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RESTORING FOREST IN WETLANDS DOMINATED BY REED CANARYGRASS: THE EFFECTS OF PRE-PLANTING TREATMENTS ON EARLY SURVIVAL OF PLANTED STOCK

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Abstract: Reed canarygrass (*Phalaris arundinacea* L.) is an aggressive and persistent invasive species in formerly forested wetlands of the northern United States. Heavy shading reduces the dominance of reed canarygrass, so a promising long-term approach to restoration of reed canarygrass–dominated wetlands is the establishment of woody plants that will overtop and shade the grass. The first step toward developing this long-term restoration method is to determine a combination of reed canarygrass control methods and suitable trees and shrubs to provide high early survival of the native woody plants. We tested 23 tree and shrub species in five treatments to determine: 1) the woody species that have the highest survival when planted in treated stands of reed canarygrass, and 2) the pre-planting treatments that lead to the highest rates of survival. Near-monocultures of reed canarygrass were herbicided, mowed and herbicided, herbicided and plowed, or herbicided and burned. One- to three-year-old, mostly bare-rooted trees and shrubs were hand-planted into these treatments and into untreated control plots at three sites, and over two growing seasons. Fall herbicide followed by spring plowing provided the highest survival for the majority of species planted. However, all experimental treatments (controlling reed canarygrass with a single herbicide application) provided reasonably high survival of the 10 most successful woody species. Those pre-planting treatments and study sites that developed the greatest herbaceous species diversity after treatment had the highest tree and shrub survival. The early establishment success we found using these methods is encouraging for development of a technique for restoring swamp forest in degraded reed canarygrass–dominated wetlands.

Key Words: glyphosate, herbicide, *Phalaris arundinacea*, swamp forest, wetland restoration

INTRODUCTION

Invasive species threaten native species, communities, and environments (Wilcove et al. 1998, Mack et al. 2000) and are among the primary concerns for natural area managers (Randall and Rice 2003). The worst invaders can decrease the value of wildlife habitat, alter ecosystem functions and disturbance regimes, and displace native species of plants and animals (D'Antonio and Vitousek 1992, Pimentel et al. 2000). Invasive species can displace native vegetation and suppress establishment of native flora through competitive dominance (Apfelbaum and Sams 1987, Galatowitsch et al. 1999, Mulhouse and Galatowitsch 2003, Spyreas et al. 2004, Zedler and Kercher 2004). Aggressive invasive species are particularly troublesome for restoration efforts. Adequate sites for seed germination, seedling growth, and establishment of vegetative propagules, both at the beginning of a restoration project and during subsequent aftercare, are necessary for normal community function to resume and for recruitment to occur naturally (Urbanska 1997).

Grasses in particular can limit the establishment and growth of woody species (D'Antonio and Vitousek 1992, Hess et al. 1999, Mazia et al. 2001, Spyreas et al. 2004). D'Antonio and Vitousek (1992) list nearly 40 non-native grass species that are serious problems somewhere around the world. The competitive advantages of grasses include effective competition for water and nutrients, soil-surface light reduction, changes to the historical fire regime, and a number of ecosystem-level effects (D'Antonio and Vitousek 1992). Some grass-dominated communities in areas that were historically forested are especially resistant to a reinvasion of the original vegetation, presumably because of a lack of establishment sites (e.g., Guariguata et al. 1995, Hill et al. 1995, Chapman and Chapman 1999, Hess et al. 1999, Mazia et al. 2001).

There has been considerable research on effective methods for restoration of wetland ecosystems, but little attention has been given to restoration of forested wetlands. Most of the work on forested wetland restoration has focused on communities in the Mississippi Delta region (Mitsch and Gosselink 1993, Battaglia et al. 2002) or on swamps in the southeastern United States (Clewel and Lea 1989, McLeod et. al. 2000). There has been very little work on restoring the swamp forests of the northern states. Many of these northern swamps were converted to agricultural use and planted with reed canarygrass (*Phalaris arundinacea* L.) for use as forage or for erosion control in poorly drained soils.

Reed canarygrass is a rhizomatous, cool-season, perennial grass that tends to form monocultures (Apfelbaum and Sams 1987), decreases species diversity (Lesica 1997), and reduces the structural complexity of the plant community (Barnes 1999) through its aggressive, rapid growth and ability to eliminate safe sites for the establishment of native plants (Paine and Ribic 2002). It is widely distributed in northern temperate regions, and has been bred for erosion control, forage and, recently, as a renewable fuel source (Alway 1931, Sahramaa et al. 2003, Lavergne and Molofsky 2004). Although it is native to the United States and Canada (Merigliano and Lesica 1998), most strains of *P. arundinacea* commonly encountered in North America act as a typical invasive plant (Lavergne and Molofsky 2004) and are thought to have originated as cultivars bred for high productivity and stress tolerance, or as hybrids between native genotypes and those cultivars (Merigliano and Lesica 1998, Galatowitsch et al. 1999). Where planted in wetlands, reed canarygrass can maintain nearly monospecific stands for long periods (Apfelbaum and Sams 1987).

High genetic diversity and phenotypic plasticity (Wilkins and Hughes 1932, Anderson 1961, Green and Galatowitsch 2001, Gifford et al. 2002, Lavergne and Molofsky 2004) allow reed canarygrass to thrive in a wide range of conditions (Morrison and Molofsky 1998, 1999). However, shade has been consistently found to limit its spread (Thompson 1995), germination (Lindig-Cisneros and Zedler 2002a, b), and growth (Kephart et al. 1992, Perry and Galatowitsch 2004). This Achilles' heel bodes well for the long-term prospects of swamp forest restoration in wetlands dominated by reed canarygrass. Additional research is needed on practical methods for restoration of reed canarygrass meadows to swamp forests. The first and most crucial step is to learn how to establish trees and shrubs by the creation of suitable safe sites to ensure substantial woody plant survival.

One of Urbanska's (1997) main requirements for a successful restoration is that safe sites be present for the initial growth and establishment of native species. We have some evidence that trees and shrubs can be successfully established in reed canarygrass monocultures following effective herbicide application, and that 10 to 20 years after such establishment the reed canarygrass can be nearly eliminated by shading (J. Reinartz, unpublished data). We describe here a two-year study to demonstrate the effectiveness of planting native trees and shrubs in reed canarygrass as a first step towards restoration of swamp forest. We compare five different pre-planting treatments and 23 tree and shrub species to determine: 1) relative survival of 11 tree and 12 shrub species planted in reed canarygrass following various pre-planting treatments, and 2) which pre-planting treatments lead to the highest woody species survival and initial growth.

METHODS

Study Species

We planted 11 tree and 12 shrub species (Table 1) that are common in local swamp forest or shrub carr communities spanning a wide range of successional stages. All but two of the planted species have a wetland indicator status of Facultative (FAC) or Facultative Wetland (FACW) (Reed 1988). Paper birch (*Betula papyrifera* Marshall) and basswood (*Tilia americana* L.) are classified as Facultative Upland (FACU), but in southeastern Wisconsin both species tolerate moist soils and are often found in wetlands.

Most species were planted as bare-rooted individuals (Table 1). Three species planted in 2003 were rooted plugs (15 cm long $\times$ 2.5 cm diameter), and all willows (*Salix* spp.) were planted as live stakes (cuttings 8–30 mm diameter, 40–60 cm length). All stock planted in 2004 was bare-rooted except for the three willow species.

Since first-year survival of willow stakes planted in 2003 was disappointingly low (3.8% to 9.2%), we planted the same three species again in 2004 using different collection and storage protocols. Equal numbers of Bebb's willow, pussy willow, and slender

Table 1. Trees and shrubs planted in reed canarygrass. Stock type abbreviations are as follows: BR = Bareroot; PL = Plug; and LS = Live stake. Live stakes were either collected (C) or purchased (P). Huiras was planted in spring 2003, Winker and Fellenz in spring 2004. Nomenclature follows Gleason and Cronquist (1991). Wetland indicator status is from Reed (1988).

