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Whole-lake Herbicide Treatments for Eurasian Watermilfoil in Four Wisconsin Lakes: Effects on Vegetation and Water Clarity

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Abstract

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Four pilot whole-lake herbicide treatments for extensive Eurasian watermilfoil (EWM) (*Myriophyllum spicatum* L.) infestations were conducted in Wisconsin between 1997 and 2001 using fluridone at a range of dosages (6-16 µg/L). Annual post-treatment data (4-7 years) were evaluated to assess (1) effects on exotic plants; (2) changes to native plant communities; and (3) effects on water clarity. Temporal shifts in treatment lakes were compared against natural fluctuations in untreated reference lakes. In conjunction with aggressive follow-up spot treatments with 2,4-D or manual removal, fluridone treatments provided between 1 and 4 years of substantial EWM relief, with the exotic ultimately re-establishing at pre-treatment levels or greater in 3 of the 4 lakes. Native plant communities shifted in all 4 lakes following fluridone treatment. The large decreases, outside the range seen in untreated lakes (first quartile of the reference lake distribution) for all treatment lakes containing EWM, *Elodea canadensis*, *Ceratophyllum demersum*, and *Najas flexilis*, strongly suggest a direct effect of the fluridone treatment. We observed large increases, outside the range seen in untreated lakes (fourth quartile of the reference distribution), for 1 of 2 treatment lakes with

Potamogeton crispus, and 1 of 2 treatments with *Chara* spp. Secchi depth decreased significantly in 2 of the 3 lakes for which data were available. Future applications should consider, among other criteria, the dominant natives in the plant community, their sensitivity to fluridone, and potential impacts associated with decreased water clarity.

Key Words: aquatic plant control, fluridone, herbicide, invasive species, lake management, macrophytes, *Myriophyllum spicatum*, water quality

Aquatic plant management in lakes frequently involves the control of exotic invasive species. Invasive taxa, particularly Eurasian watermilfoil (*Myriophyllum spicatum* L., hereafter referred to as EWM), can potentially grow in dense monotypic stands that may result in recreational impairment and ecological alterations. At the microhabitat scale, stands of EWM can modify temperature (Unmuth *et al.* 2000) and dissolved oxygen concentrations (Miranda *et al.* 2000, Unmuth *et al.* 2000). Structural alterations in plant communities corresponding to EWM proliferation can even affect fish foraging dynamics (Engel 1995). In addition, the diversity and abundance of the native plant community may decline if EWM becomes dominant (Madsen *et al.* 1991).

Given the potential negative ecological, aesthetic, and recreational impacts of EWM infestations, several biological, physical, and chemical control techniques have been developed (Madsen 2000). One chemical used to control EWM, fluridone, 1-methyl-3-phenyl-5-[3-(trifluoromethyl) phenyl]-4(1H)-pyridinone, is a systemic herbicide that disrupts photosynthetic pathways (Bartels and Watson 1978, Berard *et al.* 1978, Devlin *et al.* 1978, Smith and Barko 1990), causing plant death within 60 to 90 days (at low doses). Due to rapid dissipation (Fox *et al.* 1996) and long contact requirements (>30 days; Netherland and Getsinger 1995a, Netherland and Getsinger 1995b, Netherland *et al.* 1997), fluridone is usually applied as a whole-lake treatment. As a result, both positive and negative aspects of this type of treatment translate to the whole-lake ecosystem.

As with any control method, the ecological benefits and costs of whole-lake (*i.e.*, ecosystem) manipulations with fluridone should be systematically evaluated over both the short and long terms (weeks to multiple years). Specifically, direct effects on target and nontarget vegetation and indirect effects of large-scale vegetation removal and die-off on water quality may occur over multiple temporal scales. Short-term control of EWM (1 yr post-treatment) has been achieved with fluridone (Getsinger *et al.* 2002, Madsen *et al.* 2002), but longer-term data are lacking in the literature.

