A Comparison of *Phragmites australis* Control Measures in Wisconsin Coastal Wetlands

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ABSTRACT

The common reed, *Phragmites australis*, is a highly aggressive species that outcompetes native species on shorelines, wetlands and roadways across the Great Lakes Basin. Prior to my analysis glyphosate and imazapyr (Habitat 7) herbicides were applied to more than 600 acres at selected sites along the west shore of lower Green Bay. I explored alternative secondary control treatments for increasing the effectiveness of the aerial herbicide treatment. Sample plots were evaluated in 4 treatment categories: 1) herbicide 2) herbicide + burning, 3) herbicide + mowing, and 4) control (no treatment). Measurements within plots included percent cover of *P. australis* and all other plant species, average litter depth, and average height of *P. australis*. Results suggest that local eradication of *P. australis* is difficult, if not impossible, with one set of treatments. Treatment of *P. australis*, however, is the first step in restoring native wetlands to pre-invasion condition. Percent cover of native wetland species was much higher at treated plots compared with untreated plots. Certain native plant species can rather quickly re-populate treated wetlands after treatment. In this study, native species were found in untreated control plots, but the low species abundance and poor condition of plants in these plots suggests that time might be critical to the treatment of *Phragmites* stands. Native wetland plants probably will not persist for many years after *P. australis* invasion. I conclude that application of a secondary treatment of mowing or burning after herbicide application increases the success of *Phragmites* control and improves the ability of native species to re-establish populations in Great Lakes coastal wetlands.

INTRODUCTION

Phragmites australis, more commonly known as giant reed or common reed, is a perennial grass found on every continent except Antarctica (Tucker, 1990). While native genotypes of *P. australis* are found in the United States, non-native strains of *P. australis* are increasingly viewed as undesirable invasive plants. The prolific growth and spread of *Phragmites* over the past century have contributed to the replacement of diverse marsh plant communities with species-poor *P. australis* monocultures (Saltonstall, 2002, Roman et al., 1984; Marks et al., 1994, Fell et al. 2006). While *P. australis* is known to colonize disturbed sites, pristine areas also are being invaded (Phillips, 1987; Marks et al., 1994). Because *P. australis* can propagate through seed dispersal, rhizomes, and stolon fragments, it is an extremely successful colonizer of disturbed sites (Tucker, 1990, Chambers et al., 1999). Vegetative spread by rhizomes can result in stands as dense as 200 stems/m^2 (Haslam, 1972).

The rapid spread of *P. australis* is aided by human activities, particularly the clearing of native vegetation and nitrogen eutrophication related to shoreline development (Bertness et al., 2002). Studies in Connecticut comparing un-impounded and impounded brackish marsh systems showed that human alteration of natural hydrological cycles allows *P. australis* to enter a system and quickly dominate (Harrison and Bloom, 1977; Roman et al., 1984). In just three decades in the lower Connecticut River area, over 90% of *Typha/Scirpus* marshes have been replaced by near or complete *P. australis* monocultures. Creek and river banks and the upland border of the brackish meadow complex are now almost entirely *P. australis* dominated (Warren and al., 2001).

The characteristic dense growth of *P. australis* significantly affects the systems that it invades. Plant height and stem density reduce light at the soil surface and modify air temperatures within the *P. australis* stand. These factors may inhibit the germination or establishment of other plant species and may prolong the decomposition of organic matter. Additionally, low light levels in *P. australis* stands resulting from this biomass accumulation can significantly delay spring thawing of marsh substrates, further preventing establishment of other non-*Phragmites* plant species (Meyerson et al., 2000). The overall result of *P. australis* establishment is usually a reduction in plant species richness, particularly in the more diverse freshwater marsh systems. This may contribute to the loss of rare species already threatened by small population size. Studies comparing *P. australis*-dominated to *P. australis*-free regions within the same marsh have demonstrated higher species diversity in *P. australis*-free regions. This result was observed in all marsh types, but was especially evident in freshwater non-tidal marshes where the total number of species in *P. australis*-free regions showed a twofold increase over *P. australis*-dominated areas. Relatively high plant diversity can be recorded throughout a *P. australis*-dominated stand, but individuals of the other species are often sterile, widely scattered, and do not represent viable populations (Meyerson et al., 2000). In addition, dead culms of *P. australis* can remain standing for two or more years, inhibiting the establishment and growth of other marsh species (Meyerson et al., 2000).

P. australis management under most circumstances is technically challenging and resource intensive. Methods to control or eradicate *P. australis* have included herbicide application of glyphosate and imazapyr (Habitat 7) by aerial, backpack, and manual application. Mowing and burning treatments have also been attempted. In 1994, the state of Connecticut began a series of *P. australis* control treatments, combining mowing and herbicide application in the marshes along the lower Connecticut River (Warren and al., 2001). The research concluded that *P. australis* is extremely sensitive to glyphosate treatments (Warren and al., 2001). Late summer herbicide followed by spring mowing of *P. australis* can re-establish brackish meadow vegetation with little or no impact on the macroinvertebrate community or fish use of the high marsh. The mow-only treatment was not an effective method of *P. australis* control. The mowing depressed *P. australis* aboveground production and increased stem density. Any eventual restoration of a brackish meadow is likely to be much slower if herbicide treatment is not coupled with mowing of standing dead shoots. Brackish meadow was restored within two to three years of herbicide/mow treatment. Some *P. australis* culms survived the herbicide/mow treatment, however, and *P. australis* began to expand again in the area. (Warren et al., 2001) concluded that long-term control of *P. australis* in brackish tidal marshes will likely require ongoing management. On the other hand, Galinato and van der Valk (1986) found that colonization rates of other marsh plants were rapid following *P. australis* eradication treatments, and in most cases species diversity markedly increased. Species diversity in freshwater sites rose quickly, possibly due to germination of seeds in the soil being released by higher light levels.

In Wisconsin, *P. australis* is most commonly found in disturbed areas with altered hydrology or sedimentation. Inhabited sites include roadside ditches, wetlands downhill from active farms, and farmed wetlands. *P. australis* also is found in interior wetlands. In the most recent decade, lower water levels in the Great Lakes have facilitated an explosion in *P. australis* colonization along the Lake Michigan shoreline (Wisconsin

Wetlands, 2006). In Door County, Wisconsin, one of the most bio-diverse counties in the Midwest, *P. australis* has invaded virtually every coastal natural area.

During 2005 and 2006 the U.S. Fish and Wildlife Service (USFWS) and Wisconsin Department of Natural Resources (WIDNR) applied the herbicide Habitat 7 to more than 600 acres at selected sites along the west shore of lower Green Bay in an effort to control the spread and potentially eradicate *P. australis*. Based on successes of control measures in the eastern U.S., managers are exploring alternative secondary control treatments for increasing the effectiveness of the aerial herbicide treatment. For example, does mowing or prescribed burn following the application of the herbicide increase the growth or recolonization of native wetland species?

In this paper, we will evaluate a series of experimental treatments of *Phragmites* along Green Bay and Lake Michigan, following one of the most extensive *P. australis* control efforts in the Great Lakes Region. Our goal is to help identify the most effective, least expensive, and most ecologically benign control methods for *P. australis* in coastal wetlands. Results will help determine if current *P. australis* control measures are effective and if harm can be minimized to native beach, dune and shore species along Wisconsin's Lake Michigan shore (including Green Bay).

METHODS

Study Sites

Monitoring of *P. australis* treatments was conducted at four localities in northeastern Wisconsin (Figure 1).

Figure 1. Map of study sites. 1) Long Tail Point, 2) Ridges Sanctuary, 3) Peshtigo Harbor, and 4) Seagull Bar.

Long Tail Point

Long Tail Point, owned by the State of Wisconsin, consists of 243 hectares located on the west shore of lower Green Bay, three kilometers east of Suamico in northern Brown County. This narrow spit and associated embayment is bordered on the north by the mouth of the Suamico River. Soil characteristics are poorly drained lake plain soils with the water table at or near of the surface of the entire spit. The size and shape of the wetland is dependent on the irregular fluctuations in the water level of Green Bay.

The vegetation on Long Tail point is dominated by *Salix fragilis* (crack willow) and *Populus deltoides* (cottonwood) at the highest ground. Emergent marsh consisting of *Typha angustifolia*(cat tail), *Scirpus validus*, and *Scirpus americanus* is located westward, along with sedge meadows dominated by *Calamagrostis canadensis* (blue joint) and *Typha angustifolia*. Invasives *P. australis* and *Lythrum salicaria* (purple loosestrife) are threatening to displace native species in these wetlands (WIDNR, 2005).

Because extensive areas of wetlands have been destroyed or degraded by filling, residential development, hydrologic alterations, pollution, and the spread of invasive species, Long Tail Point provides critical habitat for native plants and animals in the lower Green Bay ecosystem, including historical habitat for the endangered Piping Plover. As a migratory stopover point, wetlands in lower Green Bay also are important for waterfowl, shorebirds, raptors and passerines. This site is an important and integral component of the productive wetland remnants in lower Green Bay (WIDNR, 2005).

Treated study sites on Long Tail Point were located on the northern end of the spit, west of the tree line in the emergent marsh and sedge meadows. Control sites were located on the northeast shoreline of Green Bay and along the northern boundary of the property along Riverside Road.

The Ridges Sanctuary

The Ridges Sanctuary occupies an area of approximately 650 ha along the eastern edge of the Door Peninsula. The majority of the Ridges Sanctuary is bounded by the north shore of Baileys Harbor, County Highway Q, and the western shores of Moonlight Bay. The defining feature of Ridges Sanctuary is a series of low, sandy ridges alternating with wet swales. These ridges run parallel to the Baileys Harbor shoreline, extending inland approximately 1 km. Each sandy ridge is stabilized by plants, including sedges and grasses near the water level, and small shrubs and trees on higher ground. The climatic effects of Lake Michigan provide cool conditions for growth of trees found more extensively in boreal forests to the north. The dominant tree species on the ridges of this site are *Picea mariana* (black spruce), *Picea glauca* (white spruce), *Abies balsamea* (balsam fir), and *Pinus strobus* (white pine), while the swales are home to swamp conifers (WIDNR, 2005). Over 500 different species of vascular plants inhabitat Ridges Sanctuary, including at least 13 endangered and threatened species and 25 of the 50 species of orchids in Wisconsin (Ridges, 2005). The Ridges Sanctuary also is home to the largest known population of the federally-endangered Hine's emerald dragonfly (WIDNR, 2005).