			Number Planted			
Scientific Name	Common Name	Stock Type	Huiras	Winker	Fellenz	Wetland Indicator Status
Trees:						
Acer rubrum	Red maple	BR	488	60	60	FAC
Acer saccharinum	Silver maple	BR	108	40	40	FACW
Betula alleghaniensis	Yellow birch	PL	93	—	—	FAC
		BR	—	60	60	
Betula papyrifera	Paper birch	BR	125	40	40	FACU+
Fraxinus nigra	Black ash	BR	216	40	40	FACW+
Fraxinus pennsylvanica	Green ash	BR	658	60	60	FACW
Larix laricina	Tamarack	BR	576	60	60	FACW
Populus tremuloides	Quaking aspen	BR	70	—	—	FAC
Thuja occidentalis	White cedar	BR	576	60	60	FACW
Tilia americana	Basswood	PL	52	—	—	FACU
		BR	—	40	40	
Ulmus americana	American elm	PL	49	—	—	FACW−
Shrubs:						
Cornus amomum	Silky dogwood	BR	108	40	40	FACW+
Cornus racemosa	Gray dogwood	BR	49	40	40	FACW−
Cornus sericea	Red-osier dogwood	BR	621	60	60	FACW
Ilex verticillata	Winterberry	BR	—	60	60	FACW+
Ribes americanum	Black currant	BR	49	—	—	FACW
Rubus occidentalis	Black raspberry	BR	49	—	—	
Salix bebbiana	Bebb's willow	LS—P	26	20	20	FACW+
		LS—C	—	20	20	
Salix discolor	Pussy willow	LS—P	238	36	36	FACW
		LS—C	—	36	36	
Salix petiolaris	Slender willow	LS—P	245	36	36	FACW+
		LS—C	—	36	36	
Sambucus canadensis	Elderberry	BR	70	28	28	FACW−
Viburnum lentago	Nannyberry	BR	129	—	—	FAC+
Viburnum opulus var. *americanum*	Highbush cranberry	BR	73	32	32	FACW

willow (*Salix bebbiana* Sarg., *S. discolor* Muhl., and *S. petiolaris* J. E. Smith) stakes were purchased from a wholesale nursery and collected from local plants. We harvested the locally collected stakes according to published recommendations (Gray and Leiser 1982) and stored them submerged in water until planting (less than 48 hours). The purchased willow stakes were delivered packed in moist mulch. The stakes we collected of all three species were significantly larger in diameter than the purchased cuttings.

Study Sites

We planted woody species in three reed canarygrass stands: Huiras Lake, the Winker Property, and Fellenz Woods. The Huiras Lake study site (Ozaukee County, Wisconsin, T.12N., R.21E., SE 1/4, SE 1/4, Sec. 9) was 1.2 hectares, with a near monoculture of reed canarygrass that was planted more than 35 years ago. Huiras Lake soil was Pella silt loam overlain by a surface layer of peaty-muck (Parker et al. 1970). Wetlands adjacent to the study site were dominated by a native red maple (*Acer rubrum* L.) hardwood swamp and contained patches of white cedar (*Thuja occidentalis* L.) and tamarack (*Larix laricina* [Duroi] K. Koch) conifer swamp. Outside of our study area, the reed canarygrass decreased in dominance near the margins of the adjacent forest, grading into a species-poor wet meadow community.

The Winker Property (T.12N., R.21E., NE 1/4, NE 1/4, Sec. 9) was similar to, and within 2 km of, Huiras. The soil at Winker was a Pella silt loam similar to Huiras Lake, but it lacked the surface layer of peaty-muck (Parker et al. 1970). Reed canarygrass dominated the vegetation at Winker, but there were patches of native graminoids and herbs plus scattered individuals of green ash (*Fraxinus pennsylvanica* Marshall) throughout the site. Winker was bordered on two sides by agricultural fields and on two sides by early successional trees and shrubs.

Fellenz Woods (Washington County, Wisconsin, T.11N., R.20E., NE 1/4, SW 1/4, Sec. 16) was located in the floodplain of the Milwaukee River. The soil at Fellenz was Aztalan loam, a black soil with slow drainage and high natural fertility (Schmude 1971). Fellenz was bordered along one side by a floodplain forest dominated by slippery elm (*Ulmus rubra* Muhl.), black ash (*Fraxinus nigra* Marshall), and quaking aspen (*Populus tremuloides* Michx.). The herbaceous community, although dominated by reed canarygrass, maintained substantial native species diversity; scattered black ash and red-osier dogwood (*Cornus sericea* L.) had also begun to invade.

On May 3, June 7, and Aug. 3, 2004, we recorded the depth from the soil surface to the water table in four water-monitoring pits per site at Winker and Fellenz and six at Huiras. In June, Fellenz was significantly wetter than Huiras and Winker due to seasonal flooding of the Milwaukee River. June water levels at Fellenz averaged 11 cm above the soil surface, while Winker and Huiras had water levels 12 to 16 cm below the soil surface. The high water period at Fellenz corresponded to the time during and shortly after planting, and many trees and shrubs were planted directly into standing water. By August 2004, average water levels did not differ significantly among the three sites and were 45 to 75 cm below the soil surface (Hovick 2005).

We compared biomass and herbaceous community composition in quadrats located within untreated, control plots at the three sites (Hovick 2005). Shannon-Wiener diversity was significantly higher at Fellenz (0.62) than at either Huiras or Winker (< 0.25) ($p < 0.05$); species evenness followed the same trend ($p < 0.05$). Reed canarygrass biomass was significantly lower at Fellenz (279 g m^{-2}) than at either Huiras (846 g m^{-2}) or Winker (1027 g m^{-2}) ($p < 0.05$). Reed canarygrass cover was also significantly lower at Fellenz, although it was greater than 70% at all three sites. Because of the exceptionally dense cover of *P. arundinacea* at Winker there was significantly less open ground there than at either Fellenz or Huiras (Hovick 2005).

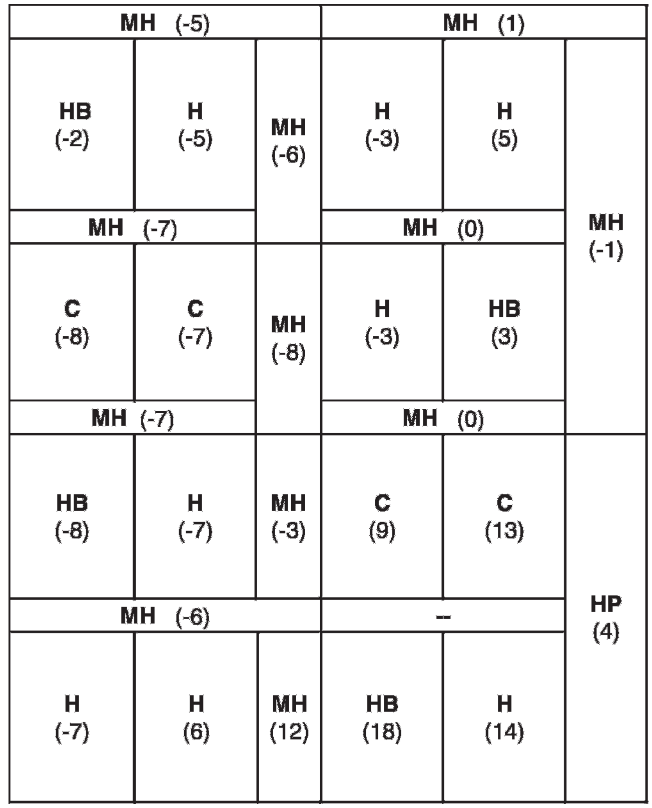

Figure 1. Pre-planting treatment layout and mean plot ground surface elevations at the Huiras Lake site. Treatments are labeled as follows: H = fall herbicide only; HB = fall herbicide + spring burn; HP = fall herbicide + spring plowing; MH = late summer mowing + fall herbicide; C = control. Plot elevations (in parentheses) are deviations from the mean elevation of all plots in centimeters.