Native vegetation may be negatively affected by fluridone (Kenaga 1993, Kenaga 1995, Welling *et al.* 1997), so setting appropriate dosages has emerged as an important issue in conducting whole-lake chemical treatments. In the past, high dose treatments (12-29 μg/L) consistently resulted in loss of native species richness and cover (Kenaga 1993, Kenaga 1995, Welling *et al.* 1997). Lower concentrations might still control EWM with limited impacts on native vegetation (Netherland and Getsinger 1995a, Netherland and Getsinger 1995b, Netherland *et al.* 1997). To date, field assessments of low concentration treatments (<8 μg/L) either did not show consistent control of target vegetation (Smith and Pullman 1997), or did not assess post-treatment changes to native species individually (Madsen *et al.* 2002). Additionally, to our knowledge, there are no published data regarding the potential water quality effects of whole-lake treatments, although O'Dell *et al.* (1995) found a reduction in water clarity and increases in total phosphorus and chlorophyll *a* after a partial-lake treatment.

With expanding invasions of Eurasian watermilfoil in Wisconsin lakes, the Wisconsin Department of Natural Resources (DNR) has received requests to permit whole-lake fluridone treatments in the state's public waters with increasing frequency. Here, we summarized results of the 4 whole-lake treatments permitted in Wisconsin between 1997 and 2001. Specifically, we assessed short- and long-term effects of whole-lake fluridone treatments on exotic vegetation, native vegetation, and water clarity to evaluate the ecological benefits and costs associated with whole-lake treatments in Wisconsin.

Methods

The geographic distribution of treated lakes varied, with one lake in central Wisconsin, two in the southeast, and one in northwestern Wisconsin (Fig. 1). Lakes were relatively small and shallow, with size and depth maxima of 85 ha and 9.8 m, respectively (Table 1). Three of the lakes (Potter, Clear, and Bughs) were seepage lakes with groundwater inflow, and the fourth lake (Random) was a drainage lake with surface water inflow. Aquatic vegetation dominated all 4 lakes (96-100% littoral area), with macrophyte communities ranging from 10 to 26 submerged and floating-leaf species.

Treatment timing (fall or spring) and dosage varied among lakes (Table 1). Certified pesticide applicators applied a liquid formulation of fluridone (Sonar® AS, SePRO Corp. Carmel, IN), to all lakes via subsurface injection booms and weighted drop lines, using a digital flow meter to ensure even chemical distribution. Consultants monitored residual concentrations regularly using SePRO's FasTEST® immunoassay to maintain concentrations greater than 4 μg/L

Figure 1.-Map showing the locations of the four lakes treated with fluridone in Wisconsin (●), the reference lakes for vegetation (▲; Fish, Mendota, Monona, Wingra), and the reference lakes for water clarity (■; Eagle Spring, Little Green, Windfall).

