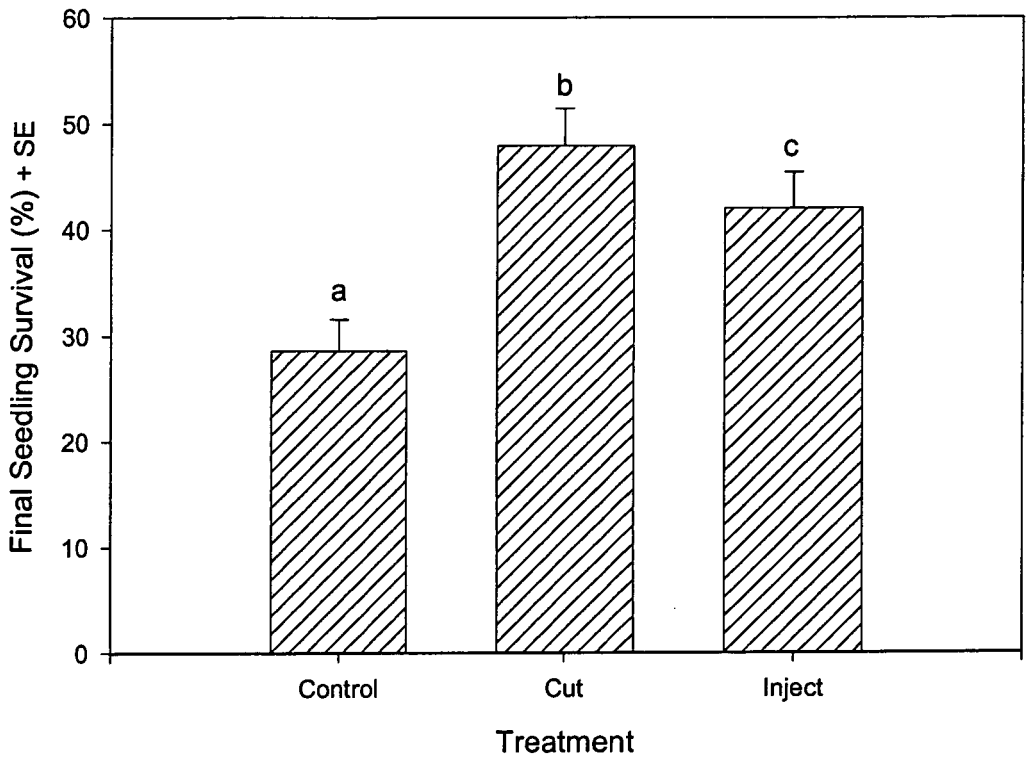
Figure 3. Weibull distribution function estimates for native seedling survival rates in three eradication treatments. Survival rates in the two eradication treatments was greater than in the control treatment plots where Amur honeysuckle was left intact. Lowercase letters indicate significant differences ($P < 0.05$) among treatments.

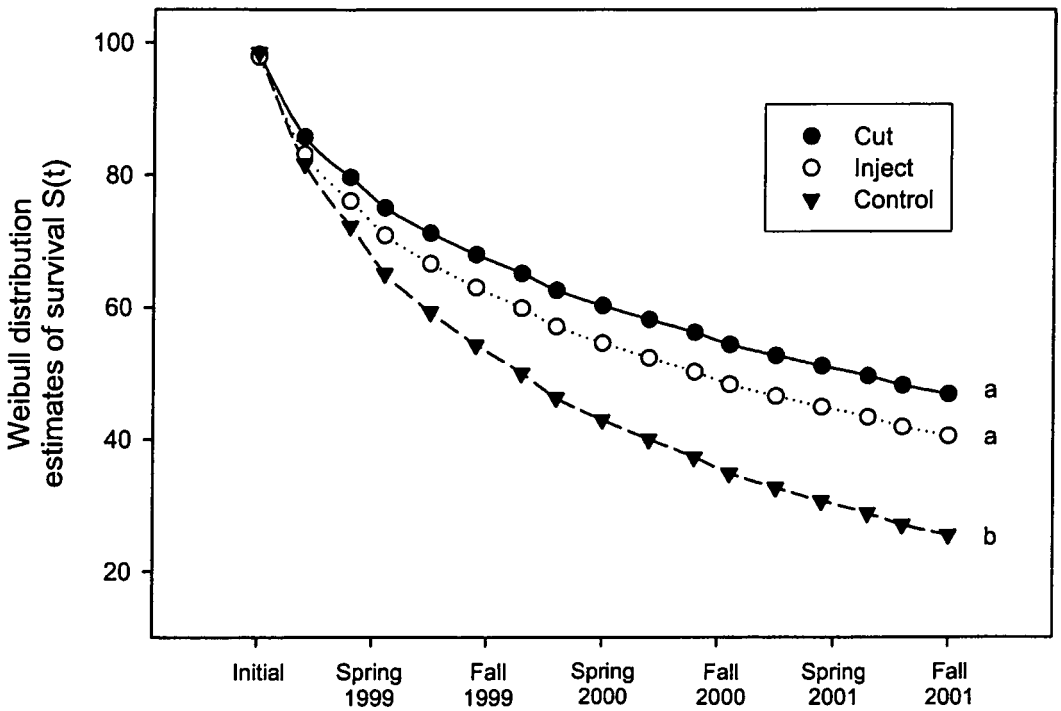
Figure 4. Weibull distribution function estimates for native species survival rates in two sites. Overall survival rates at site B was greater than at site A ($P < 0.001$). Lower case letters indicate significant ($P < 0.05$) differences among species within a site. See Figure 1 for species abbreviations.