Site Preparation and Planting

On Aug. 28, 2002, we divided the Huiras study site into eight large areas of unmowed reed canarygrass by mowing swaths between the unmowed areas (Figure 1). Two of these eight unmowed areas were randomly designated as controls and received no treatment or control of the reed canarygrass. The remainder of the study area, including both the mowed and the unmowed treatment areas, was herbicided on Nov. 9, 2002. This date was the first warm day (afternoon temperature 18°C) following a hard freeze; we previously had success using this timing of glyphosate application on *P. arundinacea*. Reed canarygrass is still actively growing on warm days following the first freeze and seems particularly susceptible to herbicide damage. We applied a 0.7% active ingredient glyphosate solution (2 oz. of 43% a.i. concentrate Roundup per gallon of water) using a boom sprayer. Following label recommendations, we applied 110 gallons of this herbicide solution to the treated area (0.9 ha), for an application rate of approximately 3 quarts of concentrate per acre of land.

We established five pre-planting experimental treatments: 1) controls (C) – no treatment of reed canarygrass; 2) herbicided fall 2002 only (H) – no other treatments applied; 3) mowed late summer 2002, followed by herbicide fall 2002 (MH); 4) herbicided fall 2002 and burned spring 2003 (HB) and; 5) herbicided fall 2002 and plowed with a moldboard plow spring 2003 (HP). All treatments other than the controls received the herbicide treatment described prior. In the herbicide-only treatment (H) the herbicided reed canarygrass was left in place; in the other three treatments, *P. arundinacea* biomass was removed either by mowing before herbicide treatment (MH), or burning or plowing the dead grass (HB and HP).

The eight unmowed areas were each divided in half, creating 16 plots. The five treatments were stratified throughout the study site to distribute the effects of varying environmental factors, and randomized within this stratified arrangement to the extent possible (Figure 1). Reed canarygrass cover was uniformly high and did not vary substantially across the study site. As may be expected in an experimental design that involves management on a large scale, there were practical constraints that prevented a strict random arrangement of the treatment locations. Control plots (C) were assigned to adjacent locations (Figure 1) because it was impractical with our equipment to prevent herbicide damage to smaller units. The mow and herbicide (MH) treatment was assigned to the mowed swaths, which were necessary to separate the other treatments and to facilitate work in the study area (Figure 1). Despite the non-random location of the C and MH treatments, they were arranged to have a range of environmental variation similar to the other treatments, and we were confident that woody plant survival in C and MH treatments could be directly compared to the herbicide-only (H) and herbicide and burn (HB) treatments. The herbicide and plow (HP) treatment, however, was restricted to the driest corner of the study site because that was the only area accessible by a tractor in the spring of 2003 (Figure 1). When we compared survival in the HP treatment with that in other treatments, we recognized that because HP was confounded with elevation, we were actually comparing herbicide and plow (HP) of a slightly higher and drier site to C, H, MH and HB applied to areas that were, on average, slightly lower and wetter.

The primary environmental factor that varied among the Huiras study site plots was a difference in elevation of the ground surface; there was a 26-cm range in mean elevation between the highest and lowest plots (Figure 1). Although this 26-cm range of elevation made no apparent difference in available soil moisture through most of the growing season, it influenced how long the soil was flooded or saturated at the beginning of the growing season, and had a potential to influence woody plant survival. By stratifying treatments across these elevation differences, we nearly eliminated any differences in mean ground surface elevation among treatments (HP = 4.0 cm deviation from the overall site mean elevation; HB = 3.0 cm; C = 2.0 cm; H = 0.3 cm; MH = −2.2 cm). There was no significant difference in mean plot elevation among treatments ($p = 0.491$); however, since the HP treatment was confined to one large plot, that treatment lacked the variation in elevation among plots characteristic of all other treatments (Figure 1).

Plots were not treated as replicates for statistical analysis of survival; the most direct way to analyze survival was with contingency table tests of independence. The purpose of the plot layout in our design was 1) to ensure that treatments were spread evenly across the slight environmental variation within the study site, 2) to ensure that trees were planted with an even density, and 3) to divide the area into manageable units for accurate census of survival.

One of the primary goals of this study was to compare the survival of 23 common woody species planted in reed canarygrass. Since survival of planted seedlings can vary dramatically from site to site and from year to year, we repeated two of our treatments, control (C) and herbicide only (H), in the following growing season at the Winker and Fellenz study sites. The purpose of this replication was to 1) test the planted species in a greater range of the environments in which reed canarygrass can be dominant, and 2) to explore whether the relative survival of these species can be generalized to other sites and growing seasons. It was beyond the scope of this study to replicate all pre-planting treatments of *P. arundinacea* at Winker and Fellenz. In fall 2003, one 400-m^2 plot at each of the Winker and Fellenz sites was sprayed with glyphosate using similar methods to those described previously for Huiras; a similar 400-m^2 plot was left untreated as a control at each site.

At Huiras we planted 22 species of trees and shrubs by hand at a density of 9,500 trees per hectare between April 25 and May 8, 2003 (Table 1). Between April 23 and May 1, 2004, we planted 21 species of trees and shrubs at Winker and Fellenz

(Table 1) at a density of 11,300 trees per hectare. The spacing of individuals at all three sites (0.95 m^2 to 1.13 m^2 per tree or shrub) was sufficient to eliminate potentially competitive interactions among the planted stock in early years of establishment.

Data Collection and Analysis

We recorded tree and shrub survival at Huiras Lake in early September 2003, before deciduous species began losing their leaves. We made similar survival counts in September 2004 at Huiras, Winker, and Fellenz. First-year survival (measured in 2003 at Huiras and 2004 at Winker and Fellenz) was compared among the three sites to analyze the combined effects of site and planting year on the relative survival of the planted species.

We measured the height and stem diameter of trees and shrubs at the time of planting to characterize the initial sizes of the different species. Height and stem diameter were also measured during fall 2004 survival counts to determine change in average size over the study period.

Between Aug. 3 and Aug. 6, 2004, we recorded a visual estimation of cover of each species in 10 1-m^2 quadrats in each treatment. Cover of each species was estimated on a scale of one to five [1 = Present to 5% cover; 2 = 6%–25%; 3 = 26%–50%; 4 = 51%–75%; 5 = 76%–100% cover (Daubenmire 1959)]. These estimates were converted to the midpoints of the cover ranges for calculating evenness and Shannon-Wiener diversity for each quadrat. We counted all tree seedlings that had established naturally in the quadrats from seed (slippery elm, red maple, and green and black ash). These seedlings were easily distinguishable from the one- to three-year-old planted trees.

We harvested living biomass from four 0.25-m^2 quadrats randomly located in each treatment at all three sites on Oct. 1, 2004. Vegetation was clipped at ground level, dried to a stable weight at 60°C, sorted to species, and weighed. We recorded the weight of living above-ground reed canarygrass biomass, the combined weight of all other biomass, and total species richness for each quadrat.

Survival analyses used categorical data and two-way contingency tables. The log-likelihood chi-square (G test) was used to test for significant departures from independence. We used the simultaneous test procedure (Sokal and Rohlf 1995) to identify species or treatment groups that were not significantly different at the $p = 0.05$ level. Recognizing that the herbicide and plow (HP) treatment was not strictly comparable to the other four treatments, we tested for significant differences in survival among treatments both including and excluding the HP treatment.

We compared mean survival of 18 species in the three planting sites using the nonparametric Kruskal-Wallis test statistic. We then used paired T-tests with an adjusted Bonferroni error rate (experiment-wise $p < 0.05$) to compare average survival after one growing season between all pairings of the three study sites (SPSS 2004).

