RESEARCH ARTICLE

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Restoring Native Vegetation to an Urban Wet Meadow Dominated by Reed Canarygrass (*Phalaris arundinacea* L.) in Wisconsin

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ABSTRACT: The perennial grass *Phalaris arundinacea* (reed canarygrass) is a widespread invader of North American wetlands. We asked if the use of herbicide with burning, clipping, or seeding could reduce the cover of *P. arundinacea* and increase cover of native species in a wet prairie that receives stormwater runoff. Two glyphosate treatments decreased *P. arundinacea* cover, and spring seeding of 33 native species doubled species richness and floristic quality compared to no seeding. Despite an initial decrease in abundance, *P. arundinacea* cover was no different than control plots two years after seeding and overall native species richness and cover decreased from the previous year. Application of the grass-specific herbicide sethoxydim in the third year of the study reduced *P. arundinacea* cover and height while allowing native forbs and graminoids to persist. Most of the 75 species identified during the study were perennial, native forbs, and most of the sown species that commonly established in the field had high germination rates in the laboratory. Continued management of *P. arundinacea* is needed to maintain desirable native wetland flora.

Index terms: glyphosate, invasive species, *Phalaris arundinacea*, reed canarygrass, sethoxydim, wetlands

INTRODUCTION

It is the responsibility of human society to sustain ecosystem functioning and preserve biodiversity for ethical, economic, and aesthetic reasons (Ehrlich and Ehrlich 1992; Folke et al. 1996; Myers 1996; Mack et al. 2000). Where biodiversity is already depleted, the challenge is to reverse that loss through restoration. Here, we attempt to reverse losses due to invasive species that form monotypes and displace native species. In such situations, the two interconnected challenges are to control the invader and reestablish the natives.

Wetlands invaded by *Phalaris arundinacea* L. (reed canarygrass, hereafter RCG) provide opportunities to test ways to restore diversity (Lavergne and Molofsky 2004). This perennial cool-season grass species aggressively invades wet meadows, streambanks, and river floodplains in the temperate United States and Canada (Apfelbaum and Sams 1987; Lavergne and Molofsky 2004). Characteristics that contribute to RCG's competitiveness include rapid vegetative spread, efficient growth, tolerance of many hydrologic regimes, and architectural plasticity (Apfelbaum and Sams 1987; Galatowitsch et al. 1999; Lavergne and Molofsky 2004; Herr-Turoff 2005). Although native to North America, invasive populations are considered descendents of European cultivars introduced after 1850 for erosion control and forage (Merigliano and Lesica 1998).

Wetlands in urban landscapes are especially vulnerable to RCG invasion, because runoff from impervious areas carries more water, nutrients, and sediments than would occur where rainfall infiltrates and soils are undisturbed (Watson et al. 1981). These three disturbances facilitate RCG invasion through both individual and synergistic effects (Kercher and Zedler 2004b). Reducing stormwater inflows may not be feasible when sources are off-site and outside managers' control. Such is the situation at the University of Wisconsin-Madison Arboretum, where four wetlands have large areas of monotypic RCG. We asked if RCG could be replaced by native species where stormwater inflows persist. Four tools available to managers to accomplish this goal are herbicide, prescribed burning, weeding, and replanting. The use of each of these tools can be varied in timing and intensity.

We attempted to restore wet meadow vegetation by treating monotypic RCG with herbicide and fire and sowing 33 native species to a site where stormwater inflows could not be curtailed. We then examined the response of RCG and native species in a restoration experiment and sought explanations of the outcome through three associated experiments: (1) germination of the 33 native species in the laboratory, (2) seeding at different times in field plots, and (3) RCG clipping in field plots. For the associated experiments, we predicted that: (1) seed germination rates of at least some species would explain their abundance in the restoration plots; (2) seeding in both fall and spring would increase native species establishment over seeding once, and that fall and spring seeding would favor different species; and (3) manual RCG removal would increase native species cover.

METHODS

Site Description

Lower Greene Prairie (LGP; 43°1'40" N 89°26'15" W) is located within the University of Wisconsin Arboretum in Dane County, Wisconsin. The site was farmed in dry years before it was acquired by the University and restored to prairie from 1942 to 1952 (Allsup 1977). Vegetation surveys and site descriptions from the 1970s indicate that RCG was uncommon and water inundation was restricted to seasonal ponds (Allsup 1977). During the subsequent twenty years, stormwater runoff increased, sediments accumulated on site, and RCG spread and by 2000 formed a monotype measuring 3 ha (Figure 1; J.B. Zedler (ed.), unpubl. data; Werner and Zedler 2002). Analyses conducted in 1999 of the top 5 cm of soil found 39 ppm available phosphorus, 0.45% total nitrogen, and 81% of emergent seedlings from the seedbank were RCG (data of S. Kercher *in* J.B. Zedler (ed.), unpubl. data). RCG is also prevalent upstream of the study site, providing a source of RCG seed into LGP during storm events.

Site Preparation

We delineated seven 63-m x 27-m macroplots in LGP. Four were experimental and three control, assigned by flipping a coin for the first plot, then alternating treatments in order to spread the experimental plots across the area dominated by RCG. All macroplots were burned on 26 or 30 May 2001 to remove dense litter. Experimental macroplots were sprayed with 2% glyphosate (as Rodeo®, Monsanto, St. Louis, MO) on 23 August 2001. We allowed the seed bank to produce seedlings and repeated the herbicide treatment on 23 August 2002. To remove the standing dead biomass and litter, we burned the experimental macroplots on 24 October 2002, and spot burned any remaining material as weather allowed, finishing on 11 December 2002.