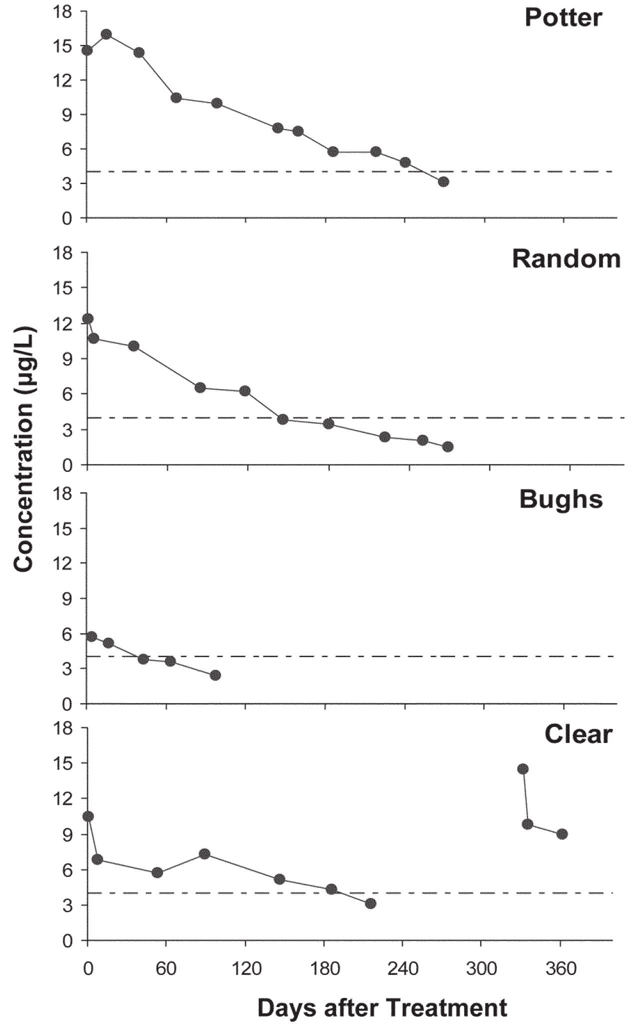

Figure 2.-Residual concentrations of fluridone as measured by FasTEST®. Dashed line indicates the concentration required for death of Eurasian watermilfoil after 60 days, 4 µg/L. Clear Lake received 2 separate applications.

for 30 days (Fig. 2), a relationship that has shown inhibition of EWM growth in mesocosm studies (Netherland and Getsinger 1995a). Treatment duration varied between 45 and 250 days above the 4 µg/L threshold. All 4 lakes received aggressive post-fluridone 2,4-D (2,4-dichlorophenoxyacetic acid) spot-treatments and/or manual removal of returning EWM to maintain the benefits of fluridone treatment for as long as possible.

Vegetation surveys were conducted during peak biomass (late June through August) both pre- and post-treatment. Evenly spaced transects extending from shore toward the center of each lake were selected (Potter, 25; Random, 19; Bughs, 8; Clear, 25), with sample sites set at depths of 0.5 m, 1.5 m, 2.7 m, and 3.7 m along each transect. Four rake tows were conducted at each site, with presence/absence of macrophytes recorded.

Species with a 10% littoral frequency of occurrence for at least one sampling event were tested for significant shifts using a χ-squared test. Change was measured as the difference between the number of occurrences prior to treatment and the number of occurrences in a given year following treatment. Changes in natives corresponding to specific ecological events were also compared, including (1) initial EWM decline, (2) subsequent successional shifts, and (3) return of EWM. Because the sampling design was not appropriate for assessing increases or decreases in rare or sparsely distributed plants, we limited our results and discussion to relatively large-scale shifts in commonly occurring species.

To assess whether community shifts in treated lakes were within or outside the range encountered due to naturally-occurring interannual variation, we selected 4 reference lakes with high densities of EWM from the North Temperate Lakes Long-Term Ecological Research (LTER) dataset. Long term data from similar, but untreated, reference lakes can provide a standard for identifying changes that are larger than those due to natural annual variation (Schindler 1987, Carpenter 1990) and can provide the basis for a randomization test that accounts for the possible temporal dependence among observations (Box *et al.* 1978, Fortin and Jacquez 2000). A

Table 1.-Physical attributes and treatment details of Wisconsin lakes treated with fluridone. Littoral area was calculated as the percent of total lake area <6 m in depth, as determined from bathymetric maps. The "10-bump-6" treatment plan on Clear Lake refers to an initial target concentration of 10 μg/L with a follow-up treatment to reestablish a 6 μg/L concentration.

Lake Name	County	Size (ha.)	Maximum Depth (m)	Primary Water Source	Inlet	Outlet	% Littoral	Year Treated	Timing	Target Concentration (μg/L)	Peak Concentration (μg/L)
Potter	Walworth	65.8	7.9	Groundwater	N	Y	98	1997	Fall	14	15.9
Random	Sheboygan	84.8	6.4	Surface water	Y	Y	96	1999	Fall	12	12.4
Bughs	Waushara	12.1	5.5	Groundwater	N	N	100	2000	Spring	6	5.7
Clear	Sawyer	31.3	9.8	Groundwater	Y	Y	96	2000, 2001	Fall, Fall	10-bump-6, 10	10.5, 14