The effect of average initial height of each species on two-year survival of the species was analyzed using linear regression (SYSTAT 2002). Survival percentages varied widely among species (0.0%–77.6%). We therefore transformed the data using an improved arcsine-square root transformation (Johnson and Kotz 1969, cited in Sokal and Rohlf 1995, p. 422), which incorporates total n and the number of survivors instead of survival percentage only, and then tested for departures from normality using the Kolmogorov-Smirnov one-sample test (SYSTAT 2002). This transformation gave all survival data in all treatments a normal distribution and was used for subsequent analyses.

We tested herbaceous cover, biomass, and the initial and final heights of the trees and shrubs for departures from normality using the Kolmogorov-Smirnov one sample test (SYSTAT 2002). Height measurements approached normality after a natural log transformation ($Y' = \ln Y$). However, variances were significantly different among treatments [$p < 0.05$; Levene's test (SPSS 2004)], so Welch's test (SPSS 2004) was used in lieu of the ANOVA F for testing treatment effects (Day and Quinn 1989). We used Dunnett's T3 for unplanned, multiple pairwise comparisons following a significant Welch test (Day and Quinn 1989).

Herbaceous cover data could not be transformed suitably to approach normality; therefore, we used the non-parametric Kruskal-Wallis test and Mann-Whitney statistic to compare cover among treatments. Unplanned multiple pairwise comparisons were made using the Fligner-Policello method (Day and Quinn 1989), a modification of the Mann-Whitney U test that is robust to variance heterogeneity, and were considered significant at $\alpha < 0.10$ because of small sample sizes and the low power of the tests.

RESULTS

Tree and Shrub Survival

Effects of Treatments. Mean survival of 22 species after two growing seasons at Huiras was significantly higher in herbicide and plow (HP) plots (50%)

Table 2. Percent survival by treatment for individual species at Huiras Lake after two growing seasons. Species are ordered by percent survival across all experimental treatments combined (excluding controls). Treatment labels are the same as those used in Figure 1. Percentages followed by the same subscript letters do not differ significantly among treatments ($p > 0.05$; df = 4). Percentages in bold indicate that the species reached its highest survival in that treatment. Statistics are based on log-likelihood chi-square tests (ns = not significant; * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$). The log-likelihood tests were calculated with and without the herbicide and plow (HP) treatment in the experimental design, since that treatment is not exactly comparable to the other four (see text). Vertical lines connect groups of species that do not differ significantly, according to the simultaneous test procedure for stepwise groupings (Sokal and Rohlf 1995).

		Survival (%)						P-value		
Species	N	All Treat.	HP	HB	H	MH	C	with HP	without HP	Groupings
Black currant	49	85.7	$\mathbf{100}_{a}$	88.9_{a}	83.3_{ab}	83.3_{ab}	28.6_{b}	*	*	
Quaking aspen	70	78.3	80.0_{a}	75.0_{a}	$\mathbf{83.3}_{a}$	73.7_{a}	0.0_{b}	***	***	
Green ash	658	75.8	$\mathbf{85.1}_{a}$	80.7_{a}	84.6_{a}	56.3_{b}	16.2_{c}	***	***	
Red-osier dogwood	621	67.9	$\mathbf{86.4}_{a}$	71.7_{ab}	69.4_{ab}	57.6_{b}	12.2_{c}	***	***	
Elderberry	70	67.8	$\mathbf{100}_{a}$	75.0_{a}	61.5_{a}	62.5_{a}	0.0_{b}	***	***	
Highbush cranberry	73	62.3	$\mathbf{100}_{a}$	76.9_{a}	38.5_{b}	75.0_{a}	8.3_{b}	***	***	
Black ash	216	59.6	$\mathbf{80.0}_{a}$	48.7_{b}	51.9_{ab}	76.0_{a}	2.6_{c}	***	***	
Nannyberry	129	58.3	$\mathbf{87.5}_{a}$	60.9_{a}	53.2_{a}	56.7_{a}	14.3_{b}	**	**	
Silky dogwood	108	57.1	62.5	**78.9**	56.4	40.0	52.9	ns	ns	
Tamarack	576	48.6	$\mathbf{70.7}_{a}$	50.0_{ab}	40.0_{b}	53.6_{ab}	7.6_{c}	***	***	
Black raspberry	49	26.2	0.0_{b}	$\mathbf{55.6}_{a}$	16.7_{ab}	25.0_{ab}	0.0_{b}	*	*	
American elm	49	26.2	25.0	**33.3**	22.2	27.3	0.0	ns	ns	
White cedar	576	23.3	$\mathbf{63.4}_{a}$	28.0_{b}	16.6_{b}	18.1_{b}	2.2_{c}	***	***	
Basswood	52	15.6	**50.0**	11.1	16.7	7.1	0.0	ns	ns	
Slender willow	245	15.0	15.8_{ab}	11.6_{ab}	11.5_{ab}	$\mathbf{22.4}_{a}$	0.0_{b}	**	**	
Silver maple	108	14.4	25.0_{ab}	5.3_{ab}	7.9_{ab}	$\mathbf{28.0}_{a}$	0.0_{b}	*	*	
Red maple	488	13.7	$\mathbf{29.7}_{a}$	16.3_{ab}	12.8_{ab}	7.8_{b}	5.1_{b}	**	ns	
Bebb's willow	26	7.7	33.3	0.0	0.0	33.3	—	ns	ns	
Paper birch	125	7.6	0.0	4.5	**11.6**	6.7	0.0	ns	ns	
Pussy willow	238	7.2	$\mathbf{17.6}_{a}$	0.0_{b}	2.2_{b}	17.2_{a}	0.0_{b}	***	***	
Gray dogwood	49	4.9	0.0	**11.1**	5.6	0.0	0.0	ns	ns	
Yellow birch	93	0.0	0.0	0.0	0.0	0.0	0.0	ns	ns	
Mean survival	4668	37.4	$\mathbf{50.5}_{a}$	40.2_{b}	33.9_{b}	37.6_{b}	7.1_{c}	***	***	

than in all other experimental treatments (34%–40%; Table 2). Because the herbicide and plow (HP) treatment was applied to plots having a narrower range of elevation and was, therefore, not strictly comparable to the other treatments, we analyzed survival both with and without the HP treatment in the experimental design (Table 2). The overall significance of the log-likelihood tests were nearly unaffected by eliminating HP from the analysis; red maple was the only species that showed significant treatment effects only when HP was included. The significance relationships of the other four treatments (HB, H, MH, and C) were unaffected by whether or not the herbicide and plow treatment was included (Table 2).

Average survival in control plots was significantly lower than in any other treatments (Table 2). Eleven of the 22 species had their highest survival (and significantly higher than at least one other treatment) in herbicide and plow (HP) plots (Table 2). If the herbicide and plow treatment is eliminated from the analysis, eleven of the 22 species had their highest survival in the herbicide and burn (HB) treatment; however, only highbush cranberry (*Viburnum opulus* var. *americanum* Aiton) had significantly higher survival in the herbicide and burn than in the herbicide-only (H) treatment. The combined survival of the five species that were planted in greatest numbers (n > 475 for each species: green ash, red-osier dogwood, tamarack, white cedar, and red maple) was also significantly higher in HP (68.6%) than in any other treatment. All five species had greatest survival in the HP treatment, and white cedar had good survival only in that treatment (Table 2). Average survival of these five species was significantly higher in the HB treatment (51.4%)

than in the MH treatment (40.6%); survival in the herbicide-only (H) treatment (47.2%) was intermediate between these two (Hovick 2005).

After two growing seasons, there was wide variation among species in average survival in the four experimental treatments combined, from 85.7% in black currant (*Ribes americanum* Miller) to 0.0% in yellow birch (*Betula alleghaniensis* Britton) (Table 2). Of the five most planted species, green ash and red-osier dogwood had very high survival across all four of the treatments, tamarack had 49% survival, and white cedar and red maple had poor overall survival.