Our study sites were located in a coastal wetland of approximately 16 ha located near the northwest shoreline of Lake Michigan in Moonlight (Mud) Bay. A portion of this study site was underwater until Lake Michigan water levels dropped beginning in the fall of 1997.

Seagull Bar

Seagull Bar consists of a sand spit and marsh in Green Bay at the southern edge of the city of Marinette. It is owned and managed by the City of Marinette and the Wisconsin Department of Natural Resources, which designated its 49 ha portion as a State Natural Area in 1962. Seagull Bar is considered critical habitat for migratory birds because it contains the only extensive beach/dune complex on Green Bay. The endangered Piping Plover has been observed on Seagull Bar and a pair attempted nesting in 2001 (WIDNR, 2005).

Seagull Bar protects a large area of shallow water with emergent vegetation. The eastern boundary has a system of sand ridges and dunes that support rich flora of various rushes, *Ammophila breviligulata* (marram grass), and *Lathyrus japonicus* (beach pea). Invasive *P. australis* has become established following the recent lower water levels in Green Bay (WIDNR, 2005).

Our study sites were located on a coastal wetland on the northwest end of Seagull Bar, concentrated on the western side of the marsh.

Peshtigo Harbor or Peshtigo River Wetland

The Lower Peshtigo River site is a 267 ha wetland complex on a poorly drained sand lake plain along the northwest shore of Green Bay, approximately 10 kilometers southeast of the city of Peshtigo. The lower 3 km of the Peshtigo River form an extensive delta, with river channels winding through large stands of good quality emergent marsh and sedge meadow. This wetland provides large, varied and high quality habitat for waterfowl, herons, gulls and terns, and shorebirds. The site is significant due to its size, diverse wetland communities, overall good condition of the vegetation, and diversity of rare resident and migratory species known to use the Lower Peshtigo. This proposed State Natural Area, arguably the most diverse and least disturbed wetland complex on the west shore of Green Bay, is owned and managed by the Wisconsin Department of Natural Resources. (WIDNR, 2005)

Our treated study sites were located on the island in the mouth of the Peshtigo River. The sites were predominately on the south-west section of the island near the Johnson Road access. Control plots were located on the mainland at the end of Pond Road.

Study Design

We sampled vegetation at untreated controls and three treatment combinations: 1) herbicide, 2) herbicide/burning, and 3) herbicide/mowing. There was no available plant community data prior to the herbicide treatments. Five 100 m by 100 m (approximately one quarter acre) plots were selected for each treatment method at each study site. Within each plot, 10 random 1 x 2 m sub-plots were sampled (Figure 2). Controls were selected in close proximity to the sprayed areas.

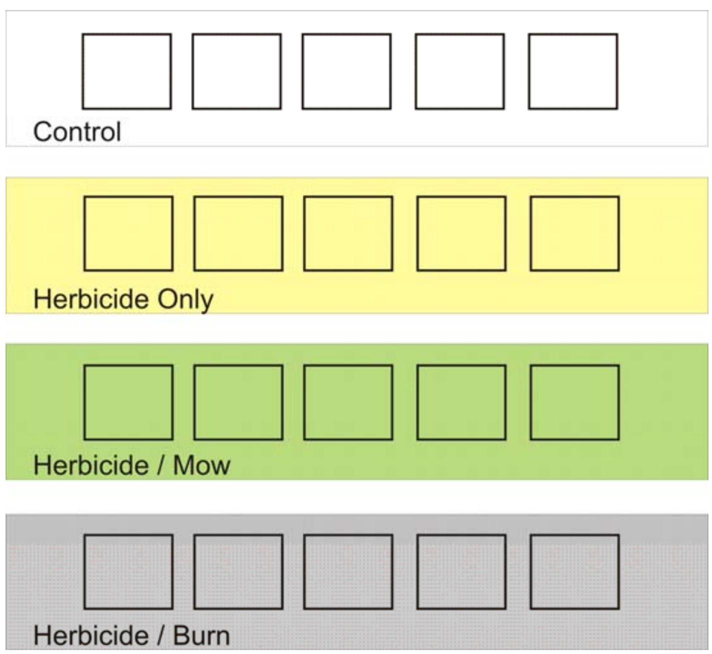

Figure 2. The study design included five plots (100 x 100 m) for each of the treatment methods and the controls. Ten random sub-plots (1 x 2 m) were sampled in each plot.

Treatments

Habitat 7 herbicide was sprayed from a helicopter on 160 ha of Long Tail Point in September 2005. The following year, 13 ha at the Ridges Sanctuary were sprayed on August 28, 2006, and 25 ha in Peshtigo Harbor and 40 ha on Seagull Bar were sprayed on September 9, 2006. The Habitat 7 concentration was 32 ounces of herbicide in 10 gallons of water per acre of spray. Sun-Wet methylated seed oil (16 ounces per gallon of water per acre) was added as a surfactant.

The secondary treatments on Long Tail Point took place during the spring of 2006. An area was mowed in March of 2006 to act as a burn break for a planned prescribed burn of the wetland. The mowing was completed with a rubber tracked skid steer with a front mounted rotary mower. This type of mower discharges the mowed material to the side and does not run over *P. australis* before the stems are cut. The prescribed burn took place April 4, 2006.

The secondary treatments at the other 3 research sites took place during the spring and summer of 2007. The Ridges site was mowed in February 2007. Peshtigo Harbor was mowed during February 2007 and was burned June 25-27, 2007. A burn break was mowed on Seagull Bar during February 2007 and the island site was burned May 14, 2007.

The mowing on Peshtigo Harbor was performed with a tractor and bush hog (John Huff – WIDNR, pers. comm.). The bush hog uses rotary blades and is towed behind a tractor. *P. australis* is driven over before being cut and the cut material remains in the path of the equipment.

An herbicide/burn treatment was not applied at The Ridges Sanctuary due to the proximity of houses. The controls at Ridges were not representative of the dense *P. australis* monotypes found at the other sites. Aerial spraying was so comprehensive at the site that there were no untreated stands of *P. australis* available for sampling. Therefore, the Ridges "control" plots are probably more representative of the communities before or during the early stages of *P. australis* invasion.

The prescribed burn at Peshtigo Harbor escaped the designated area, leaving only one herbicide/mow plot, three control plots, and four herbicide-only plots. Seagull Bar experienced an arson fire in late June 2008 that interfered with the herbicide-only site. As a result, only three herbicide-only sites were available at Seagull Bar.

In each of the treatment plots, ten 1 x 2 m sub-plots were randomly selected for field sampling from GIS maps of the treated areas (Figure 3). Sampling frequency and subplot size were determined by pilot sampling during May 2007, designed to assure that we obtained representative estimates of plant species richness. Field sampling took place August 13-31, 2007. All of the sample plots (Table 1) were georeferenced for future monitoring, and a marker was placed at one corner to identify the locality.

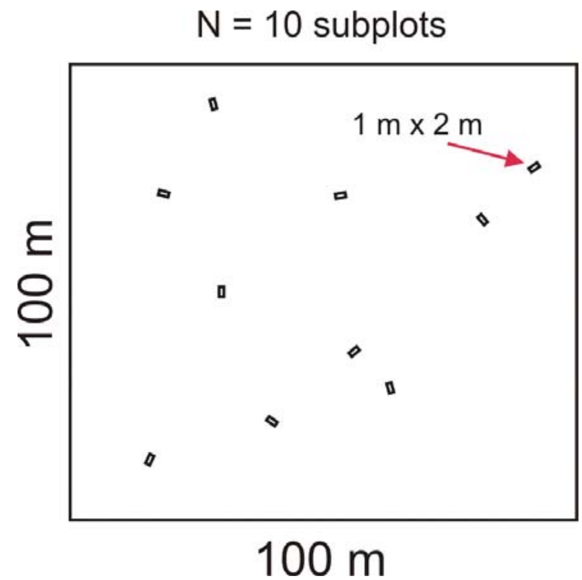

Figure 3. Each 100 meter by 100 meter plot contained 10 randomly selected 1 meter x 2 meter sub-plots. Field sampling took place in the sub-plots during August 2007.

Table 1. Summary of study sites, treatments and numbers of field samples.

Site	Treatment	Plots 100 m x 100 m	Sub-plots 1 m x 2 m
Long Tail Point	Control	5	10
	Herbicide Only	5	10
	Herbicide/Mow	2	10
	Herbicide/Burn	5	10
Ridges Sanctuary	Control	5	10
	Herbicide Only	5	10
	Herbicide/Mow	5	10
Peshtigo Harbor	Control	3	10
	Herbicide Only	4	10
	Herbicide/Mow	1	10
	Herbicide/Burn	5	10
Seagull Bar	Control	5	10
	Herbicide Only	3	10
	Herbicide/Mow	5	10
	Herbicide/Burn	5	10
Totals		63	630

The field team collected data in each of the 630 1 m x 2 m sub-plots. Field assistants recorded the percent cover of *P. australis* and all other plant species, average litter depth, and average height of *P. australis*. Plant samples for laboratory identification were collected during this phase of the field work and later identified using Voss (1996).

Plant cover was estimated at the ground level so that all species would be included in the analysis. Plants that consisted of one stem or a few stems with minimal cover were recorded as 1%. The percent cover data collection also included estimates of bare ground, litter, and dead *Phragmites* culms to account for the entire subplot area. We measured 4 randomly selected *P. australis* stems in each subplot and calculated an average height. The field team measured the depth of the leaf litter by placing a ruler in the biomass until it reached solid ground. We measured four replicates of the litter depth in each sub-plot and calculated an average for use in statistical analyses.

Data Analysis

Percent cover (arcsine square root transformed) of live *P. australis* was compared among treatment categories and sites by analysis of variance (ANOVA). Control plots were not considered in some of the analyses because the estimates of cover (= nearly 100%) are expected to yield obviously significant differences when analyzed with the treated sub-plots or plots.