To quantify water level changes across LGP, we placed wells along transects that bisected each of the experimental macroplots and measured water levels weekly from 18 June to 1 October 2003 (Wilcox 2004). We used rainfall data for 2003, 2004, and 2005 from the UW-Arboretum weather station (NOAA 2005).

We selected 33 wetland species native to Wisconsin to sow within the experimental macroplots (Tables 1 and 2; nomenclature follows the Wisconsin State Herbarium, Madison, WI). The seed mix had even proportions of graminoids and forbs, and we favored species commonly available, affordable, and typical of southern Wisconsin wet meadows. Species were also selected that were capable of germinating

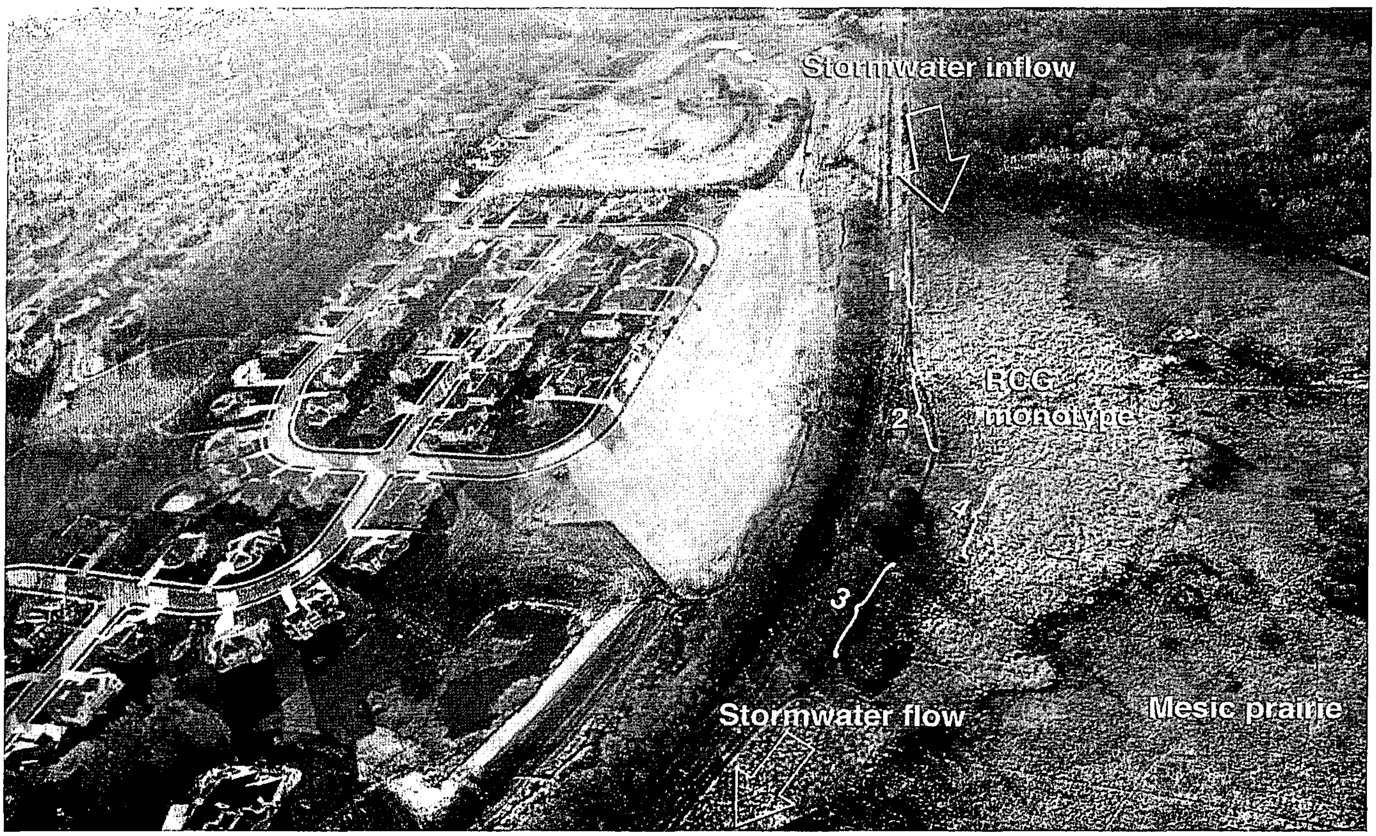

Figure 1. Aerial view of lower Greene Prairie (right) and neighboring residential areas (left) on 1 November 2005. Stormwater flows into the site from the top of the photo. We treated restoration macroplots 1, 2 and 3 with sethoxydim; we conducted the seeding and RCG clipping experiments in macroplot 4. Shadows due to late afternoon photography accentuated the reed canary grass boundary next to the prairie, and shade from trees darkened macroplot 3.