Effects of Elevation. While there was no difference in the mean ground surface elevation among treatments, plot elevation did vary within treatments (Figure 1), so we could determine whether elevation had a significant effect on survival in our study site. Since HP plots were only located at relatively high elevations, they were eliminated from this analysis of the effect of elevation on survival. With the HP treatment excluded, survival of all species pooled was marginally higher in plots with lower-than-average elevation (35.8%) versus high elevation plots (33.2%; G = 3.582, 1 df, p = 0.058). The growing season following the 2003 plantings was dry; 14.6 cm of rainfall was recorded at the University of Wisconsin – Milwaukee Field Station (14.5 km SSE of the Huiras Lake) for the months of June, July, and August, compared to a long-term average rainfall of 28.4 cm for those three months.

The effect of elevation on survival was small; survival for only six of the 22 individual species differed significantly by elevation category. White cedar had significantly higher average survival at high elevation (27.1%) than in low elevation plots (12.5%). Green ash, red-osier dogwood, paper birch, and slender willow collectively averaged 10.6% higher survival in relatively low elevation plots. Only black ash showed a large difference in survival between lower (61.9%) and higher (31.2%) than average elevation plots.

Effects of Size of the Planted Stock. We calculated the average height of each species planted in Huiras Lake plots, but did not follow the heights of individual stems. The average height of planted stock varied widely, from 11 cm for basswood to 98 cm for silky dogwood (*Cornus amomum* Miller); however, in a comparison among species, the rate of survival was not significantly correlated with the average height of the planted stock (p = 0.082). Two-year survival at Huiras was also not significantly correlated with the relative amount of growth (as a percent of mean initial height for that species) over two growing seasons (p = 0.140). Mean height after two growing seasons at Huiras Lake did not differ among treatments for most species. Green ash, black ash, and tamarack grew significantly less in control plots, but their growth rates followed no consistent pattern among the other four treatments.

Effects of Planting Site and Year of Planting. In order to test the planted species over a greater range of environmental conditions, and to determine whether the relative survival of species could be generalized over sites and growing seasons, we planted 17 of the same species planted at Huiras in 2003 at two additional sites in the following growing season, 2004 (Tables 1 and 3). We compared survival of trees and shrubs after one growing season at the Fellenz-04 and Winker-04 study sites with first-year survival at Huiras-03 in the control (C) and herbicide-only (H) treatments to assess the effects of planting site and year on survival (Table 3). After one growing season, average survival of all species in the herbicide-only plots was significantly higher at Winker-04 than at Huiras-03, with survival at Fellenz-04 intermediate to the other two sites. Average survival in controls at Winker-04 and Fellenz-04 did not differ, but survival at each was nearly four times as high as survival in controls at Huiras-03 (Table 3). Rainfall from June through August 2004 (23.4 cm) nearly equaled the long-term average (28.4 cm), compared to the low rainfall (14.6 cm) for those months in 2003.

Within the herbicide-only and control treatments, survival for nearly every species differed among sites (Table 3). In both herbicide-only and control plots, most species had lower survival at Huiras-03 than at the Winker-04 and Fellenz-04 sites. The difference in survival between the herbicide-only and control treatments was much more marked at Huiras-03 than at the 2004 sites, and for Fellenz-04 the average survival of all species did not differ between control and herbicide-only treatments (Table 3). Where survival differed between treatments, mortality was always higher in controls.

Mean first-year survival was greater than 75% for eight species in herbicide-only plots at the three sites (Table 3). In general, all species that had greater than 50% average survival in the herbicided plots tended to survive reasonably well in all three sites. Those species with less than 50% average first-year survival in herbicided plots generally had very low survival at either Huiras-03 or Fellenz-04 and relatively better survival at Winker-04 (Table 3).

Table 3. Comparison of first-year survival among sites (H = Huiras-03; W = Winker-04; F = Fellenz-04). The G-test was used to test for differences in individual-species survival among sites and between treatments. Average survival percentages of all species in each treatment were compared among sites with paired T-tests, using the Bonferroni experimentwise error adjustment (see Methods). Survival percentages for two sites that are followed by the same lower case letter did not differ significantly. The column on the far right notes species that had significantly higher survival in herbicide-only versus control plots at each site. (ns = not significant; * = $p < 0.05$; ** = $p < 0.01$; *** = $p < 0.001$).

	Herbicide-only (Percent Survival)					Control (Percent Survival)					Control vs. Herbicide-only		
	H	W	F	Mean	P	H	W	F	Mean	P	H	W	F
Silky dogwood	92.3	100.0	100.0	97.4	ns	64.7	80.0	80.0	74.9	ns	*	*	*
Green ash	88.0	96.7	93.3	92.7	ns	42.9_b	100.0_a	90.0_a	77.6	***	***	ns	ns
Red-osier dogwood	77.9_b	100.0_a	100.0_a	92.6	***	17.3_b	73.3_a	93.3_a	61.3	***	***	***	ns
Elderberry	77.0	92.9	100.0	90.0	*	0.0_b	21.4_b	100.0_a	40.5	***	***	***	ns
Winterberry	NA	80.0	86.7	83.4	ns	NA	73.3_b	100.0_a	86.7	***	NA	ns	ns
Black ash	57.0_b	100.0_a	90.0_a	82.3	***	10.5_b	100.0_a	100.0_a	70.2	***	***	ns	ns
Silver maple	60.5	85.0	85.0	76.8	*	16.7_c	90.0_a	55.0_b	53.9	***	**	ns	*
Highbush cranberry	53.8	87.5	87.5	76.3	*	8.3_b	81.3_a	56.3_a	48.6	***	**	ns	*
Red maple	49.4_b	76.7_a	66.7_{ab}	64.3	**	14.1_b	80.0_a	70.0_a	54.7	***	***	ns	ns
Paper birch	69.8_a	65.0_{ab}	30.0_b	54.9	*	0.0_b	35.0_a	15.0_{ab}	16.7	**	***	ns	ns
Basswood	50.0_b	90.0_a	5.0_b	48.3	***	0.0_b	40.0_a	0.0_b	13.3	***	**	**	ns
Gray dogwood	44.4_{ab}	80.0_a	20.0_b	48.1	***	0.0_b	65.0_a	10.0_b	25.0	***	**	ns	ns
White cedar	48.3_a	36.7_a	0.0_b	28.3	***	13.0_b	60.0_a	0.0_c	24.3	***	***	ns	ns
Pussy willow	4.5_b	33.3_a	44.4_a	27.4	***	0.0_b	11.1_{ab}	16.7_a	9.3	*	ns	ns	ns
Bebb's willow	0.0_b	60.0_a	20.0_{ab}	26.7	**	NA	0.0	0.0	0.0	ns	NA	**	ns
Yellow birch	3.0_b	50.0_a	26.7_a	26.7	***	0.0_b	46.3_a	43.3_a	29.9	***	ns	ns	ns
Slender willow	12.6_b	16.7_{ab}	50.0_a	26.4	**	0.0	0.0	11.1	3.7	ns	**	*	**
Tamarack	44.4_a	30.0_a	3.3_b	25.9	***	10.9	3.3	3.3	5.8	ns	***	**	ns
Species average	49.0_b	71.1_a	56.0_{ab}	58.7	*	12.4_b	53.3_a	46.9_a	37.5	***	***	***	ns

Diameter was not related to survival of willow stakes; living and dead willow stakes did not differ in diameter at either Winker-04 (T-test; $p = 0.579$) or Fellenz-04 ($p = 0.206$). In general, willow stake survival was low and was not affected by source (collected versus purchased), except that collected *Salix petiolaris* stakes had significantly higher survival (55.6%) than purchased stakes (16.7%) at Winker-04 ($p = 0.013$).

Treatment Effects on the Plant Community

Combined biomass of all species other than reed canarygrass after two growing seasons at Huiras Lake decreased as reed canarygrass biomass increased (Non-RC(g) = -0.958 RC(g) + 5.291; $p < 0.001$, $R^2 = 0.691$). At all three sites considered together, species richness was negatively related to reed canarygrass biomass (Figure 2a) and positively related to non-reed canarygrass biomass (Figure 2b). Total biomass was not correlated with species richness ($p = 0.968$).