We used one-way ANOVA to compare the percent cover of *P. australis* and species richness among treatments. Percent *P. australis* cover was transformed using the arcsine square root of the proportion. We performed a Tukey HSD analysis to screen for significance between the treatment methods. We performed a two-way ANOVA using treatment and site as factors for both percent *P. australis* cover and species richness.

To assess the quality of plant species present after *P. australis* treatment, we categorized species as native, introduced, or invasive using classifications from the University of Wisconsin – Madison's WISFLORA**:** Wisconsin Vascular Plant Species. Plants identified only to the genus level that could be classified confidently as native were also included in this analysis. For example, all of the *Eleocharis* species in Wisconsin are native so specimens identified as *Eleocharis* sp. were classified as native. We calculated the abundance of native species in each 1 m x 2 m subplot by summing the percent coverage of the individual native species.

We used a two-way ANOVA to compare the native species richness and the native species abundance (% cover, arcsine transformed) among treatment categories and sites. We performed a Tukey HSD analysis to screen for significance between the statistical groupings.

We evaluated the average Coefficient of Conservation and the Floristic Quality Index (Bernthal, 2003, Swink and Wilhelm, 1994) to compare the quality of plant species assemblages before and after treatments for *P. australis*. The coefficient of conservatism (C) is based on the likelihood that a species is present in a landscape relatively unaltered from pre-settlement conditions. The Floristic Quality Index (FQI) is calculated by multiplying the Mean C by the square root of the total number of species in a sample:

$$\text{FQI} = \text{Mean C} * \sqrt{\text{N}}.$$

Non-native species were assigned a coefficient of conservatism of "0" (Bernthal, 2003; Swink and Wilhelm, 1994). Both the FQI and the Mean Coefficient of Conservatism were compared among treatments and sites by analysis of variance (ANOVA). All statistical tests were performed using Systat Version 11.

RESULTS

We recorded 176 plant taxa at the four study sites. Of these, we were able to identify 139 to the species level, 25 to genus, and 2 to family. We were unable to assign any taxonomic category to 10 plants that were too immature for identification. Of the 139 that were identified to species level, 111 were native, 15 were introduced, and 13 were invasive or potentially invasive as classified by University of Wisconsin – Madison's WISFLORA**:** Wisconsin Vascular Plant Species (Table 2).

Table 2. Summary of plant species identified at each site and treatment. The percent native is the average percent of the species identified that were native at each site and treatment. The percent native cover is the average coverage of native species in each treatment category.

Site	Method	# of Species	% Native	% Native Cover
Long Tail Point	Control	81	63	21
	Herbicide only	71	66	52
	Herbicide/Mow	50	58	74
	Herbicide/Burn	88	59	66
Ridges	Control	51	56	49
	Herbicide only	59	52	22
	Herbicide/Mow	58	50	24
Peshtigo	Control	30	77	5
	Herbicide only	58	61	25
	Herbicide/Mow	39	50	34
	Herbicide/Burn	45	50	56
Seagull Bar	Control	60	64	14
	Herbicide only	42	48	22
	Herbicide/Mow	54	61	28
	Herbicide/Burn	48	53	24

Effectiveness of treatments for P. australis

Of the 45 treated plots, 40 (89%) contained live *P. australis*. Of the treated sub-plots, 236 of 452 (52%) contained live *P. australis*. All treatment methods (herbicide, herbicide/burn, herbicide/mow) significantly reduced the percent of *P. australis* compared with the control (ANOVA, $p < 0.001$). Among the three treatment methods, however, we found no significant differences in the reduced percent cover of *P. australis* (Figure 4).

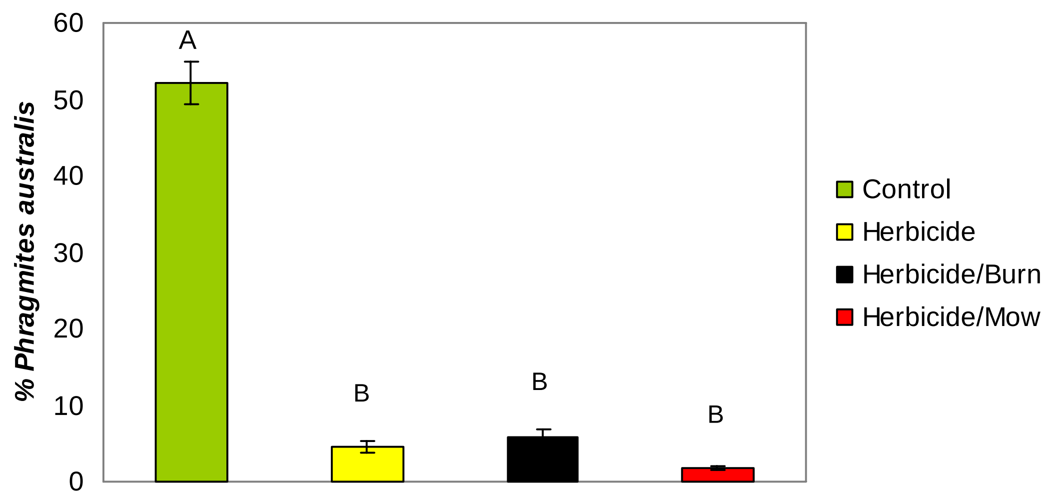

Figure 4. Effects of post-herbicide treatment on percent cover of indicated significant differences between all treatments and the control, but no significant differences between *P. australis* treatments. Letters (A and B) refer to groupings from Tukey HSD paired comparison tests.

Site and treatment showed a statistically significant interaction on the percent cover of *P. australis* (Table 3). All sites showed the same pattern of a high percent cover of *P. australis* in the control sites and significantly less *P. australis* cover in the treated sites, with the exception of Ridges Sanctuary, where *P. australis* showed significantly lower percent cover in the controls (Figure 5).

Table 3. Two-way ANOVA describing the effects of post-herbicide treatment on % *P. australis* cover (arcsine square-root transformed) at 3 study sites (Long Tail, Peshtigo, and Seagull Bar). Treatments included herbicide, herbicide/burn, herbicide/mow, and control (no treatment). Results from The Ridges Sanctuary were excluded. Multiple $R^2 = 0.81$.

Source	df	Mean-square	F-Ratio	P
Treatment	3	25.56	604.5	0. 00
Site	2	0.07	1.6	0.20
Treatment x Site	6	0.74	17.4	0.00
Error	468	0.04		

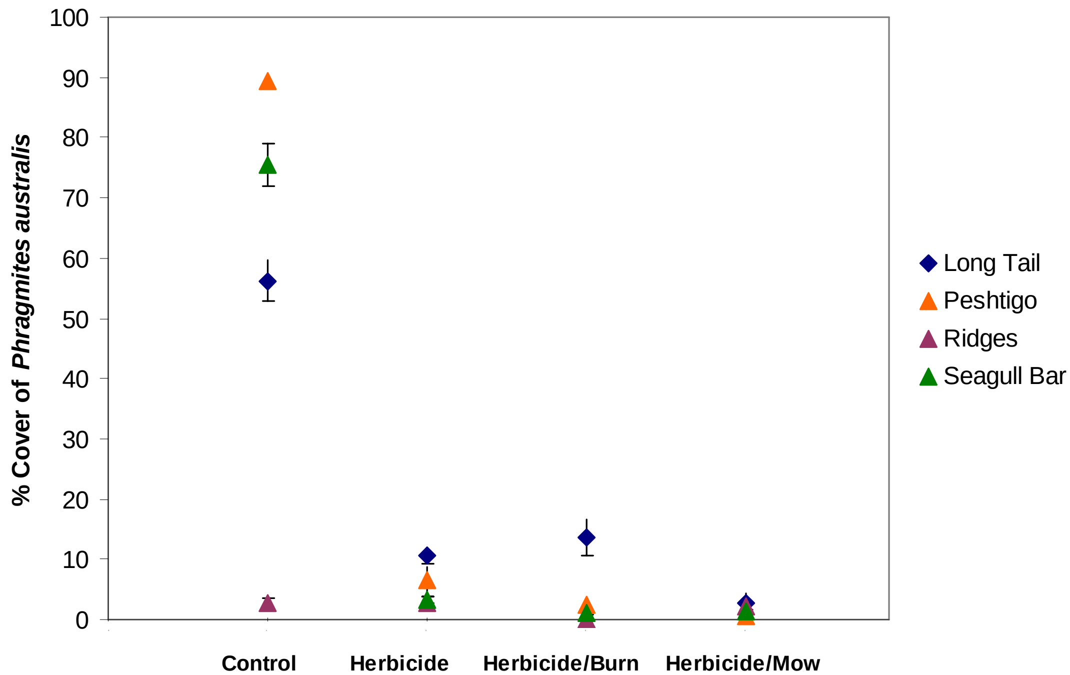

Figure 5. Effects of post-herbicide treatment on percent cover of *P. australis*. Symbols represent mean values by site and treatment; error bars show +/- standard error.

Species Richness Analysis

Both treatment method and site significantly affected average species richness among sub-plots (ANOVA, $p < 0.001$). The herbicide/mow treatment had significantly higher average species richness than the other treatments and the control, while the herbicide/burn and herbicide only treatments were not significantly different and the control and herbicide/burn treatments were not significantly different (Figure 6).