Table 1. Seeding and germination rates for the 14 graminoid species planted at lower Greene Prairie. Year collected refers to the date the seed companies collected the seed for that species.

	mg per m^2	Approx. seeds per m^2	Year collected	% germination	Year collected	% germination
Calamagrostis canadensis (Michx.) Beauv.	5.4	53	2001	47	2001	33
Carex hystericina Muhl. ex Willd.	39.8	42	1999	0	2001	61
Carex scoparia Schkuhr ex Willd.	14.3	42	2000	58	2001	47
Carex stipata Muhl. ex Willd.	36.5	46	1998	0	2001	12
Carex vulpinoidea Michx.	12.3	61	2000	78	2001	78
Eleocharis acicularis (L.) Roemer & J.A. Schultes	17.7	44	2000	3	2000	12
Eleocharis obtusa (Willd.) J.A. Schultes	12.0	42	2000	7	2000	48
Glyceria striata (Lam.) A.S. Hitchc.	8.1	46	2001	67	2001	66
Juncus effusus L.	1.6	56	1999	51	2001	16
Juncus torreyi Coville	0.9	35	1999	45	2001	41
Leersia oryzoides (L.) Sw.	35.1	42	1999	3	2001	51
Schoenoplectus tabernaemontani (K.C. Gmel.) Palla	39.6	53	1999	31	2000	41
Scirpus atrovirens Willd.	2.7	27	1998	2	2001	99
Scirpus cyperinus (L.) Kunth	1.1	68	2001	89	2001	93

under water to accommodate increased water levels at LGP (Middleton 1999). Fifteen species were recorded as being historically present at or near the site, but none of these species were common at LGP prior to seeding (Allsup 1977; Werner and Zedler 2002; S. Kercher, unpubl. data). Coefficients of conservatism (C; Swink and Wilhelm 1994) for the sown species ranged from two to seven using Wisconsin ratings (Bernthal 2003).

Separate batches of seeds were ordered from local suppliers for use in fall and spring (Wilcox 2004). Ten species had seeds collected in the same year for both batches (Tables 1 and 2). The species composition of the two batches was identical except that in the fall batch, we were sent *Rumex orbiculatus* instead of *Rumex verticillatus*. This error was detected after species were sown and had flowered.

Areas within experimental macroplots designated as "fall" were seeded in December 2002, and "spring" macroplots were seeded in May 2003. During both seeding periods, seeds were broadcast at a rate of 1300 seeds per m^2 (Wilcox 2004). This high seed rate was selected to give many different native species an opportunity to compete with RCG seedlings, since reed canarygrass heavily dominated the seed bank and competition among seedlings was inevitable. The fall batch overwintered in the field, and the spring batch was moist-cold stratified at 5° C for eight weeks before sowing.

Germination

To test the germination rates of each batch of seeds, we moist-cold stratified 100 seeds of each species at 5° C for eight weeks. Those seeds were then placed in a growth chamber set for 14 hours of light at 30° C and 10 hours of dark at 20° C for one month (Baskin and Baskin 1998; van der Valk et al. 1999). We counted any seed with an emerging radicle during that month as "germinated." Data from the two *Rumex* species were not used to compare seed batches.

Seeding and RCG Clipping Experiments

Two experiments were established using 4 x 4 Latin squares in the experimental macroplot with the least through-flowing water and lowest potential for seed movement. Each square had four treatments replicated four times, yielding thirty-two 5-m x 6.5-m plots in the macroplot.

The Latin square used for the seeding experiment varied both timing and intensity (no seeding, fall seeding, spring seeding, and seeding in both fall and spring). The RCG clipping experiment compared removal of RCG with fall seeding, removal of RCG with spring seeding, no RCG removal plus fall seeding, and no RCG removal plus spring seeding. The RCG removal treatment was applied in June 2003. We cut RCG culms taller than ~15 cm and applied glyphosate to the culm stumps. An average of 560 ± 140 g (mean ± SE) of RCG biomass (dried for at least

Table 2. Seeding and germination rates for the 19 forb species planted at lower Greene Prairie. Year collected refers to the date the seed companies collected the seed for that species.

	Seeding rate		Fall batch		Spring batch	
	mg per m^2	Approx. seeds per m^2	Year collected	% germination	Year collected	% germination
Acorus americanus L.	152.3	35	1999	74	1999	89
Alisma subcordatum Raf.	16.6	41	1999	1	2000	0
Asclepias incarnata L.	78.8	13	2001	53	2002	76
Aster novae-angliae L.	15.3	38	2001	9	2001	31
Aster puniceus L.	15.7	42	2000	4	2002	97
Bidens cernuus L.	47.6	35	2001	73	2001	90
Boltonia asteroides (L.) L'Hér	6.5	37	1999	38	2001	38
Epilobium coloratum Biehler	2.2	39	1998	49	2002	86
Eupatorium maculatum L.	10.5	35	1999	69	2000	79
Eupatorium perfoliatum L.	6.1	35	1999	6	2001	86
Euthamia graminifolia (L.) Nutt.	2.7	33	2000	51	2002	64
Helenium autumnale L.	7.5	37	1999	38	2002	95
Lobelia siphilitica L.	1.9	38	2001	76	2001	70
Lycopus americanus Muhl. ex W. Bart.	7.6	35	2000	25	2001	78
Mentha arvensis L.	3.1	32	2000	68	2001	36
Mimulus ringens L.	0.9	48	1998	26	2001	91
Rumex sp. (*R. orbiculatus* A. Gray or *R. verticillatus* L.)	55.0	19	2001	44	2000	16
Sagittaria latifolia Willd.	16.5	36	1999	34	1999	37
Verbena hastata L.	11.2	56	2000	70	2002	89

48 hours at 65°C) was collected from each removal plot.

Once all treatments were complete, we randomly placed five 0.1-m^2 quadrats within plots of each experiment. In 2004, the five 0.1-m^2 quadrats were randomly placed in locations different from the previous year. By 2005, species purposefully sown into plots could have spread naturally into control plots, so no further data were collected.