After two growing seasons at Huiras, reed canarygrass biomass was highest in control plots (846 g m^{-2}) and lowest in the herbicide and plow (HP) treatment (8 g m^{-2}; $p = 0.002$). Both the HP and HB treatments had significantly less reed canarygrass biomass than did controls. Total biomass and percent open space did not differ among treatments (Hovick 2005). Herbicide-only plots at Winker and Fellenz had significantly lower reed canarygrass biomass, total biomass, and reed canarygrass cover, and a significantly higher percentage of open ground after one growing season than did control plots (Hovick 2005).

Evenness, richness, Shannon-Wiener diversity, native species richness, and the number of volunteer tree seedlings were significantly lower in control plots than all other treatments (Table 4). The treatments that were plowed or burned after herbiciding (HP and HB) had significantly higher evenness and diversity than herbicide-only (H) plots, and the HP treatment alone had significantly higher richness than herbicide-only plots (Table 4). After one growing season diversity, richness, and evenness were all significantly lower in control plots than in herbicide-only plots at Winker but not at Fellenz (Table 4).

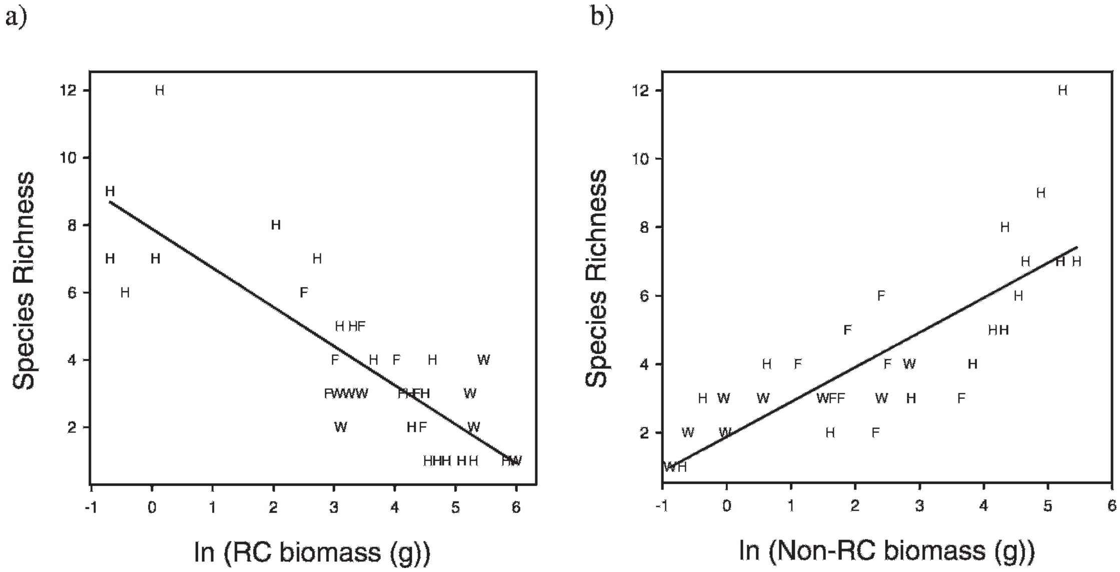

Figure 2. Response of species richness to aboveground living biomass in 0.25-m^2 quadrats at Huiras Lake (H) after two growing seasons and at Winker (W) and Fellenz (F) after one growing season: a) Reed canarygrass mass only [Richness = −1.159(RC biomass) + 7.880; $p < 0.001$; $R^2 = 0.681$]; b) Combined mass of all other species [Richness = 1.018 (Non-RC biomass) + 1.869; p = < 0.001, $R^2 = 0.696$]. The single "H" in the lower-left corner of b) represents 6 samples from Huiras with non-RC biomass equal to zero. When both tests were repeated excluding the outlier at the top of each graph, the relationship did not change substantially for either RC ($R^2 = 0.675$) or non-RC biomass ($R^2 = 0.726$).

Table 4. Among-treatment variation in herbaceous community statistics after two growing seasons at Huiras and one growing season at Winker and Fellenz (n=10 for each treatment). For Huiras, all means (SE) were compared with Welch's test and multiple pairwise comparisons were made using Dunnett's T3 procedure ($p > 0.05$). Treatments followed by the same lower case letter do not differ significantly. Comparisons of data from Winker and Fellenz were made with separate variance T-tests. Treatment labels are the same as those used in Figure 1.

	Diversity	Richness	Evenness	Native Richness	Volunteer Tree Seedlings (No. / m^2)
Huiras treatments:					
HP	1.8_a (0.06)	11.3_a (0.6)	1.7_a (0.04)	5.5_a (0.6)	56.2_a (15.8)
HB	1.7_a (0.1)	9.3_{ab} (0.7)	1.7_a (0.07)	6.3_a (0.7)	20.6_a (5.8)
MH	1.3_{ab} (0.2)	8.4_{ab} (1.0)	1.4_{ab} (0.2)	5.7_a (0.7)	45.9_a (15.7)
H	0.8_b (0.2)	6.6_b (0.9)	1.0_b (0.1)	4.0_a (0.6)	25.1_a (5.7)
C	0.1_c (0.05)	1.9_c (0.4)	0.2_c (0.09)	0.9_b (0.4)	3.5_b (1.9)
df_1	4	4	4	4	4
df_2	20.7	21.7	20.2	22.1	22.0
W	76.0	28.5	27.9	11.1	6.8
P	< 0.001	< 0.001	< 0.001	< 0.001	0.001
Winker treatments:					
H	0.7_a (0.1)	3.2_a (0.2)	1.3_a (0.2)	1.7 (0.2)	12.5_a (4.8)
C	0.2_b (0.03)	2.3_b (0.2)	0.4_b (0.5)	1.3 (0.2)	1.9_b (0.8)
df	11.7	16.2	14.2	16.5	17.8
T	−5.515	−2.941	−5.908	−1.338	−3.873
P	< 0.001	0.010	< 0.001	0.199	0.001
Fellenz treatments:					
H	1.0 (0.2)	4.7 (0.9)	1.4 (0.3)	2.6 (0.7)	2.5_a (1.0)
C	0.6 (0.1)	3.6 (0.7)	1.2 (0.2)	2.2 (0.5)	0.1_b (0.1)
df	14.3	16.3	14.0	15.3	10.1
T	−1.128	−0.837	−0.464	−0.050	−2.863
P	0.278	0.415	0.650	0.961	0.017

Table 5. Number of 1 m^2 quadrats (out of 10 sampled in each treatment) in which a species occurred and average cover in those quadrats in which the species was present for the most common species at the Huiras Lake study site. Treatment designations are the same as in Figure 1.

		HP		**HB**		**H**		**MH**		**C**	
Treatments: Species	**Nat. / Intro**	N	Mean Cover	N	Mean Cover	N	Mean Cover	N	Mean Cover	N	Mean Cover
Common ragweed (*Ambrosia artemisiifolia* L.)	N	1	15.0	1	37.5	0		2	32.5	0	
Canada thistle (*Cirsium arvense*)	I	8	10.3	5	9.5	3	2.5	6	8.3	0	
Horseweed (*Conyza canadensis* L.)	N	5	5.0	6	10.4	3	10.8	4	2.5	1	2.5
Queen Anne's lace (*Daucus carota* L.)	I	2	8.8	4	5.6	2	2.5	2	51.3	0	
Spike rush (*Eleocharis palustris* L.)	N	0		0		1	2.5	2	37.5	0	
Eastern willow herb (*Epilobium coloratum* Biehler)	N	1	2.5	7	9.6	4	5.6	5	5.0	1	2.5
Boneset (*Eupatorium perfoliatum* L.)	N	5	5.0	4	8.8	0		5	12.5	0	
Prickly lettuce (*Lactuca serriola* L.)	I	7	18.2	5	12.5	4	5.6	4	2.5	0	
Reed canarygrass (*Phalaris arundinacea* L.)	I	4	17.5	8	34.1	10	73.0	8	31.3	10	87.5
Stinging nettle (*Urtica dioica* L.)	N	0		0		1	62.5	1	37.5	0	
Common vervain (*Verbena hastata* L.)	N	10	20.3	5	16.5	4	2.5	7	2.5	0	

The treatment of the reed canarygrass affected the composition of the herbaceous community that invaded the treated plots at the Huiras Lake study site (Table 5). Reed canarygrass was present in all 10 quadrats sampled in the herbicide-only treatment, and had reestablished an average cover of 73% two years after treatment, while it was present in only four of 10 sample quadrats in the herbicide and plow treatment. Canada thistle (*Cirsium arvense* [L.] Scop.), prickly lettuce (*Lactuca serriola* L.), and common vervain (*Verbena hastata* L.) were all most common in the herbicide and plow treatment. Burning following the herbicide treatment (HB) appeared to favor the establishment of horseweed (*Conyza canadensis* [L.] Cronq.) and eastern willow herb (*Epilobium coloratum* Biehler) (Table 5).