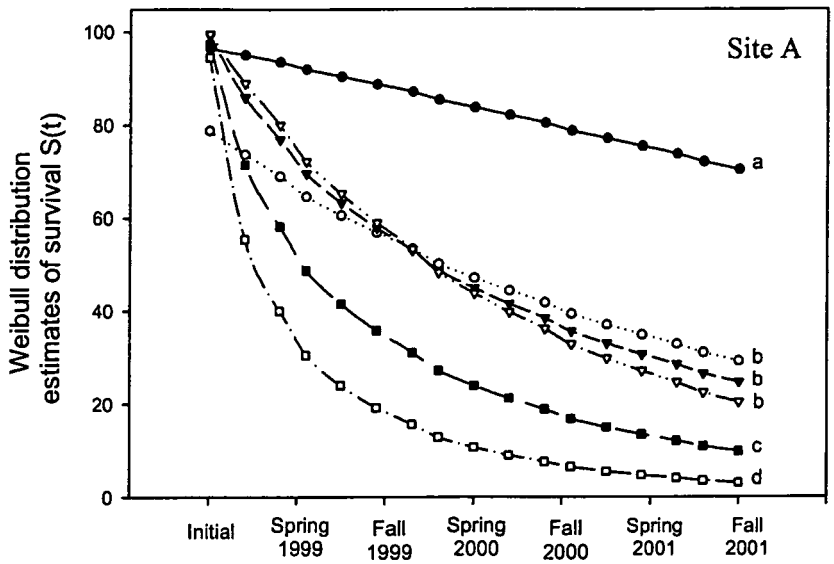
Figure 5. Seedling height growth for 1999 – 2000. Lower case letters denote significant ($P < 0.05$) differences among species within a single year. No significant differences ($P < 0.05$) existed among species for 2000. See Figure 1 for species abbreviations.

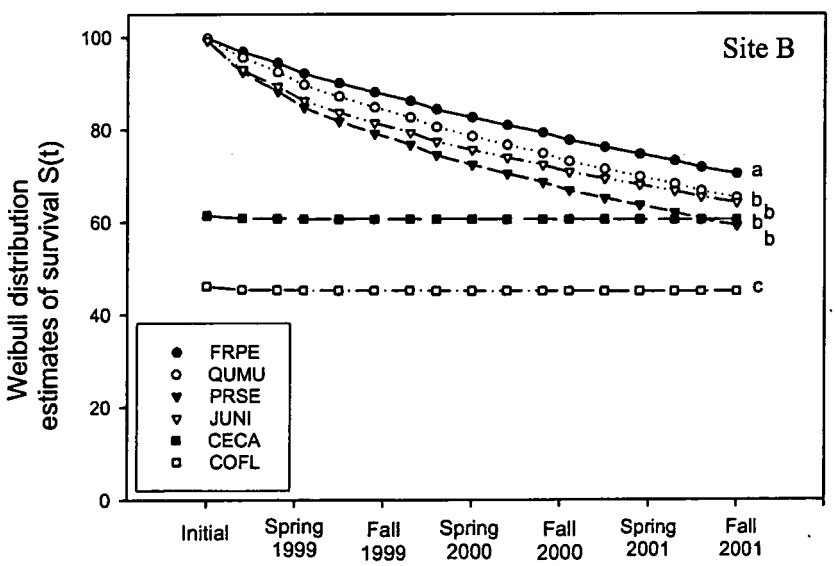
Figure 6. Results of seedling diameter growth for 1999 – 2000. Lower case letters denote significant differences among species within a single year. No significant differences ($P < 0.05$) existed among species for 1999 and 2000. See Figure 1 for species abbreviations.