Restoration Experiment

Water gradually flowed across the three experimental macroplots of the restoration experiment from west to east, and we expected some seed might move with the spring snowmelt. Thus, we seeded the western half of each macroplot in fall, when the site was dry, and the eastern half in spring, when localized areas had standing water, yielding a total of six 11.5-m x 29.5-m seeded plots. There were no storm events immediately following the seeding periods. We placed twelve 0.1-m^2 quadrats randomly within each of the six plots, for a total of 72 quadrats. In summer 2004, we placed twelve 1-m^2 quadrats with nested 0.1-m^2 quadrats in new random locations within each seeded plot and eight 1-m^2 quadrats with nested 0.1-m^2 quadrats within each of three control macroplots (where variability was low due to the monotypic vegetation).

In fall 2004, we established new RCG management treatments in the southern half of each of the three experimental macroplots, bisecting the seeding treatments and yielding a total of twelve 11.5-m x 14.8-m plots. On September 29, 2004, we treated the six RCG management plots with 2.25% sethoxydim (Vantage®, MicroFlo Company LLC, Memphis, TN), a grass-specific herbicide, and 1% non-ionic spreader activator (Activate Plus®, Terra International, Inc., Sioux City, IA). We then burned these plots on 1 November 2004 to remove the standing dead grasses. We repeated the 2.25% sethoxydim treatment on 3 June 2005 using 1% water conditioner (Request®, Helena Chemical Company, Collierville, TN) and 1% ammonium sulfate non-ionic sticker-spreader (DyneAmic®, Helena Chemical Company, Collierville, TN) with the herbicide. Once all treatments were completed, we randomly placed eight 1-m^2 quadrats within each of the twelve plots in the experimental macroplots, for a total of 96 quadrats.

Data Collection and Analyses

We sampled the vegetation within quadrats during three periods: 22 July to 4 August 2003; 20 July to 4 August 2004; and 24 to 29 August 2005 (years one, two, and three, respectively). During each sampling period, we recorded the presence of all vascular plant species rooted within quadrats. We visually estimated cover of each species using six cover classes (<1%, 1-5%, 6-25%, 26-50%, 50-75%, and >75%) modified from Braun-Blanquet (1932), and we recorded the height of the tallest individual of each species, measured from plant base to the highest stretched leaf or flower.

We used cover class midpoints to calculate mean cover. When we computed mean cover, we assigned a zero to quadrats where a species (or group of species) was absent. Mean C for quadrats was calculated by averaging the C values of species found in the quadrat.

Correlation among germination rates in different batches and between germination rates and field frequencies were tested using Spearman's rank procedure. For the seeding experiment, we tested treatment effects with one-way ANOVAs. Two-way ANOVAs with RCG removal and seeding as independent variables were conducted for the RCG clipping experiment. If ANOVAs were significant, we tested differences among treatment means using Fisher's LSD procedure. We examined differences between experimental and control treatments in the restoration experiment using Welch's two-sample t-tests. We conducted statistical tests using R 2.2.1 (R Development Core Team 2005), except LSD tests, which we analyzed using SAS 8.0 (SAS Institute 2000). Tests were considered significant at the $p \leq 0.05$ level, unless noted otherwise.

RESULTS

Environmental Conditions

Mean monthly precipitation in Dane County, Wisconsin, for May, June, July, August, and September was 8.3, 10.3, 10.0, 11.0, and 7.8 cm, respectively (data from 1971-2000 at the Dane County Regional Airport; NOAA 2005). In 2003, mean precipitation measured at the Arboretum was 9.4 cm higher than normal in May, 8.2 cm lower than normal in August, and 3.2 cm higher than normal in September (NOAA 2005). In 2004, mean precipitation was 20.0 cm higher than normal in May and 2.8 cm lower than normal in August (NOAA 2005). 2005 was a drier than normal year, with mean precipitation for the April-September period nearly 5 cm below normal each month with the exception of May (NOAA 2005).

Water levels measured over 16 weeks in 2003 showed that inundation varied across the site and over time (Figure 2). On average, water levels were lower in the seeding and RCG clipping macroplot than in the restoration macroplots and water levels at any given well followed rainfall patterns.

Germination Rates

In the laboratory, germination rates of the 33 sown species ranged from 0 to 99% (Tables 1 and 2). The mean rate in the fall seed batch (39 $\pm$ 5%) was two-thirds that of the spring batch (60 $\pm$ 5%). There was no correlation between germination rates of the same species in the two batches ($r^2 = 0.29$, $p = 0.1$). Ten of the species had been collected in the same year, and their fall and spring germination rates were strongly correlated ($r^2 = 0.87$, $p = 0.003$), while rates of the 22 species collected in different years were not correlated ($r^2 = 0.14$, $p = 0.52$).

As predicted, germination rates and species' field abundances were correlated, with a stronger relationship for the spring batch ($r^2 = 0.56$, $p = 0.001$ for lab rate x frequency in all quadrats seeded in spring and sampled in year one). The correlation for the fall batch sampled at the same time was low ($r^2 = 0.32$, $p = 0.08$). In year two, neither the spring nor fall batch was correlated with field occurrences ($r^2 = 0.24$, $p = 0.18$; $r^2 = 0.16$, $p = 0.37$, respectively). *Alisma subcordatum, Carex stipata, Leersia oryzoides,* and *Mentha arvensis* were found more frequently in the field in years two and three than predicted by germinations rates. Species with germination rates greater than 50% that were uncommon in the field throughout the experiment included *Carex vulpinodea, Eupatorium*