At Huiras, Shannon-Wiener diversity declined dramatically at a threshold of reed canarygrass cover between 63% and 88% (Figure 3). If samples were grouped into those having high (> 80%) and low (< 70%) reed canarygrass cover, samples with greater than 80% reed canarygrass cover had significantly lower species richness, evenness, Shannon-Wiener diversity, and number of volunteer tree seedlings than those with less than 70% cover (Hovick 2005).

DISCUSSION

There is a great need for development of methods to restore northern, reed canarygrass–dominated wetlands to swamp forest (National Research Council 2001). There has been some experimentation to determine whether herbaceous species can be used to competitively control reed canarygrass for restoration of sedge meadow (Perry and Galatowitsch 2003, Perry et al. 2004, Foster and Wetzel 2005), but there have been few attempts to establish woody species in reed canarygrass wetlands. Foster and Wetzel (2005) tried planting one willow (*Salix sericea* Marshall) and common elderberry (*Sambucus canadensis* L.) in reed canarygrass monocultures that were either burned or herbicided, but had very poor survival of the woody species.

We designed and tested simple and low-cost methods for preparing and planting sites dominated by reed canarygrass to achieve high survival of planted woody species. We also tested a wide variety of woody species that are common components of swamps and shrub carrs to determine which species are the best candidates for establishment of an early successional woody community in reed canarygrass. Of four pre-planting treatments tested, fall herbicide application followed by spring plowing provided the highest survival of woody plants overall, and the best survival for the greatest number of individual species. Plowing following herbicide application has been shown to effectively control reed canarygrass in wet meadow restorations because herbicide-damaged rhizomes desiccate when they become exposed to the air (Kilbride and Paveglio 1999). Although plowing following herbicide resulted in the highest woody plant survival, all of the herbicided treatments, including the herbicide-only treatment, provided reasonably high survival compared to controls.

Previous studies suggest that herbicide is essential for reed canarygrass control on all but very small-scale projects (Apfelbaum and Sams 1987, Kilbride and Paveglio 1999, Tu 2004). Recent management recommendations suggest using multiple treatment methods in combination to control reed canarygrass

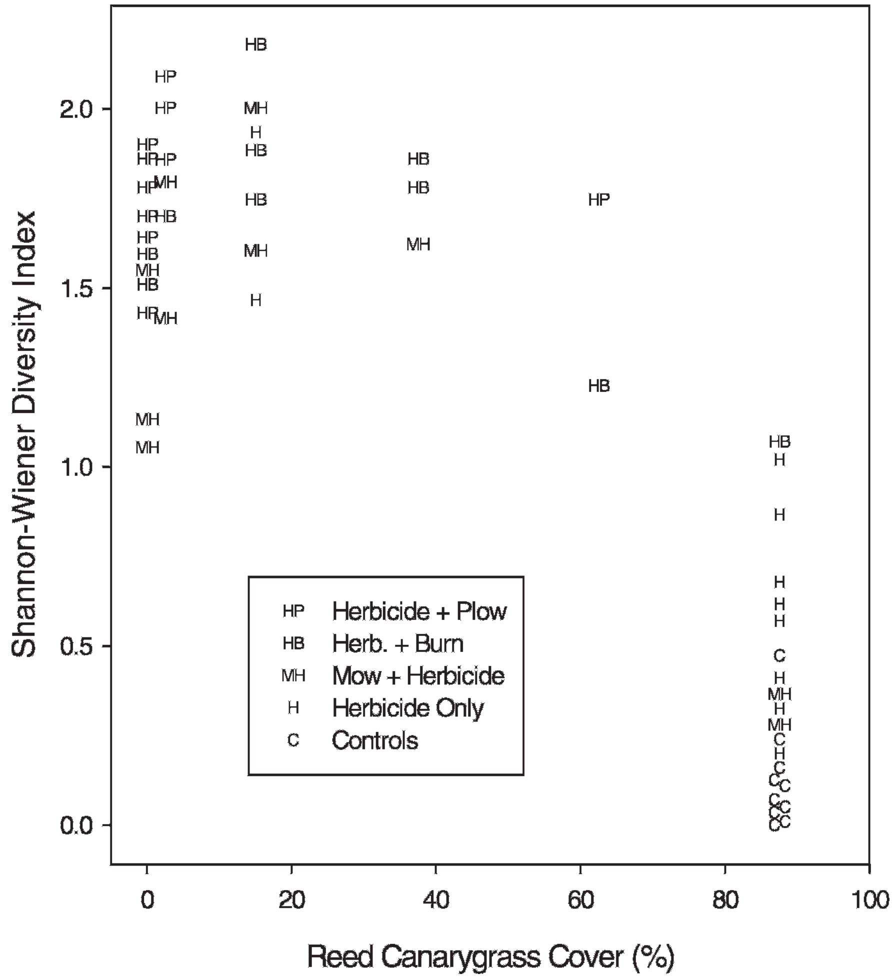

Figure 3. Shannon-Wiener diversity values from quadrats with differing reed canarygrass cover estimates. Symbols have been offset slightly to minimize overlapping.

because a single herbicide application is often insufficient for long-term suppression (Henderson 1991, Kilbride and Paveglio 1999, Tu 2004). Prescribed burning has been recommended for reed canarygrass control when seeds of a native fire-adapted community are present in the seed bank (Hutchison 1992), but improper timing of burns may actually stimulate reed canarygrass growth (Howe 1995). Fire is clearly not an available management tool when the restoration goal is swamp forest. Although mowing has been found to decrease reed canarygrass cover when it was cut five times in a single growing season (Wilkins and Hughes 1932), this method has not been used effectively in large-scale restorations (Tu 2004).

All of the pre-planting treatments tested in this study will be practical for use in some wetland restoration projects; however, spring plowing will not be possible in many wetlands or years because of soft wet soil. A burn in the spring following a fall herbicide treatment produced the second highest survival of planted woody stock; however, fire significantly reduced the natural establishment of tree seedlings from local seed sources, which was an important component of total woody plant density in all three of our study sites. A single fall herbicide application alone did not provide optimal woody plant survival; however, this simple treatment requires relatively little labor or equipment, and yielded good survival for many of the species planted in this study. The average two-year survival of the top 10 species in our study was 62% in the herbicide-only treatment, compared with 85% in the herbicided and plowed treatment, and only 14% in the untreated reed canarygrass controls. With any of our treatments, we predict that a follow-up application of grass-selective herbicide (e.g., sethoxydim or fluazifop) in subsequent years might enhance survival by delaying the reestablishment of reed canarygrass until the trees and shrubs are well established. Use of grass-selective herbicide in subsequent years may be particularly beneficial following the herbicide-only treatment since reed canarygrass reestablished cover most quickly following that treatment.