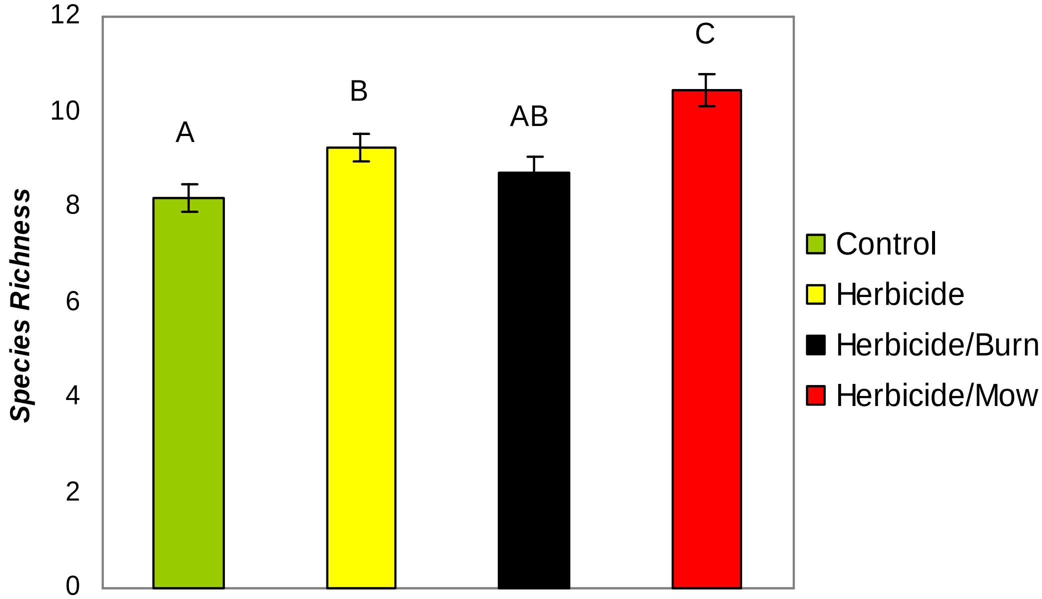

Figure 6. Effects of post-herbicide treatment on average species richness by treatment indicated significant differences between the herbicide/mow treatment and all other treatments and the control. The herbicide/burn and herbicide only treatments were not significantly different and the control and herbicide/burn treatments were not significantly different. Letters (A,B,AB, and C) refer to groupings from Tukey HSD paired comparison tests.

Site effects on average species richness were highly significant (Table 4). Long Tail Point yielded significantly higher average species richness than the other treatments and the control (Figure 7).

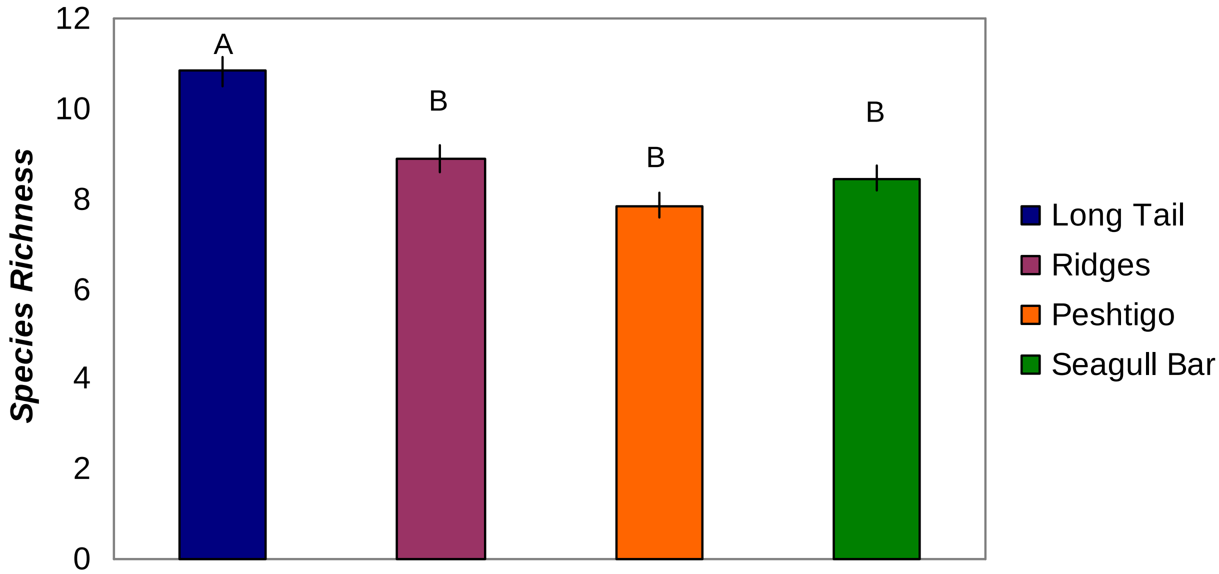

Figure 7. Effects of post-herbicide treatment on average species richness by site indicated that Long Tail had significantly higher average species richness than the other sites. There were no significant differences between Ridges, Peshtigo and Seagull Bar. Letters (A and B) refer to groupings from Tukey HSD paired comparison tests.

The interaction between site and treatment again showed a statistically significant effect on average species richness (Table 4). All of the sites except Long Tail Point showed the same pattern, where the highest average species richness was observed in the herbicide/mow treatments. The herbicide/burn treatment at Long Tail Point yielded significantly higher average species richness (Figure 8).

Table 4. Two-way ANOVA describing the effects of post-herbicide treatment on average species richness among sub-plots at 3 study sites (Long Tail, Peshtigo, and Seagull Bar). Treatments included herbicide, herbicide/burn, herbicide/mow, and control (no treatment). The results from Ridges were excluded. Multiple $R^2 = 0.81$.

Source	df	Mean-square	F-Ratio	P
Treatment	3	75.566	5.708	0. 001
Site	2	241.826	18.266	0.000
Treatment x Site	6	55.658	4.204	0.000
Error	468	13.239		

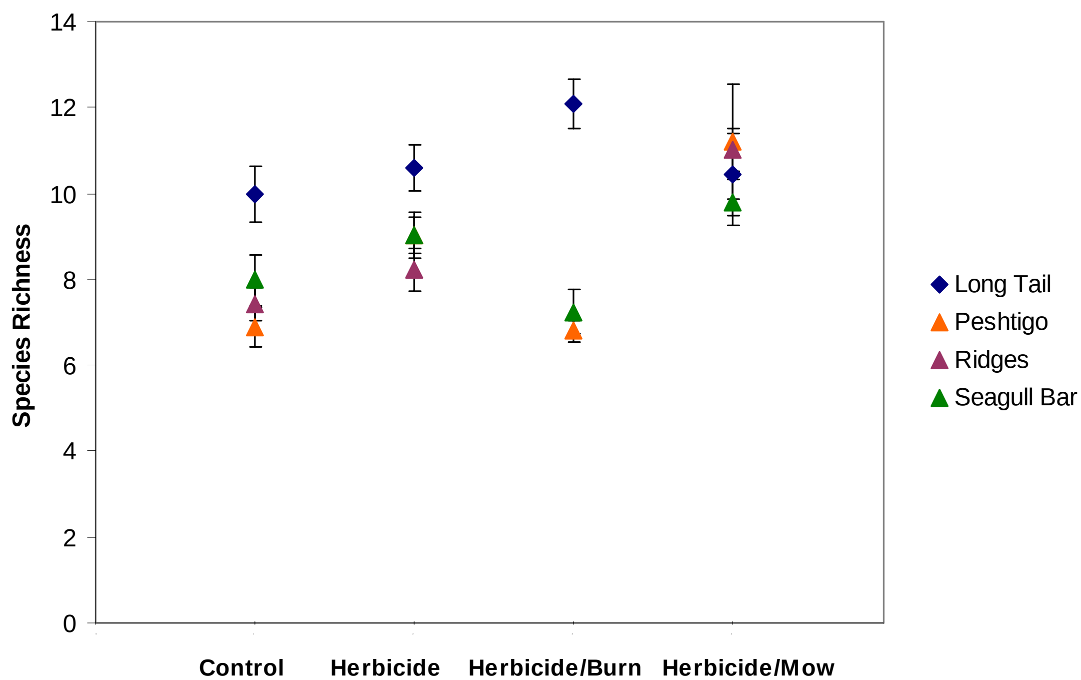

Figure 8. Effects of post-herbicide treatment on average species richness among sub-plots. Symbols represent mean values by site and treatment; error bars show +/- standard error.

Treatment method did not significantly affect cumulative species richness among sub-plots (Figure 9). Site effects on cumulative species richness were highly statistically significant (ANOVA, $p < 0.001$). Long Tail had significantly higher cumulative species richness than Peshtigo and Seagull Bar (Figure 10). Ridges Sanctuary was excluded from this analysis.

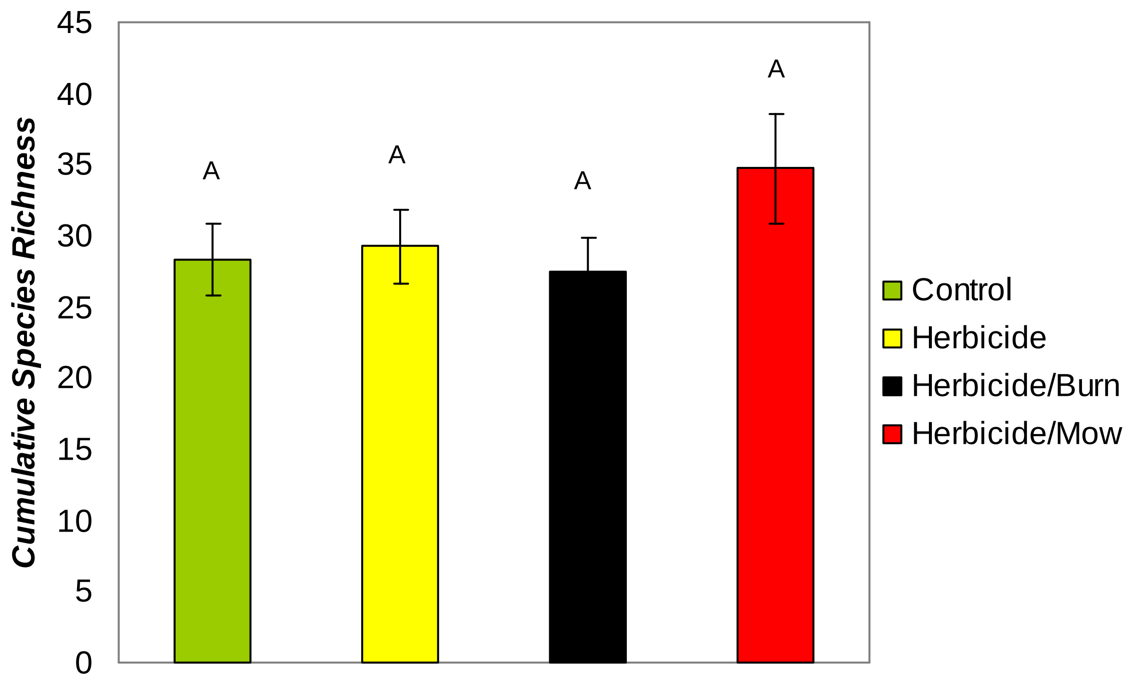

Figure 9. Effects of post-herbicide treatment on cumulative species richness by treatment indicated no significant differences between the treatments of Control, Herbicide Only, Herbicide/Burn and Herbicide/Mow. Letter (A) refer to the grouping from Tukey HSD paired comparison tests.