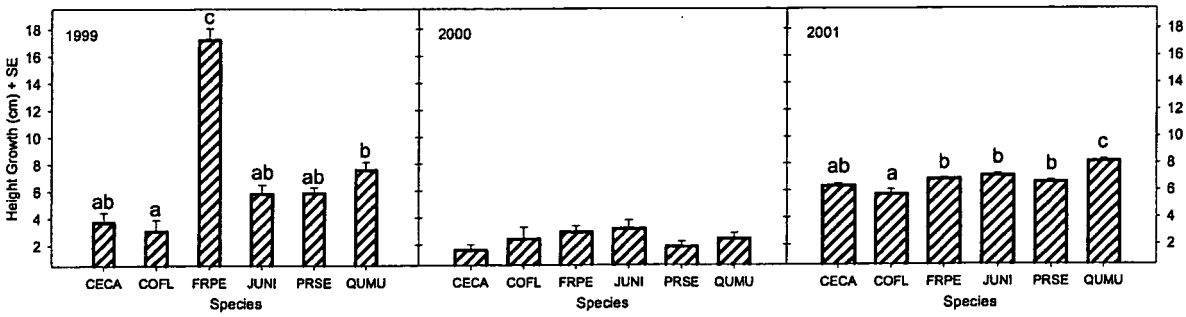

A: Control
B: Cut
C: Inject
Site A
Site B
Mean Survival (%) + SE
0
20
40
60
80
100
CECA
COFL
FRPE
JUNI
QUMU
PRSE
Species

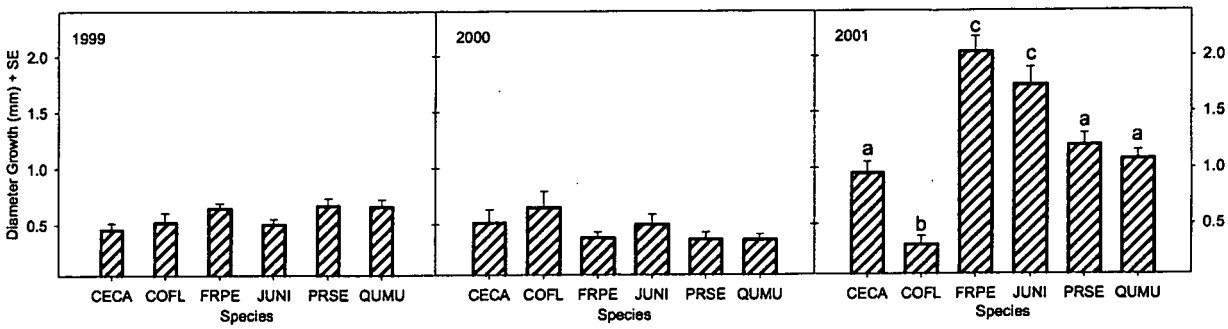

60
50
40
30
20
10
0
Final Seedling Survival (%) + SE
a
b
c
Control
Cut
Inject
Treatment


Weibull distribution estimates of survival S(t)
100
80
60
40
20
Cut
Inject
Control
a
a
b
Initial
Spring 1999
Fall 1999
Spring 2000
Fall 2000
Spring 2001
Fall 2001


Site A
Weibull distribution estimates of survival S(t)
100
80
60
40
20
0
a
b
b
b
c
d
Initial
Spring 1999
Fall 1999
Spring 2000
Fall 2000
Spring 2001
Fall 2001


Site B
Weibull distribution estimates of survival S(t)
100
80
60
40
20
0
a
b
b
b
b
c
FRPE
QUMU
PRSE
JUNI
CECA
COFL
Initial
Spring 1999
Fall 1999
Spring 2000
Fall 2000
Spring 2001
Fall 2001


1999
2000
2001
Height Growth (cm) + SE
18
16
14
12
10
8
6
4
2
ab
a
c
ab
ab
b
ab
a
b
b
b
c
CECA
COFL
FRPE
JUNI
PRSE
QUMU
Species


1999
2000
2001
Diameter Growth (mm) + SE
2.0
1.5
1.0
0.5
a
b
c
c
a
a
CECA COFL FRPE JUNI PRSE QUMU
Species