Table 2.-Change in occurrence of submerged and floating-leaf vegetation on Potter, Random, Bughs, and Clear Lakes. Species with a 10% frequency of occurrence for at least one sampling point were tested for significant shifts using a χ-squared test. Changes were measured as the difference in the number of occurrences prior to treatment and the number of occurrences in a given year following treatment. Direction of change indicated by + (increase, $P < 0.05$) or – (decrease, $P < 0.05$). Nonsignificant changes ($P > 0.05$) are indicated by ns. Shading indicates return of milfoil to >50% frequency of occurrence. Arrows indicate 2,4-D spot-treatments.

Lake		Potter							Random			Bughs			Clear		
Year after treatment		**1**	**2**	**3**	**4**	**↓5↓**	**6 Jun↓**	**6 Sep**	**1**	**2**	**3↓**	**1**	**2**	**3↓**	**0**	**1 ↓**	**2 ↓**
Exotic																	
Myriophyllum spicatum	Eurasian watermilfoil	-	-	-	-	-	-	ns	-	-	+	-	ns	+	-	-	-
Potamogeton crispus	Curly-leaf pondweed	-	ns	ns	ns	ns	ns		ns	+	+			-			
Native																	
Ceratophyllum demersum	Coontail	ns	-	-	-	-	-	-						-			
Chara spp.	Muskgrass	ns	ns	+	+	+	ns	-	+	ns	+			ns			
Elodea canadensis	Canadian waterweed	-	-	-	-	-	-	-						-	-	-	+
Heteranthera dubia	Water star-grass														ns	+	ns
Najas flexilis	Slender naiad	-	-	-	-	-	-	ns						-	-	-	-
N. marina	Spiny naiad								-	-	-						
Nitella spp.	Stoneworts														ns	+	+
Nuphar spp.	Yellow water lily								ns	ns	ns						
Potamogeton amplifolius	Large-leaf pondweed														ns	-	ns
P. diversifolius	Water-thread pondweed														-	-	-
P. foliosus	Leafy pondweed														-	-	ns
P. illinoensis	Illinois pondweed								ns	ns	+			+			
P. natans	Floating-leaf pondweed								ns	ns	+						
P. robbinsii	Robbin's pondweed														ns	+	+
P. zosteriformis	Flat-stem pondweed														ns	+	+
Stuckenia pectinata	Sago pondweed	ns	ns	+	+	+	ns	-	+	+	+				ns	+	+
Vallisneria americana	Water celery														ns	+	+

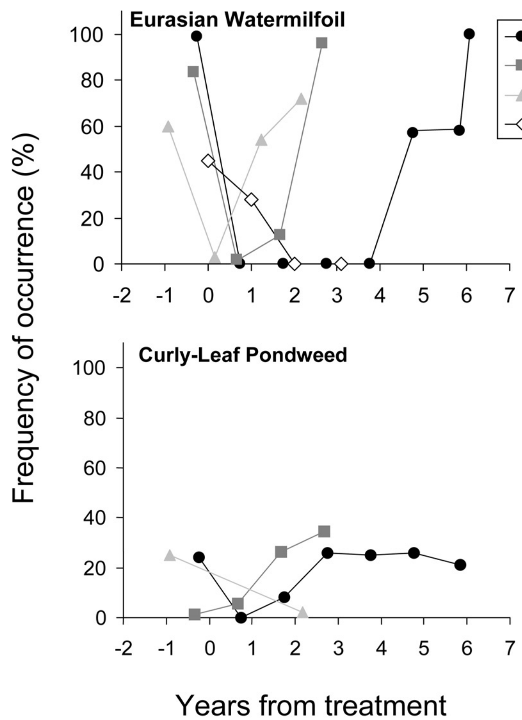

Figure 3.-Frequencies of occurrence for the exotic plant species present in treated Wisconsin lakes. Statistically significant changes are given in Table 2. Post-treatment Eurasian watermilfoil frequencies for Bughs Lake were estimated based on distributions known from informal surveys and lake area permitted for follow-up Eurasian watermilfoil control.