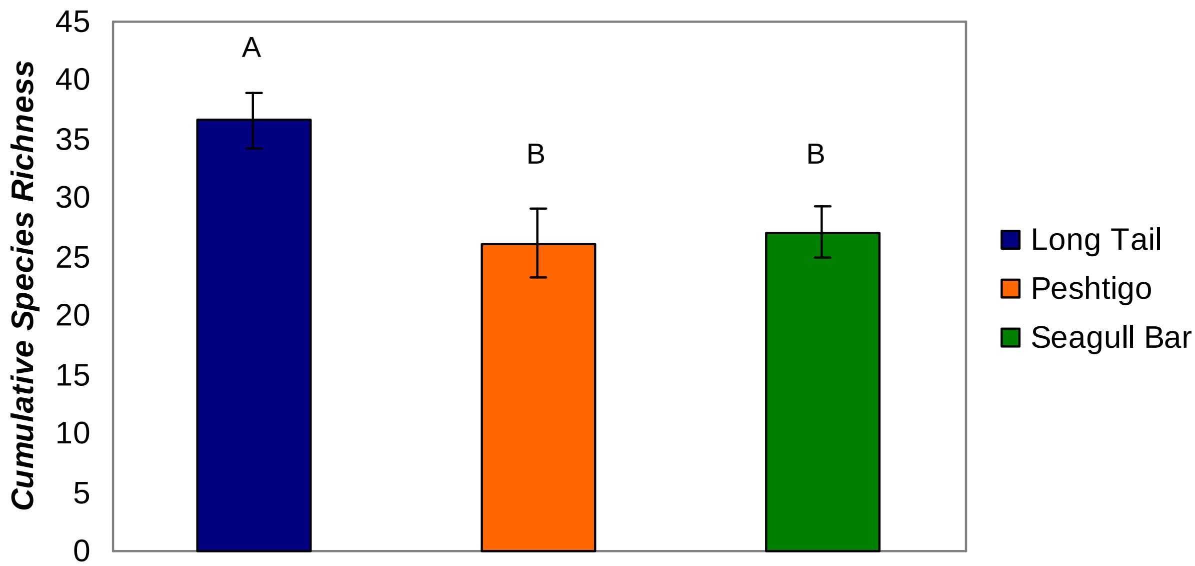

Figure 10. Effects of post-herbicide treatment on cumulative species richness by site indicated that Long Tail had significantly higher species richness than the other sites. There were no significant differences between Peshtigo and Seagull Bar. Letters (A and B) refer to groupings from Tukey HSD paired comparison tests.

If we compare average and cumulative species richness from the 10 subplots in each 100 m x 100 m plot rather than for each subplot individually, treatments (control, herbicide only, herbicide/burn, and herbicide/mow) show much weaker effects (Tables 5 and 6). Treatment effects on average species richness (among plots rather than sub-plots) are not statistically significant due to smaller sample size and reduced influence of sites with the maximum of 50 subplots. Site effects, on the other hand, are still are significant. Treatment effects on cumulative species richness (total number of species in all 10 subplots combined) also were not statistically significant, suggesting that at least some of the spatial variation in species composition was retained even in the untreated *P. australis* (control) plots. In other words, the species recorded in untreated stands of *P. australis* were not simply the same tolerant species from plot to plot.

Table 5. Two-way ANOVA describing the effects of post-herbicide treatment on average species richness among 100 m x 100 m plots at 3 study sites (Long Tail, Peshtigo, and Seagull Bar). Species richness values for each plot were averaged from 10 sub-plots. The results from Ridges were excluded. Multiple R^2 = 0.36.

Source	df	Mean-square	F-Ratio	P
Treatment	3	7.61	1.16	0.339
Site	2	23.87	3.64	0.036
Treatment x Site	6	5.54	0.84	0.544
Error	36	6.57		

Table 6. Two-way ANOVA describing the effects of post-herbicide treatment on cumulative species richness among 100 m x 100 m plots at 3 study sites (Long Tail, Peshtigo, and Seagull Bar). Species richness values for each plot describe the total number of species in 10 sub-plots combined. The results from Ridges were excluded. Multiple R^2 = 0.45.

Source	df	Mean-square	F-Ratio	P
Treatment	3	70.01	0.87	0.465
Site	2	457.81	5.69	0.007
Treatment x Site	6	122.06	1.52	0.200
Error	36	80.40		

Quality of plant species assemblages

The effect of treatment methods on percent cover of native species was highly statistically significant (ANOVA, $p < 0.0001$). The herbicide/burn treatment had significantly higher percent native species cover than the treatments of herbicide/mow, herbicide only and the control. The herbicide/mow and herbicide only treatments also had significantly higher native percent cover than the control (Figure 11).

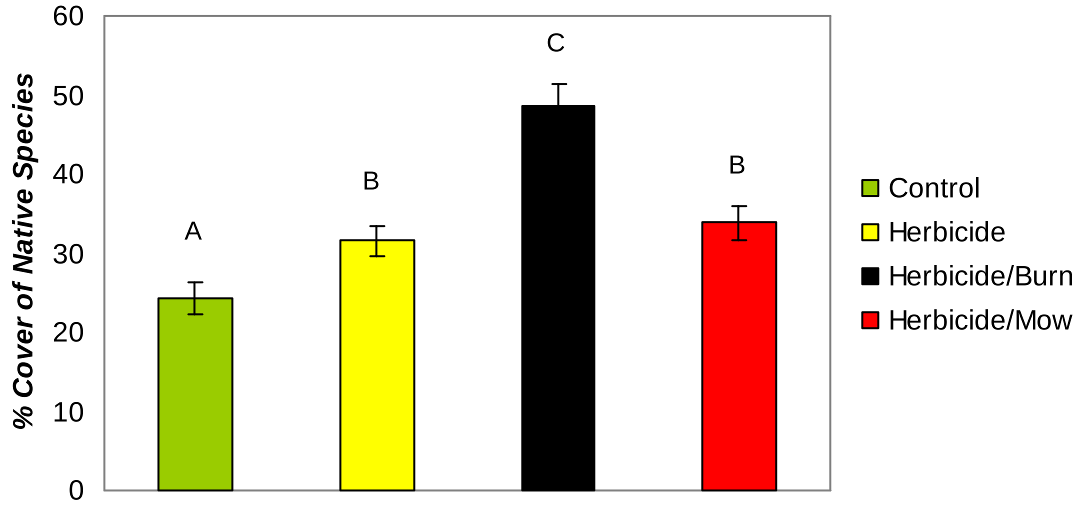

Figure 11. Effects of post-herbicide treatment on mean % cover of native species by treatment indicated significant differences between the herbicide/burn treatment and all other treatments and the control. The herbicide/mow and herbicide only treatments were not significantly different and the control and herbicide/burn treatments were not significantly different. Letters (A, B and C) refer to groupings from Tukey HSD paired comparison tests.

Site effects on percent cover of native species were highly statistically significant (ANOVA, $p < 0.001$). Long Tail had significantly higher native percent cover than the other sites. Peshtigo and Ridges had significantly higher native percent cover than Seagull Bar (Figure 12).

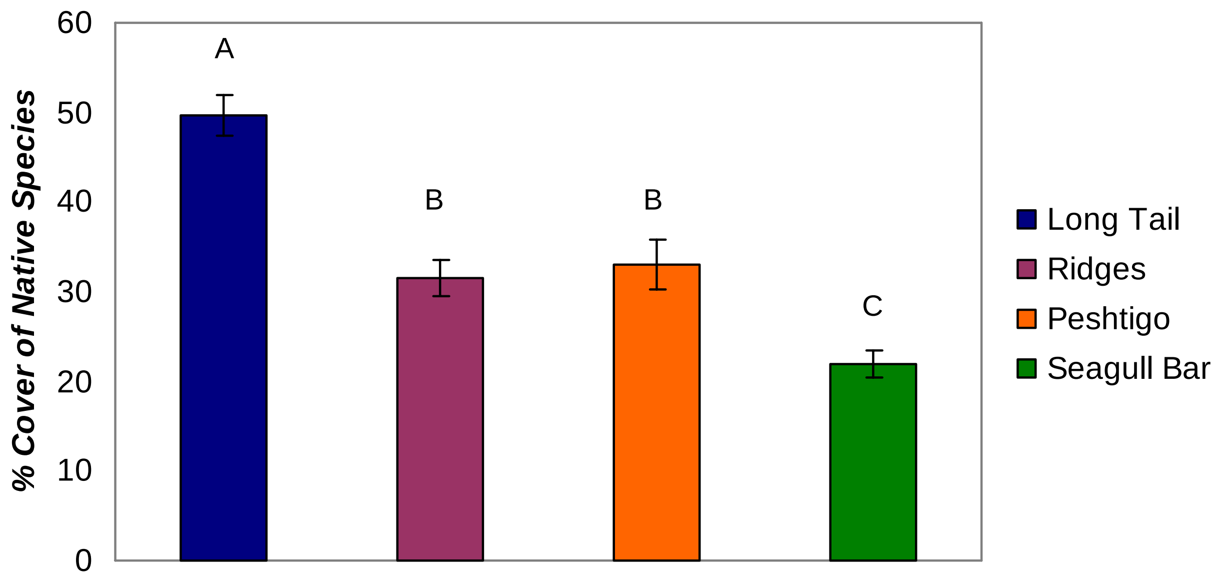

Figure 12. Effects of post-herbicide treatment on mean % cover of native species in sub-plots. Long Tail had significantly higher native mean % cover than the other sites. Ridges and Peshtigo also showed a significantly higher native mean % cover than Seagull Bar. . Letters (A, B and C) refer to groupings from Tukey HSD paired comparison tests.