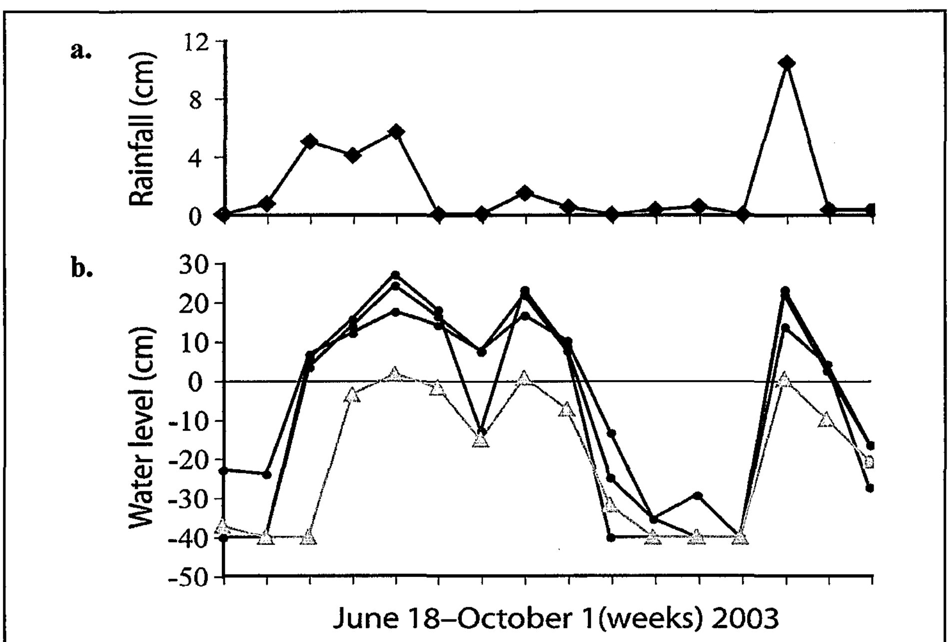

Figure 2. Comparison of rainfall and water levels at lower Greene Prairie during year one. a) Total rainfall from the previous week. b) Water levels (ground level = 0 cm) measured each week for 16 weeks. Lines with circles are for the experimental macroplots of the restoration experiment, and the line with triangles is for the seeding and RCG clipping experiment.

maculatum, *Glyceria striata*, and *Scirpus cyperinus*.

Seeding and RCG Clipping Experiments

In the seeding experiment, plots that were not seeded had the fewest native species (only about one native species per 0.1 m^2 quadrat) and the lowest mean C. RCG reestablished 52% cover in year one and 72% in year two and grew 57 cm tall in year one and over three times that in year two (Tables 3 and 4).

With seeding of native species, RCG was still abundant, but more species were present, especially when seeds were sown in spring. Native species richness was greater in the spring and fall+spring seeded treatments than without seeding, but high native species richness did not persist through the second year (Tables 3 and 4). The changes in mean C showed a similar pattern. In year one, species with $C \geq 2$ had significantly greater cover in seeded treatments than the non-seeded treatment, but this difference disappeared during year two. Double seeding (spring+fall) never increased native species cover beyond that achieved with single seeding in spring or fall. Although several sown species seemed to establish differentially with season of seeding, only four species were common enough for differences in cover among treatments to be significant (Table 4).

The seeding treatments had no effect on RCG cover, RCG height, or cover of other weedy species such as *Polygonum hydropiper* L. (Tables 3 and 4). Cover of weedy species ($C < 2$) increased in year two.

In the RCG clipping experiment, we found no treatment responses in species richness, species cover, species height, or mean C for the RCG removal factor. The seeding factor of this experiment produced patterns similar to those in the seeding experiment.

Restoration Experiment

Where attempts were made to remove RCG, the species sown in LGP established, as well as volunteer species. Pooling data from all field experiments and years of study, we identified 75 species: 52 in year one, 48 in year two, and 52 in year three. Most of these species were native perennial forbs, but the most common forb found at LGP was *Polygonum hydropiper*, a volunteer annual. Non-sown native species included *Carex lacustris* Willd., *C. tribuloides* Wahlenb., *C. trichocarpa* Willd., *Helianthus grosseserratus* M.Martens, *Lycopus uniflorus* Michx., *Lythrum alatum* Pursh, *Poa palustris* L., *Ranunculus pensylvanicus* L.f., and *Scutellaria galericulata* L., but these volunteers were found in fewer than 10% of quadrats in the restoration experiment.

Most of the 33 sown species were encountered in the restoration experiment; of these, nine were found in more than 10% of sampled quadrats in the restoration experiment in both years two and three (Table 5). The four sown species recorded in LGP in the first two years but not in year three were all forbs: *Boltonia asteroides*, *Eu-*

Table 3. Species richness, mean coefficient of conservatism (C), and species height in each treatment of the seeding experiment in late July of 2003 (year one) and 2004 (year two) and the analysis of variance for treatment effects. Fall = fall seeded plots, Spring = spring seeded plots, Fall + Spring = seeded in the fall and spring, Control = not seeded. Means with different letters were significantly different following LSD tests. *Overall mean for year one differed significantly from year two, p < 0.05.