We also found an apparent correlation between woody plant survival and diversity of the community that established after treatment of the reed canarygrass. The plowed treatment had the highest tree survival, the greatest herbaceous species diversity, evenness and richness, and the lowest reed canarygrass cover and biomass. The rank order of herbaceous species diversity among our experimental treatments directly mirrored the relative order of average survival of the planted trees and shrubs. A diverse native community may enhance survival of a variety of woody plants because of the protection forbs provide young trees and shrubs (Clewel and Lea 1989) or because of the variety of safe sites the community has available for establishment (sensu Urbanska 1997). Although previous research has both supported (Berkowitz et al. 1995, McLeod et al. 2001) and refuted (Hill et al. 1995, Mazia et al. 2001) the positive effect existing vegetation can have on the establishment and growth of young woody vegetation, the composition of the plant community has consistently been found to be important (Kruse and Groninger 2003). Differences between the Winker and Fellenz study sites in the relative response of woody plant survival to different treatments also support the idea that herbaceous species diversity is positively related to survival. Untreated reed canarygrass control plots at the Fellenz site had lower reed canarygrass biomass and higher species diversity than control plots at Winker or Huiras, and there was less difference in woody plant survival between the control and herbicided plots at Fellenz than at either Winker or Huiras.

While the focus of our study was the survival of hand-planted one- to three-year-old seedlings, an unexpected result was the number of tree seedlings that established from natural seed rain after reed canarygrass cover had been dramatically reduced by herbicide. The Huiras Lake study site was surrounded on three sides by established swamp forest. Although spatial variation in seedling density was high and related to proximity to fruiting trees, we found an average of 37 naturally established tree seedlings per m^2 in herbicide treated plots, while volunteer tree seedling establishment was nearly absent in control plots. Plowed and mowed plots, those treatments that removed the killed reed canarygrass mulch without the use of fire, had average naturally established tree seedling densities of 51 trees per m^2. Removal of the reed canarygrass mulch appeared to offer the advantage of substantial natural tree regeneration if natural sources of seed were present. Our study suggests that direct seeding should be explored as a low-cost means for restoring swamp forest or augmenting a seedling planting, in agreement with recent recommendations for restoring bottomland hardwoods (Groninger 2005). It may also be advantageous to time reed canarygrass control to coincide with mast seed production of adjacent trees (e.g., Kruse and Groninger 2003).

Planting site conditions and weather during the season of planting can dramatically affect the survival of planted bare-root seedlings. Rainfall during summer 2003, the year that the Huiras site was planted, was only about half of the long-term average; this drought was almost certainly the reason that overall first-year survival of plants at Huiras was lower than at the similar Winker site planted in 2004. However, despite that drought in 2003, we demonstrated considerable success at planting several species of trees and shrubs into dense stands of reed canarygrass across a range of biotic and abiotic conditions. Moreover, we found a set of species that had consistently high survival over sites, years, and a range of reed canarygrass control treatments that can be recommended as an early successional woody community to replace reed canarygrass.

A set of "top 10" species emerged from this study based on their consistently high survival. Green ash, black ash, red-osier dogwood, silky dogwood, highbush cranberry, and common elderberry all had excellent two-year survival in all pre-planting treatments at Huiras, and first-year survival in all three site-by-year treatment combinations. Quaking aspen, black currant, and nannyberry (*Viburnum lentago* L.) had excellent survival over two growing seasons in all reed canarygrass control treatments at Huiras, although they were not tested at the other two sites. While tamarack did not survive well when planted in 2004 at the Fellenz site, it had very high two-year survival in all four of the herbicide treatments at Huiras. The historic loss of tamarack from wetlands in southern Wisconsin, and the fact that it is known to establish best in early successional habitats, encourages us to include tamarack in our list of recommended species.

Green ash, black ash, and quaking aspen are early succession wetland species that can grow quickly to overtop reed canarygrass and also produce large seed crops. Green ash was found to constitute a significant proportion of a naturally regenerating swamp forest in southern Illinois (Kruse and Groninger 2003) and is planted widely in bottomland hardwood restoration throughout the southeastern United States due to its broad environmental tolerance (Groninger et al. 2000). A closed green ash canopy provides sufficient shade to prevent reed canarygrass dominance of the herbaceous understory (Thompson 1995). Very high first-year survival of silver maple (*Acer saccharinum* L.) (77%), red maple (64%), and paper birch (55%) across all three sites initially indicated that these trees would also be valuable for the first stage of swamp forest restoration. However, all three species had very poor survival between their first and second growing seasons in all pre-planting treatments at Huiras, resulting in a two-year survival of less than 15% for all three species. The cause of the high mortality of these species between the first and second growing seasons was not clear; damage by herbivores or covering by competitors did not appear to explain the loss.

Successful establishment of a number of shrub species was encouraging, since shrubs are often found invading open wetlands in early stages of succession to swamp forest (e.g., Conner 1995, Battaglia et al. 2002). Rapid establishment of a dense shrub canopy may effectively suppress reed canarygrass recolonization. Silky dogwood in particular had uniformly high survival across all treatments, sites, and years. Willows form a major portion of the shrub communities that invade wet meadows in southern Wisconsin (White 1965); however, in our study unrooted stakes of three willow species (Bebb's, pussy, and slender) had disappointingly poor survival across sites and pre-planting treatments despite two planting attempts employing different stock and storage techniques. Soil moisture in these reed canarygrass–dominated habitats may be lower than is optimal for establishment of unrooted willow stakes. Two-year survival of slender willow at Huiras was significantly higher in lower-than-average elevation than in higher plots (Hovick 2005). In a smaller experiment we found that rooted individuals of Bebb's willow, pussy willow, and slender willow were considerably more successful than unrooted stakes (Hovick 2005). The cost and labor of planting unrooted stakes is low relative to planting rooted stock, so some projects may have the capacity to plant a large number of willow stakes to compensate for low expected survival.

One of the initial goals of this project was to develop methods to restore white cedar and tamarack, conifers that were once common in southeastern Wisconsin but now are found only in a few of the least-disturbed wetlands (Hapner and Reinartz 2005). Neither of the conifers was in the group of nine species that had consistently high survival across all pre-planting treatments and sites. White cedar had consistently poor survival in all but the herbicided and plowed treatment. White cedar is a later successional species and apparently could not tolerate the conditions in our study sites, an especially disappointing finding given the poor natural regeneration of the species throughout the upper Great Lakes region (Rooney et al. 2002). White-tailed deer (*Odocoileus virginianus*) browsing often limits the growth and survival of white cedar (Rooney et al. 2002) but was not a factor during the course of this study (S. Hovick, personal observation). While tamarack did not survive well at Fellenz, it had excellent survival at Huiras, and is typically present in early stages of swamp forest succession, becoming established after major wind damage or fire. Our results suggest that tamarack holds more promise than does white cedar for the early phase of conversion of reed canarygrass to swamp forest, and we include it in our list of top 10 species.

Most of the species that were not in the group of top 10 survivors in our study shared three characteristics: 1) two-year survival at Huiras was not consistent among pre-planting treatments, but was dramatically higher in some treatments than in others; 2) first-year survival in 2004 was considerably greater at the drier Winker Property than at the flooded Fellenz Woods; and 3) mortality was very high between the first and second growing season for those species planted at Huiras Lake (survival in herbicide-only plots dropped from the 45%–70% range to the 0%–22% range). These patterns suggest that these less successful species have narrower ranges of environmental tolerance than do the more successful, top 10 species. While we found these less successful species to be poor candidates to form an initial canopy in our reed canarygrass–dominated study sites, some would undoubtedly be successful in a narrower range of conditions.

We will continue monitoring tree and shrub survival, and the composition of the herbaceous community, and will evaluate the long-term effectiveness of these treatments and species for conversion of reed canarygrass to swamp forest. We expect reed canarygrass to continue to reestablish following our original herbicide application, and anticipate using follow-up applications of a grass-selective herbicide. However, based on the present conditions, we expect the planted trees and shrubs to overtop the reed canarygrass and form a canopy that will eventually weaken its competitive effect on the native plant community and lead to the restoration of swamp forest.

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