The interaction effect between site and treatment on mean percent cover of native species was highly statistically significant (Table 5). All treatments showed the same patterns with the exception of the Peshtigo herbicide/burn sites. Long Tail Point had consistently higher mean percent cover of native species in all treatments and Seagull Bar had consistently lower in all treatments. At Peshtigo, the herbicide/burn plots had significantly higher mean percent cover of native species than the other treatments (Figure 13).

Table 7. Two way ANOVA describing the effects of post-herbicide treatment on percent cover of native species at 3 study sites (Long Tail, Peshtigo, and Seagull Bar). Treatments included herbicide, herbicide/burn, herbicide/mow, and control (no treatment). The results from Ridges were excluded. Multiple $R^2 = 0.470$.

Source	df	Mean-square	F-Ratio	P
Site	2	39799.103	86.205	0. 000
Treatment	3	30059.646	65.109	0.000
Site x Treatment	6	5556.783	12.036	0.000
Error	468	461.682		

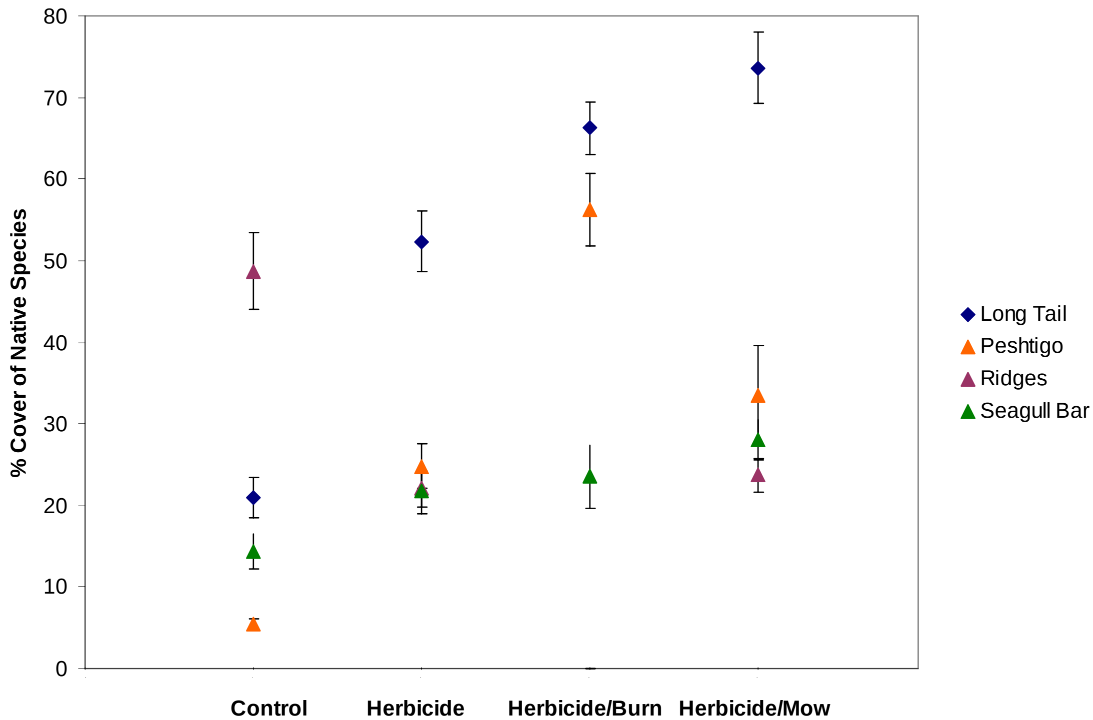

Figure 13. Effects of post-herbicide treatment on % cover of native plant species. Symbols represent mean values by site and treatment; error bars show +/- standard error.

Mean Coefficients of Conservation (Wisconsin Floristic Quality Assessment 2008) did not differ significantly among treatments (ANOVA, p = 0.753) or sites

(ANOVA, p = 0.638). Floristic Quality Index also did not differ significantly among treatments (ANOVA, p = 0.846) or sites (ANOVA, p = 0.214). Nevertheless, the assessment did indicate the presence of some highly desirable plants in the wetland study sites. We found 15 species with coefficient of conservation values (Bernthal, 2003) of 7 or larger (Table 6).

Table 8. Summary of plants with coefficient of conservation values of 7 or above.

Campanula aparinoides (7)	*Carex stricta* (7)
Cicuta bulbifera (7)	*Cirsium muticum* (8)
Eleocharis flavescens (8)*	*Epilobium leptophyllum* (8)
Equisetum variegatum (7)	*Liparis loeselii* (7)
Lobelia kalmii (9)	*Lysimachia thyrsiflora* (7)
Parnassia glauca (8)	*Picea glauca* (7)
Pilea fontana (7)	*Thuja occidentalis* (9)
Zizania palustris (8)	

*species of concern in Wisconsin

In an effort to understand whether the herbicide treatment eradicated any species locally, we have listed 29 plant species that were observed in the control plots but were not present in the treatment plots at the same site (Table 7). *Monarda fistulosa*, *Mentha arvensis*, *Convolvulus arvensis* were present in the control plots of multiple sites but were entirely absent from the corresponding treatment plots.

Table 9. Summary of plants that were present in control (no treatment) plots but were absent in treatment plots at the same site.

Plant Species	Long Tail	Ridges	Peshtigo	Seagull Bar
Agalinis tenuifolia				x
Agrostis gigantea				x
Agrostis stolonifera	x			
Anemone sp.		x		
Argentina anserina				x
Bolboschoenus fluviatilis		x		
Carex lacustris			x	
Convolvulus arvensis	x	x		
Cornus stolonifera				x
Epilobium angustifolium				x
Epilobium leptophyllum				x
Galium trifidum				x
Glechoma hederacea	x			
Laportea canadensis	x			
Lysimachia ciliata	x			
Lythrum salicaria		x		
Mentha arvensis		x		x
Monarda fistulosa	x	x		
Penthorum sedoides			x	
Poa palustris				x
Potentilla norvegica		x		
Ranunculus pensylvanicus	x			
Ranunculus sp.	x			
Rumex crispus			x	
Salix purpurea		x		
Salix sp.		x		
Solanum dulcamara	x			
Sonchus arvensis		x		
Sparganium eurycarpum				x

P. australis height

All treatment methods (herbicide, herbicide/burn, herbicide/mow) significantly reduced the average height of *P. australis* compared to the control plots (ANOVA, $p < 0.001$). Among the three treatment methods, the herbicide/mow plots had significantly shorter *P. australis* stems than at the herbicide only and herbicide/burn plots (Figure 14).

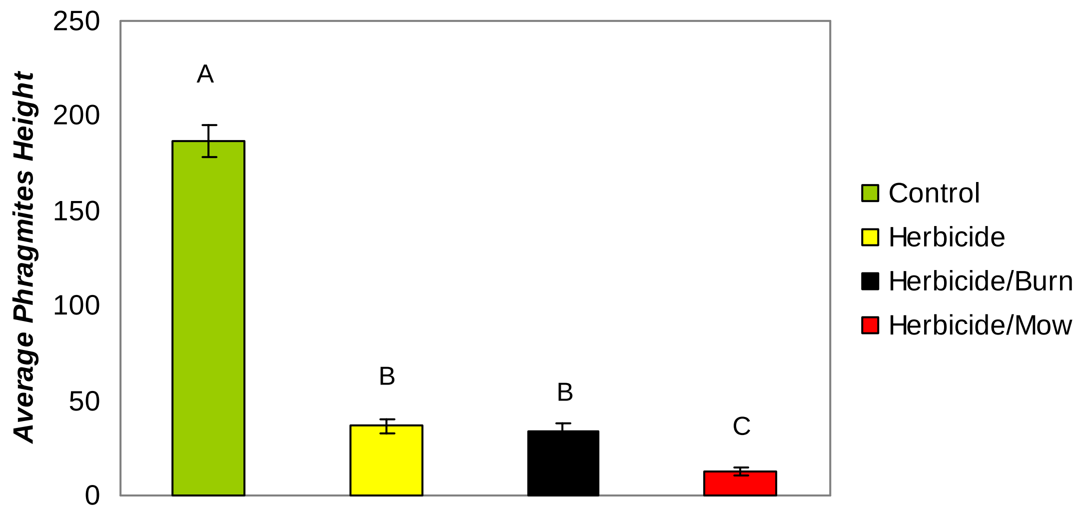

Figure 14. Effects of post-herbicide treatment on average height of *P. australis* by treatment indicated significant differences between the controls and all treatments. The herbicide/mow treatment resulted in statistically significantly shorter of *P. australis* than the herbicide and herbicide/Burn treatments. Letters (A, B and C) refer to groupings from Tukey HSD paired comparison tests.

The height of *P. australis* stems was significantly shorter at Ridges (19.6 cm ± 2.4 cm and taller at Long Tail (112.3 cm ±8.6 cm) compared to Peshtigo (81.3 cm ±8.7) and Seagull Bar (77.7 cm ±7.9 cm) (ANOVA, $p < 0.001$, Tukey HSD).

DISCUSSION

Because a high percentage of plots contained *P. australis* after treatment (89%), we can easily see that none of the treatments tested completely eradicated *P. australis*. These findings are consistent with previous conclusions that no known treatment will eradicate *P. australis* in one application (Derr, 2008; Teal and Peterson, 2005; Sun et al., 2006; Silliman and Bertness, 2004; Turner and Warren, 2003; Buchsbaum et al., 2006; Warren et al., 2001). Turner and Warren (2003) reasoned that the percent cover of *P. australis* eventually will increase if the spray program is reduced or stopped. However, we found that all three of the treatments did significantly reduce the occurrences of *P. australis* in the plots, showing that the treatment methods can be effective in controlling *P. australis*, at least in the short term. Reduction in the number of *P. australis* stems is different from simply reducing the height of *P. australis* plants because of mowing, for example. Our findings are consistent with research conducted by Teal and Peterson (2005), who suggested that herbicide treatment killed the above ground biomass and a significant number of rhizomes. Herbicide treatment, however, did not kill all of the rhizomes. We found the same pattern of *P. australis* reduction at all sites except for the Ridges Sanctuary, where *P. australis* was sparse even in the control plots. The low percent cover of *P. australis* in the control plots at the Ridges can be explained by the difficulty in locating true control sites. Past control efforts at the Ridges have been so successful that few untreated stands of *P. australis* were available. The Ridges control plots are probably more representative of the communities before or during the early stages of the *P. australis* invasion.