Group		Treatments: Fall	Spring	Fall + Spring	Control	F	p
Richness (per 0.1 m^2)							
All species	Year 1	4.4ab	6.8a	6.6a	3.2b	5.6	0.04
	Year 2	3.6bc	4.5ab	5.2a	2.6c	11.2	0.01
Native species*	Year 1	2.9ab	5.3a	4.8a	1.2b	7.0	0.02
	Year 2	1.9bc	3.1ab	3.9a	0.9c	12.7	0.01
Coefficient of conservatism							
Mean C	Year 1	2.4a	3.1a	2.9a	1.1b	8.8	0.01
	Year 2	1.8b	2.8a	3.3a	1.0b	16.1	<0.01
Height (cm)							
*Phalaris arundinacea**	Year 1	45.4	60.7	46.5	56.9	0.4	0.76
	Year 2	147.7	173.0	133.9	176.1	1.3	0.37
Native species*	Year 1	37.6	35.8	38.5	34.7	<0.1	0.99
	Year 2	55.8	61.2	61.2	38.4	1.7	0.27

patorium maculatum, *Lobelia siphilitica*, and *Mimulus ringens*. Conversely, *Carex hystericina*, *Juncus effusus*, and *Scirpus cyperinus* were the three graminoids found only during year three. Sown species never encountered at LGP after three years were all graminoids: *Calamagrostis canadensis*, *Eleocharis acicularis*, *Juncus torreyi*, and *Scirpus atrovirens*.

In year one, RCG was found in only 33% of experimental quadrats (seeded and treated with glyphosate and fire) with mean cover of 23.6% and mean height of 60.6 cm. By year two, the RCG that occurred in experimental quadrats more than doubled in height from year one and was about 85% as tall as that in control plots, a significant difference ($p<0.01$; 160 cm versus 187 cm, respectively). However, mean cover of RCG in experimental and control plots in year three was not significantly different ($p>0.05$; 41.6% versus 57.0%, respectively). RCG remained the monotypic dominant in the control macroplots, where native species richness averaged 0.5 native species/ 1 m^2 quadrat. By comparison, our restoration treatment produced a significant increase of 4.5 native species/ 1 m^2 quadrat ($p<0.001$).

Experimental plots treated with sethoxydim in year three had shorter RCG, less RCG cover, and greater native species richness and floristic quality than untreated experimental plots (Table 6). The two most common grass species other than RCG, *Echinocloa muricata* and *Leersia oryzoides*, did not have significantly less cover in sethoxydim-treated plots compared to untreated experimental plots. Despite the reduction in RCG abundance, RCG remained the dominant in all experimental plots treated with sethoxydim.

DISCUSSION

Altered hydrologic conditions and reed canary grass invasions severely limit restoration of wetland biodiversity in Wisconsin wetlands (Bernthal and Willis 2004). After three years, we found that our restoration efforts did not achieve the desired outcome of a dense native plant community that could resist RCG reinvasion. Our treatments increased species richness, but RCG readily regained dominance. There were, however, three unexpected findings from our experimentation. First, germination

Table 4. Cover (%) of species or groups of species for each treatment of the seeding experiment in late July of 2003 (year one) and 2004 (year two) and the analysis of variance for treatment effects. Cover for groups of species was determined by summing cover measured for individual species. Fall = fall seeded plots, Spring = spring seeded plots, Fall + Spring = seeded in the fall and spring, Control = not seeded. Means with different letters were significantly different following LSD tests. *Overall mean for year one differed significantly from year two, p < 0.05.

		Treatments					
Group		Fall	Spring	Fall + Spring	Control	F	p
Cover (%)							
*Phalaris arundinacea**	Year 1	24.8	37.3	28.9	52.5	2.0	0.21
	Year 2	58.3	75.9	52.1	72.1	1.4	0.33
*Bidens cernuus**	Year 1	28.6a	14.0b	22.4ab	0.0c	10.6	0.01
	Year 2	7.3	0.0	6.1	0.0	1.0	0.45
Eupatorium perfoliatum	Year 1	0.0	2.1	0.3	0.0	1.9	0.22
	Year 2	0.0b	2.0a	0.4ab	0.0b	4.0	0.07
Lycopus americanus	Year 1	0.2b	3.7ab	7.4a	1.1b	3.4	0.09
	Year 2	0.3	4.3	6.9	1.8	3.3	0.10
Polygonum hydropiper	Year 1	14.5	17.3	16.7	13.4	<0.1	0.99
	Year 2	16.6	1.3	11.0	4.2	1.3	0.36
*Verbena hastata**	Year 1	2.3b	10.4a	6.1ab	0.2b	5.8	0.03
	Year 2	0.0	0.5	0.2	0.0	2.1	0.20
Cover of groups of species (%)							
Species with C < 2*	Year 1	43.9	56.2	49.4	68.4	1.8	0.24
	Year 2	74.9	77.2	63.1	77.9	1.0	0.45
Species with C ≥ 2*	Year 1	67.2a	72.2a	63.4a	13.1b	6.9	0.02
	Year 2	28.2	26.0	32.9	8.3	1.8	0.24

potential (rather than seeding treatments) appeared to explain the establishment of several species; second, forbs were the most common sown species that established, not graminoids; and third, fall+spring seeding did not enhance native species establishment. Below we highlight the dual challenge of removing RCG and reestablishing native species in the presence of stormwater inflows.

Resilience of RCG

Although glyphosate initially killed most of the RCG canopy, RCG reestablished from seed and from surviving rhizomes. Within two years, RCG height and cover in treated macroplots were similar to control macroplots. Foster and Wetzel (2005) also found that a one-time treatment of glyphosate initially reduced RCG; however, by the second year, RCG cover was not different from unsprayed areas.

Table 5. Frequency of occurrence in 1-m^2 quadrats for the 13 species found with > 10% frequency in either 2004 or 2005 within the three seeded macroplots of the restoration experiment.