Other studies have shown that invasion of wetlands by *P. australis* results in a three-fold (Silliman and Bertness, 2004) to five-fold (Bertness et al., 2002) decrease in species richness and a significant reduction (up to 94%) in the abundance of many native wetland species (Silliman and Bertness, 2004). Our analysis of average species richness and the percent cover of native species revealed significant differences in species richness between the treatment options. The herbicide/mow treatment showed significantly higher species richness than any of the other treatments, the only exception occurring at Long Tail Point, where the herbicide/burn treatment had significantly higher species richness. All treatments significantly improved the percent cover of native species, with the herbicide/burn treatment showing the strongest improvement. Similar findings were reported by Meyerson et al. (2000) where marsh plants recolonized freshwater systems following *P. australis* control treatments and in all cases species diversity increased two fold over *P. australis* dominated systems. Warren et al. (2001) found that a brackish meadow could be restored in two to three years if herbicide treatment was combined with mowing.

Long Tail Point differed from the other study sites in the timing of the aerial herbicide and secondary treatments. Long Tail Point was treated one year before the other sites so it had an extra growing season for recovery of native wetland species. The timing of the burn at Peshtigo Harbor also complicates the interpretation of our results. The burn took place late in the season, leaving only 1-2 months between the burn and the field sampling. The late burn could have negatively affected species richness due to the short growing time after the burn. From this analysis we cannot determine whether herbicide/mow or herbicide/burn is most effective in facilitating the return of native species. The significant positive effects of herbicide/mow on average species richness and the significant positive effects of herbicide/burn on percent cover of native species clearly imply that a secondary treatment is preferred over herbicide treatments alone. Similarly, Turner and Warren (2003) suggested that a combination of herbicide application and mowing or burning would continue to be a preferred approach to controlling *P. australis*.

Considering only the results of the two-year post-treatment sampling at Long Tail Point, herbicide/burn was the most promising treatment for both species richness and percent cover of native species. Sun et al. (2007) also found that burning in combination with tidal enhancements was the most promising treatment for reduction in *P. australis* density and height at a New Jersey saltwater wetland. Future research should be conducted to understand if the herbicide/burn treatment has a more positive impact 2-3 years after treatment when compared to herbicide/mow.

While the control plots at all of the sites had higher average species richness than expected, the percent cover of native species was alarmingly low in *P. australis*-dominated plots, and the plants that were present often were etiolated. The cumulative species richness analysis reinforced that the number of species present in control plots is not significantly different that the treated plots. Invasion of *P. australis* dramatically reduced the percent cover of native species, but did not dramatically reduce the species richness. Restoration potential exists in the invaded community through the presence of native species surviving in the understory, but the etiolated condition of the surviving species indicate that time is critical to restoration success. Meyerson et al. (2000) came to similar conclusions based on observations at sites dominated by *P. australis*. They found relatively high plant diversity in *P. australis* stands, but many of the populations were sterile and non-viable. Warren et al. (2001) found that vestiges of the pre-invasion community may persist as understory for a decade or more.

The high percent cover of native species at Peshtigo's herbicide/burn plots is influenced largely by the presence of *Calamagrostis canadensis*. Sub-plots showed up to 97% cover of *C. canadensis*, the most extensive native species overall in this treatment. *C. canadensis* is a rhizomatous plant that could have survived the prescribed burn as underground vegetative ramets, which would subsequently provide rootstock for regeneration (Voss, 1996). Warren et al. (2001) found similar results on the Connecticut

River with rhizomatous grasses in the genus *Agrostis stolonifera* after herbicide/mow treatments. They concluded that *Agrostis stolonifera* was probably more important to the brackish meadow community after invasion of *P. australis* because of its ability to recolonize quickly after treatments. The importance of *Agrostis* decreased over time as other plants recolonized, but it was critical to the initial re-establishment of the community.

While the Wisconsin Floristic Quality Assessment showed no significant differences among the treatment methods, the presence of high quality plants in the study sites (Table 6) is a strong indication that healthy wetlands can be restored at sites dominated by *P. australis*. Time will be needed to assess whether this restoration potential will last for more than a few years. Of the 17 species with high coefficients of conservation, 9 were present at Ridges, 6 were present at Long Tail Point, 4 were present at Seagull Bar, and 2 were present at Peshtigo Harbor. One Wisconsin species of concern, *Eleocharis flavescens*, was observed during this study. Other uncommon or rare species include *Cicuta bulbifera*, which was found at all study sites, and *Campanula aparinoides* and *Pilea Fontana*, which were found at three study sites. *Cicuta bulbifera* and *Pilea fontana* were the only recognized ecologically sensitive species found widely in our study areas, suggesting that other sensitive wetland plant species might have already been lost due to small population sizes. Our results suggest that some of the identified plant species could be especially sensitive to the Habitat 7 herbicide (Table 7). We recorded 29 species in control plots that were not present in treatment plots. This study did not directly address the question of risks associated with herbicide use, but future studies should address the question. A study comparing *P. australis* marshes that were treated annually *vs.* marshes where treatment was intermittent concluded that annual herbicide treatment appears to sustain higher quality habitat and with less herbicide overall than occasional applications when *P. australis* cover is at its maximum (Turner and Warren, 2003).

The cost for aerial spraying in northeastern Wisconsin varied depending of the acreage being treated. The cost of the helicopter and herbicide ranged from $109 per acre in the Green Bay area, where 400 acres were sprayed in 2006, to $175 per acre at the Ridges, where 32 acres were sprayed in 2007. According to the Wisconsin Department of Natural Resources, the estimated cost of the mowing secondary treatment ranged from $70-100 per acre (Mark Martin and Gary VanVreede personal communication). The estimated cost for a prescribed burn is $50-100 per acre depending on the size. Most contractors have a minimum charge (e.g., $750, pers. comm., Wade Oehmichen – Wisconsin DNR Biologist). As with the aerial spraying, the costs of mowing and burning per acre decrease as the treatment area is expanded. From a management perspective, herbicide only is the most cost-effective of the treatments tested in this study. The secondary treatments of herbicide/mow or herbicide/burn add costs, but this study indicates that the extra costs clearly lead to more rapid and perhaps more complete wetland restoration. A secondary treatment of burning or mowing to complement herbicide application should be the standard treatment for *P. australis*. The distinction between herbicide/mow and herbicide/burn is not clear from this study. In many cases, the decision between herbicide/mow and herbicide/burn is dictated by the location of the area to be treated and local regulations. Because of high treatment costs, prioritization of critical habitat must take place for the most effective use of limited funding.

As with most experimental designs, this study encountered limitations due to the realities of field research. Site effects were prominent, timing of treatments seemed to have significant effect on the outcome, and the treatments were complicated by factors like unintended spread of fire. Hence, this study cannot provide unambiguous recommendations about *Phragmites* control, but the results are especially meaningful when combined with results from other field studies. We have shown that local eradication of the invasive species *P. australis* is difficult, if not impossible, in Wisconsin with one set of treatments, consistent with the conclusions of Derr (2008), Teal and Peterson (2005), Sun et al. (2006), Silliman and Bertness (2004), Turner and Warren (2003), Buchsbaum et al. (2006), and Warren et al. (2001). Treatment of *P. australis*, however, is the first step in restoring native wetlands to pre-invasion condition. Treated plots were inhabited by native species at a much greater percent cover of native wetland species compared to plots without any treatments. These findings show that certain native plant species are present and can rather quickly re-populate treated wetlands, supporting the research of Warren et al. (2001) and Meyerson et al. (2000). This study and that of Meyerson et al. (2000) have documented the presence of native species in untreated control plots, but the low species abundance and poor condition of plants in these plots suggests that time might be critical to the treatment of *Phragmites* stands. There is a risk that native wetland plants will not persist for many years after *P. australis* invasion. Our study, along with Turner and Warren (2003), supports the application of a secondary treatment of mowing or burning in addition to herbicide to increase the success of the return of native species. Because *P. australis* can out-compete native wetland plants, long-term maintenance likely will be required to sustain native wetland vegetation in the Great Lakes coastal zone. Aerial spraying is an effective first step in the control of *P. australis*, but follow-up treatments are necessary for the long term.

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treatments and the control. The herbicide/mow and herbicide only treatments were not significantly different and the control and herbicide/burn treatments were not significantly different. Letters (A, B and C) refer to groupings from Tukey HSD paired comparison tests.

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Figure 13. Effects of post-herbicide treatment on % cover of native plant species. Symbols represent mean values by site and treatment; error bars show +/- standard error.

Figure 14. Effects of post-herbicide treatment on average height of *P. australis* by treatment indicated significant differences between the controls and all treatments. The herbicide/mow treatment resulted in statistically significantly shorter of *P. australis* than the herbicide and herbicide/Burn treatments. Letters (A, B and C) refer to groupings from Tukey HSD paired comparison tests.

APPENDIX A

Table 8. Summary of plants sampled by site and treatment with their corresponding classification and Floristic Quality Assessment coefficient of conservation value. C = control, H = herbicide, B = herbicide/burn, M = herbicide/mow.

Plant Name	Native/ Introduced/ Invasive	Coefficient of Conservation	Long Tail Point	Ridges Sanctuary	Peshtigo Harbor	Seagull Bar
Acer negundo	NAT	0		M	H	
Agalinis tenuifolia	NAT	6	H			C
Agrostis hyemalis	NAT	4				C,M,B
Agrostis gigantea	INT	0	C,H,B	C,H,M	C,M	C
Agrostis stolonifera	INT	0	C			
Alisma subcordatum	NAT	3			C,H	
Amaranthus tuberculatus	NAT	3	H,M,B		H	B
Ambrosia artemisiifolia	NAT	0	H,M,B			
Anemone sp.		0		C		
Argentina anserina	NAT	4		C,H,M		C
Asclepias incarnata	NAT	5	B		C,H,M,B	C,H,M,B
Aster lateriflorus	NAT	3	H		C	
Aster pilosus	NAT	1		C,H,M		
Aster puniceus	NAT	5	H	C		
Aster sp.	NAT	0	C,H,B	C,H,M	M	H,M
Aster umbellatus	NAT	6	H,M,B	H,M		C,H,M,B
Aster lanceolatus	NAT	4	H			
Betula papyrifera	NAT	3		H,M	H	M
Bidens cernuus	NAT	4	H,M,B	H,M	C,H,M	H,M,B
Bidens comosus	NAT	5	H,M		M	
Bidens frondosus	NAT	1	H,M		H	
Bidens sp.	NAT	0	H,B	M	H,M	C,M,B
Boehmeria cylindrical	NAT	6	C,B			
Bolboschoenus fluviatilis	NAT	6	C,H,M,B	C	C,H,B	M
Brassicaceae		0			B	
Calamagrostis canadensis	NAT	5	C,H,M,B	C,H,M	C,H,M,B	C,H
Campanula aparinoides	NAT	7	H	C,H,M		M,B
Carex bebbii	NAT	4	C,H,M,B		H,B	C,B
Carex flava	NAT	6		C		
Carex hystericina	NAT	3	C,H			C,M
Carex lacustris	NAT	6	C		C	
Carex pellita	NAT	4	B		M	C
Carex sp.	NAT	0	C,H,B	C,H,M	C,H,M,B	C,H,M,B
Carex stricta	NAT	7		C		
Carex viridula	NAT	6				C,M,B
Carex vulpinoidea	NAT	2	C,H,B			

Cicuta bulbifera	NAT	7	C,H,M,B	C,M	C,H,M,B	C,H,M,B
Cirsium arvense	INV	0	C,M,B	C,H,M	M,B	C,H,M,B
Cirsium muticum	NAT	8	B			
Cirsium palustre	INV	0	B			M
Cirsium sp.		0	C,H,M,B	H,M	M,B	H,M,B
Cirsium vulgare	INV	0	H,B	H,M		
Convolvulus arvensis	INV	0	C	C	B	
Conyza canadensis	NAT	0	M,B	H,M	M	H,M
Cornus stolonifera	NAT	3	C,B			C
Cyperaceae	NAT	0	H,M			
Cyperus odoratus	NAT	4	B		H	
Cyperus bipartitus	NAT	3	C,M,B			H,M,B
Echinochloa crus-galli	INT	0	B			C,H,M,B
Echinocystis lobata	NAT	2	C,B			
Eleocharis sp.	NAT	0	C,H,B	H,M	H,M,B	C,H,M,B
Eleocharis erythropoda	NAT	3	C,H,M,B	C	C,H,M,B	C,M
Eleocharis flavescens	NAT	8				M,B
Epilobium angustifolium	NAT	3	H			C
Epilobium ciliatum	NAT	3	C,H,B			
Epilobium leptophyllum	NAT	8	C,H,B			C
Equisetum arvense	NAT	1		C,H		
Equisetum variegatum	NAT	7		M		
Erechtites hieracifolia	NAT	2	B		H	C,H,M,B
Erigeron annuus	NAT	0	M,B			
Eupatorium maculatum	NAT	4	C,H,M,B	C,H,M	B	H,M
Eupatorium perfoliatum	NAT	6	C,H,M,B	C,H,M	C,H,M,B	C,H,M,B
Euthamia graminifolia	NAT	4	C,H,M,B	C,H,M	C,H,M	C,H,M,B
Fragaria virginiana	NAT	1	C			
Fraxinus sp.	NAT	0	C,M,B	H		
Galium trifidum	NAT	6	C,H,M,B	C,M	C,H,M,B	C
Geum canadense	NAT	2				M
Glechoma hederacea	INV	0	C			
Hieracium caespitosum	INT	0		H,M		H,M
Hieracium sp.	INT	0		C,H,M		
Hypericum majus	NAT	5				M
Hypericum perforatum	INV	0			H	C,H,M,B
Hypericum sp.	INV	0		H,M		
Impatiens capensis	NAT	2	C,H,M,B	C,H	H,B	C,M
Juncus arcticus	NAT	5	C	C,H,M	H,M,B	C,H,M,B
Juncus dudleyi	NAT	4	B			
Juncus effusus	NAT	4	C		C	M
Juncus nodosus	NAT	6	B	H		C
Juncus sp. (1)	NAT	0	C,H,B	C,H,M	H,M,B	C,H,M,B
Juncus sp. (2)	NAT	0	B	H,M		
Lactuca sp.		0		H	H,M	M,B

Laportea canadensis	NAT	4	C			
Leersia oryzoides	NAT	3	C,H,M,B	H,M	C,H,M	C,H,M,B
Lepidium campestre	INV	0	H			
Lindernia dubia	NAT	6			H	
Liparis loeselii	NAT	7	C,H			
Lobelia sp.	NAT	0				B
Lobelia kalmii	NAT	9		M		
Lobelia siphilitica	NAT	5	H			
Lobelia spicata	NAT	6				B
Lycopus sp.	NAT	0	C,B	C,H,M	H	C,H,M,B
Lycopus americanus	NAT	4	C,H,B	C,H,M	H,M,B	C,H,M,B
Lycopus uniflorus	NAT	4	C,H	C		
Lysimachia ciliata	NAT	5	C			
Lysimachia thyrsiflora	NAT	7		H		
Lythrum salicaria	INV	0	C,H,M,B	C	C,H,M,B	C,H,B
Mentha arvensis	NAT	3		C		C
Mimulus ringens	NAT	6	C,M			
Monarda fistulosa	NAT	3	C	C		
Oenothera biennis	NAT	1	H,B			
Onoclea sensibilis	NAT	5	H			
Panicum sp.		0	B			
Panicum capillare	NAT	1	C,M,B		H,M	H,M,B
Panicum dichotomiflorum	NAT	0	B			B
Parnassia glauca	NAT	8		H,M		
Penthorum sedoides	NAT	3	H		C	
Phalaris arundinacea	INV	0	C,M,B	C,H,M	H,M,B	C,M,B
Phragmites australis	INV	0	C,H,M,B	C,H,M	C,H,M,B	C,H,M,B
Picea glauca	NAT	7	H	M		
Pilea fontana	NAT	7	C,H,M,B	H		H,M,B
Plantago major	INT	0	H,B			
Poa compressa	INT	0		M		
Poa palustris	NAT	5				C
Poa sp.		0	M			
Polygonum sp.		0	C,H,M,B		H	C,H,M
Polygonum amphibium	NAT	5	C	C,H	B	C
Polygonum hydropiper	INT	6	H,B			
Polygonum lapathifolium	NAT	2	M,B		H,B	
Polygonum persicaria	INT	0	H			H
Polygonum punctatum	NAT	5	C,H,M,B		C,H,B	C
Polygonum sagittatum	NAT	6			B	
Polygonum scandens	NAT	3	M,B			
Populus deltoides	NAT	2	C,B	M	H	
Populus grandidentata	NAT	3	M	M		
Populus tremuloides	NAT	2	C,B	H	H	M,B
Potentilla norvegica	NAT	0	B	C	B	

Ranunculus pensylvanicus	NAT	5	C			
Ranunculus sp.		0	C			
Rorippa palustris	NAT	3	H,B			
Rubus sp.	NAT	0		M	H	
Rumex maritimus	NAT	2	C,B			
Rumex crispus	INV	0			C	
Sagittaria latifolia	NAT	3	C,H,M,B		C,H,M	C,M,B
Salix eriocephala	NAT	4	C,H		C,H	C
Salix exigua	NAT	2	C		C,H,B	C
Salix purpurea	INT	0		C		C,B
Salix sp.		0	C,H	C	C,H,M,B	C,H
Schoenoplectus acutus	NAT	6		C,H,M	C,H,M,B	C,H,M,B
Schoenoplectus pungens	NAT	5	C,H,B	C,H,M	C,H,M,B	C,H,M,B
Schoenoplectus tabernaemontani	NAT	4	C,H,M,B	C,H,M	C,H,M,B	C,H,M,B
Scirpus sp. (1)	NAT	0	C,H,B	C,H,M	H,M,B	C,B
Scirpus sp. (2)	NAT	0	C,B	M	M,B	
Scirpus atrovirens	NAT	3	C		C	C,M
Setaria pumila	INT	0	M,B			
Solanum dulcamara	INV, INT	0	C		B	
Solidago canadensis	NAT	1	C,H,B	C,H,M	M	C,H
Solidago gigantea	NAT	3	C,H,M,B	H,M		C,H,M,B
Solidago sp.		0	C,H,M,B	C,H,M	H,M,B	C,H,M,B
Sonchus arvensis	INT	0		C		
Sonchus asper	INT	0		H,M		H,B
Sonchus oleraceus	INT	0	C,M,B			
Sonchus sp.		0	C,B	M	H,M,B	C
Sparganium eurycarpum	NAT	5	C,H,M,B		H,B	C
Spiraea alba	NAT	4		C,H,M		
Stachys palustris	NAT	5	B	C,H		
Taraxacum officinale	INT	0	C,M	H,M	H,M	M,B
Thalictrum dasycarpum	NAT	4	C,B			
Thuja occidentalis	NAT	9		H,M		
Trifolium repens	INT/INV	0	C,H		B	H
Typha angustifolia	INT	0	C,H,M,B		C,H,B	C,H,B
Urtica dioica	NAT	1	C,B	C,H	H,B	
Verbena hastate	NAT	3	C,B			
Veronica sp.		0	B			
Vitis riparia	NAT	2	C,H,M,B			
Xanthium strumarium	NAT	1	C,B			
Zizania palustris	NAT	8			H	
unknown aster #1		0		H		
unknown dicot seedling		0	C,M,B	C,H,M	H	C,H,M
unknown immature grass		0	C,H,B	H,M	H	C
unknown species 10		0	M			

unknown species 11	0	H	C,H	B	
unknown species 2	0	M	C,H		
unknown species 3	0	H,M,B	C,H,M	H,B	
unknown species 4	0	H	M		
unknown species 6	0		H	B	M
unknown species 7	0			H,M,B	C,H,B