	2004 frequency	2005 frequency
Graminoids		
> 10% increase in frequency		
*Carex stipata**	6	19
< 10% change in frequency		
Phalaris arundinacea	92	96
*Leersia oryzoides**	32	34
*Schoenoplectus tabernaemontani**	14	18
Echinochloa muricata (P. Beauv.) Fernald	4	13
Forbs		
> 10% increase in frequency		
Polygonum hydropiperoides Michx.	1	15
Polygonum sagittatum L.	0	11
< 10% change in frequency		
*Mentha arvensis**	40	34
*Alisma subcordatum**	31	23
*Bidens cernuus**	24	26
*Helenium autumnale**	24	16
*Asclepias incarnata**	22	14
*Rumex orbiculatus**	17	8
*Verbena hastata**	17	11
*Eupatorium perfoliatum**	13	5
*Acorus americanus**	11	9
> 10% decrease in frequency		
Polygonum hydropiper L.	76	63
*Lycopus americanus**	33	21
*Aster puniceus**	29	16
*Epilobium coloratum**	15	5
*Boltonia asteroides**	11	0

* *Sown species*

At LGP, water levels fluctuated widely within each growing season but rarely exceeded 20 cm in depth, an unnatural condition caused by urban runoff that likely favored RCG recovery. RCG thrives under continuous flooding with shallow water (~15 cm; Kercher and Zedler 2004b) but is also highly tolerant of variable hydroperiods (Galatowitsch et al. 1999; Kercher and Zedler 2004a). Coops et al. (1996) found that RCG has less tolerance of constant water depths of more than 35 cm. Stormwater entering LGP carries dissolved nutrients and sediments that can also assist RCG reestablishment (Green and Galatowitsch 2001; Werner and Zedler 2002; Kercher and Zedler 2004b). In a greenhouse study, Perry et al. (2004) found that RCG outcompeted the native sedge *Carex hystericina* in nitrogen-rich soil but not in soils with low inorganic nitrogen availability (< 30 mg kg^{-1}). Similarly, in a mesocosm study, a grass-specific herbicide had little effect on RCG biomass when flooding and nutrients were continued, but native species were favored where these "stormwater disturbances" were discontinued (Herr-Turoff 2005).

The openings in the early-season canopy that we created by clipping RCG helped both RCG and native seeds germinate and establish. Contrary to our prediction, clipping RCG did not decrease end-of-season cover of RCG. More frequent clipping might have been more effective. In a Canadian prairie restoration, clipping all vegetation four times a year for two years reduced cover of the introduced cool-season grass *Agropyron cristatum* (L.) Gaertn. 90% and doubled native grass cover (Wilson and Pärtel 2003).

We expected seeding to produce native canopies that could reduce RCG dominance, but this was not observed. In the

Table 6. Species richness, mean C, height and cover of selected species for experimental plots treated and untreated with sethoxydim in year three (late August 2005). All experimental plots were previously treated with glyphosate and seeded with native species.

Group	Sethoxydim	No sethoxydim	P-value
Native species richness	5.5	2.1	<0.001
Mean C	2.8	1.8	<0.001
Height (cm)			
Phalaris arundinacea	123.9	185.7	<0.001
Cover (%)			
Phalaris arundinacea	47.0	66.8	<0.001
Echinocloa muricata	2.2	2.6	0.985
Leersia oryzoides	2.9	6.1	0.143
Species with $C \geq 2$[a]	50.2	15.2	<0.001

[a]Cover for this group of species was determined by summing cover measured for individual species with $C \geq 2$.

first year of the seeding experiment, the cover of moderately conservative native species ($C \geq 2$) was about twice that of RCG cover in seeded plots, but RCG was already about 20% taller than the native species. In the second year, RCG had about twice as much cover and height as moderately conservative natives in seeded plots. While native species were common, they did not overtop RCG. *Asclepias incarnata* was the only sown species with height comparable to RCG in summer 2003, but it subsequently suffered from herbivory (pers. observation). Other studies examining the establishment of sown natives in relation to introduced grasses have had conflicting results (i.e., native establishment decreased exotic abundance in a Kansas prairie (Kindscher and Tieszen 1998), and natives had no effect in a Canadian prairie or Tennessee wetland (Wilson and Pärtel 2003; Foster and Wetzel 2005, respectively)).

Herbicide use, combined with burning, was the only action that decreased RCG abundance in our study. Wilson and Pärtel (2003) found they could reduce cover of a widespread invasive grass and increase cover of a native grass by selectively applying glyphosate to the invasive grass over a period of seven years, but spraying a broad-spectrum herbicide on one targeted species without killing newly established natives is challenging. A more promising approach is to use an herbicide that selectively kills grasses. Our application of sethoxydim reduced RCG cover and height by about 30%, providing an opportunity for natives to persist. However, Annen et al. (2005) found that the RCG biomass in sethoxydim-treated plots was no different from control plots in the year following application. Additionally, the efficacy of sethoxydim treatment can be reduced by UV light, pH, temperature, and choice of adjuvants (Harker 1995; Matysiak 1999). Even after use of a selective herbicide, RCG remained the dominant species at LGP, and we expect that continuous applications would be needed to minimize RCG abundance.

Native Species Establishment

In some settings, native vegetation can be recruited from the seedbank or from neighboring communities after exotic vegetation is removed or human disturbances are reduced (D'Antonio et al. 1998; Kellogg and Bridgham 2002). However, natural recolonization proved to be insufficient for recovering diverse vegetation in LGP. Unseeded plots had low species richness, and native species that volunteered into the site remained uncommon. In such cases, it is necessary to sow seeds or plant seedlings to restore diversity (Hutchings and Booth 1996; Bakker and Berendse 1999).

Higher germination rates for the spring seed batch explained greater establishment of native species in spring than fall. *Aster puniceus*, *Eupatorium maculatum*, *Helenium autumnale*, and *Mimulus ringens* all had higher spring than fall germination rates and were mostly found in spring seeded plots or fall+spring seeded plots. However, only the spring germination rates were significantly correlated with field frequency, indicating that other factors, such as seed predation over winter or flooding in early spring, decreased establishment of some species in fall-seeded plots. Overall, we conclude that using seeds with high laboratory germination rates was important to encouraging native species establishment.

We expected fall seeding to favor some species, spring to favor others, and double seeding to include both groups. Instead, spring and fall+spring seeding produced similar assemblages, but we are unsure of the reasons. The one-time seeding rate was 1300 seeds/m^2, which was already high, and doubling it could have supersaturated the site or attracted more granivores. Furthermore, we are cautious about attributing outcomes to seeding once vs. twice, because the single and double seeding occurred at different times, and seeds used in fall and spring seeding differed in germination rates.

At the end of year three, the forb composition at LGP was similar to that of southern Wisconsin sedge meadows. Three of the nine forbs that Curtis (1959) found to be common in natural sedge meadows were also common at LGP (*Asclepias incarnata*, *Aster puniceus*, and *Lycopus americanus*), and three more were present (*Eupatorium maculatum*, *Eupatorium perfoliatum*, and *Galium obtusum* Bigelow). Two annual forbs, *Bidens cernuus* and *Polygonum hydropiper*, were abundant in year one but decreased in frequency and cover in year two, as expected of annual plants in restoration sites (Pywell et al. 2003). Both of these species subsequently increased in quadrat frequency in year three compared to year two, likely due to the disturbance introduced using fire and herbicide treatments. In years two and three, the frequency and cover of many sown forbs noticeably decreased, and four dropped out. The overall decreased abundance of native species was probably due to the dense canopy cover and root and rhizome mats produced by RCG.

Unlike our restoration site, Wisconsin's natural wet meadows have many native graminoids (e.g., *Carex* species and *Calamagrostis canadensis* typically dominate (Curtis 1959)). We did not sow *Carex strica* seeds because establishment from seed is known to be difficult (van der Valk et al. 1999). We sowed other species of *Carex*, but only *C. stipata* was found in more than 10% of quadrats by year three. Two other native graminoids that became common include *Leersia oryzoides* and *Schoenoplectus tabernaemontani*, which are characteristic of emergent aquatic communities (Curtis 1959) and accordingly were found in depressions with standing water.

Graminoids often establish easily in restoration sites due to greater herbivore resistance and more rapid spread than forbs (Pywell et al. 2003). However, despite sowing equal proportions of graminoids and forbs, few native graminoid species became common at LGP. Perhaps the sown graminoids recruited poorly from seed (Hölzel and Otte 2003); perhaps they need to be restored by planting seedlings. Planting seedlings can increase *Carex* establishment, but Budelsky and Galatowitsch (2004) found that *Carex stricta* seedlings did not persist when exposed to prolonged flooding or when suppressed by RCG. In a mesocosm study, Green and Galatowitsch (2002) found that nitrate addition favored growth of wetland forbs over wetland graminoids, a pattern that could explain the abundance of forbs at LGP. However, three sown graminoids were not found until year three, perhaps indicating that the native graminoids may simply need more time to establish (provided RCG abundance can be controlled).

Our initial restoration efforts (applying glyphosate, burning the thatch to expose the soil, and seeding 33 native species) failed to establish a dense, tall, canopy of native species that could resist RCG reinvasion. The fact that native grasses were a minor component at LGP opened the door for control of RCG using a grass-specific herbicide. Restoration plots treated with sethoxydim had a greater species richness, mean C, and cover of moderately conservative species than untreated plots. We were surprised that cover of *Leersia oryzoides* and *Echinocloa muricata* was not significantly lower in sethoxydim-treated plots compared to untreated experimental plots. These grasses might owe their persistence to interception of sethoxydim by dense RCG. Subsequent treatments could negatively affect these grass species.

Conclusions

We conclude that land managers should be selective in allocating efforts to control monotypes of invasive species. Where wetlands continually receive stormwater, the replacement of invasive monotypes by native species will be difficult and the effort required might be prohibitively expensive and ineffective in the long-term. Highest priority should go to sites that can best support native species restoration. If the invasive species is RCG, optimal sites will be ones where stormwater inflows can be treated or curtailed.

Herbicide following fire was the most effective combination of tools to reduce RCG in our study, but we encourage further experimentation on the use of herbicide to control RCG. We need to know more about the effects of grass-specific chemicals on native graminoids and wetland fauna prior to widespread use of such herbicides to control RCG. Even if sethoxydim proves to be benign, continual application might still be ineffective, because members of the *Phalaris* genus have developed resistance to this herbicide (Tal and Zarka 1996). We agree with Lavergne and Molofsky (2006) that future restoration projects should include plans to manage weeds.

We found that using fresh seed that performed well in germination tests was important to establish native species in the field. We recommend taking advantage of opportunities to test seed mixes so that seeds purchased in bulk are species that will most likely thrive and compete with RCG. Because only a small portion of the species sown to LGP developed >10% cover, we advise small-scale experimentation prior to whole-site planting to tailor native plant seed mixes to variable habitats.

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