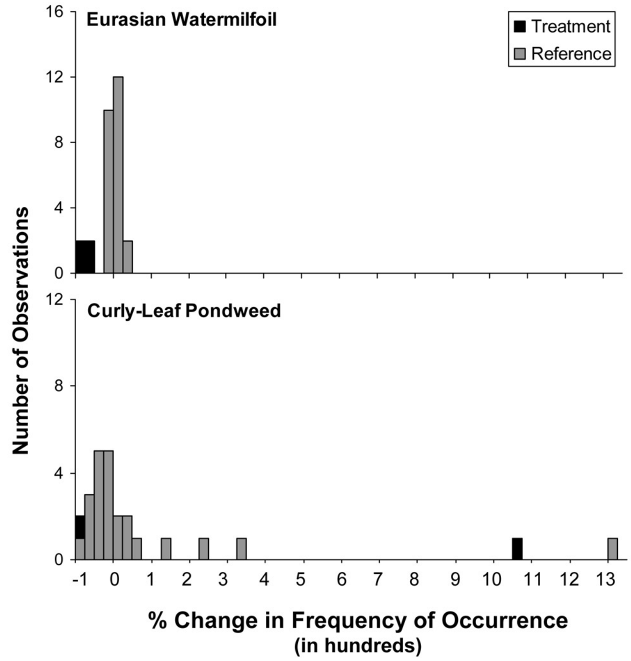

Figure 4.-Reference and treatment distribution of exotic species as percent change in frequency of occurrence. Percent change for treatment lakes is from pre-treatment to the average of 2 years post-treatment; for reference lakes it is the percent change of running 3-year trends (percent change from 1st year to the average of 2nd and 3rd years) from 1995 to 2004.

reference distribution for naturally occurring changes was constructed and compared to changes observed in treatment lakes. Change for both treatment and reference lakes was measured for all species >10% frequent as the percent difference between pre-treatment and post-treatment (an average of 1 and 2 years post-treatment). For reference lakes, this was calculated for all possible years from 1995 to 2004 to obtain the maximum number of year-to-year changes. Observed changes in the study lakes were then compared to the percentiles of the reference distribution; changes at extreme ends, or outer quartiles (Hoenig *et al.* 1987, Schupp 1992), of the reference distribution suggested a likely effect due to treatment.

Trained volunteers involved in the Wisconsin DNR Self-Help water quality monitoring program recorded Secchi depth in the deepest area of each lake. Additional Secchi measurements for Potter Lake were available from the United States Geological Survey. There were no pre-treatment Secchi data for Bughs Lake, but limited post-treatment data were available from a third-party consultant. The 2-yr mean pre- and post-treatment Secchi depth for pooled July and August samples were compared on all lakes with a student's t-test. These late summer samples provided the best estimate of peak algal growth. In addition to these in-lake comparisons, we subjected the Secchi data to the same reference distribution constructions as the vegetation data, using DNR Self-Help data for 4 lakes in close geographic proximity, size, and trophic status to treatment lakes.

Results

Potter Lake

Prior to fluridone treatment (October 1997, 15.9 μg/L), EWM dominated the lake with a 99% frequency of occurrence. Following treatment, we did not detect EWM for 4 summers (1998-2001; Fig. 3), although regrowth was observed beginning the third year following treatment (2000). This initial decrease can be directly attributed to treatment because it represents an outlier in comparison to natural EWM fluctuations in reference lakes (< 1st percentile; Fig. 4). These small areas of regrowth were treated with 0.4-ha (2001) and 2-ha (2002) 2,4-D treatments. Despite the attempted follow-up control, in 2002 (5 seasons after treatment), EWM frequency of occurrence increased from 0 to >50% in 6 weeks. After this rapid reestablishment, the lake received a third 8.1-ha follow-up treatment of 2,4-D. By early September 2003, EWM

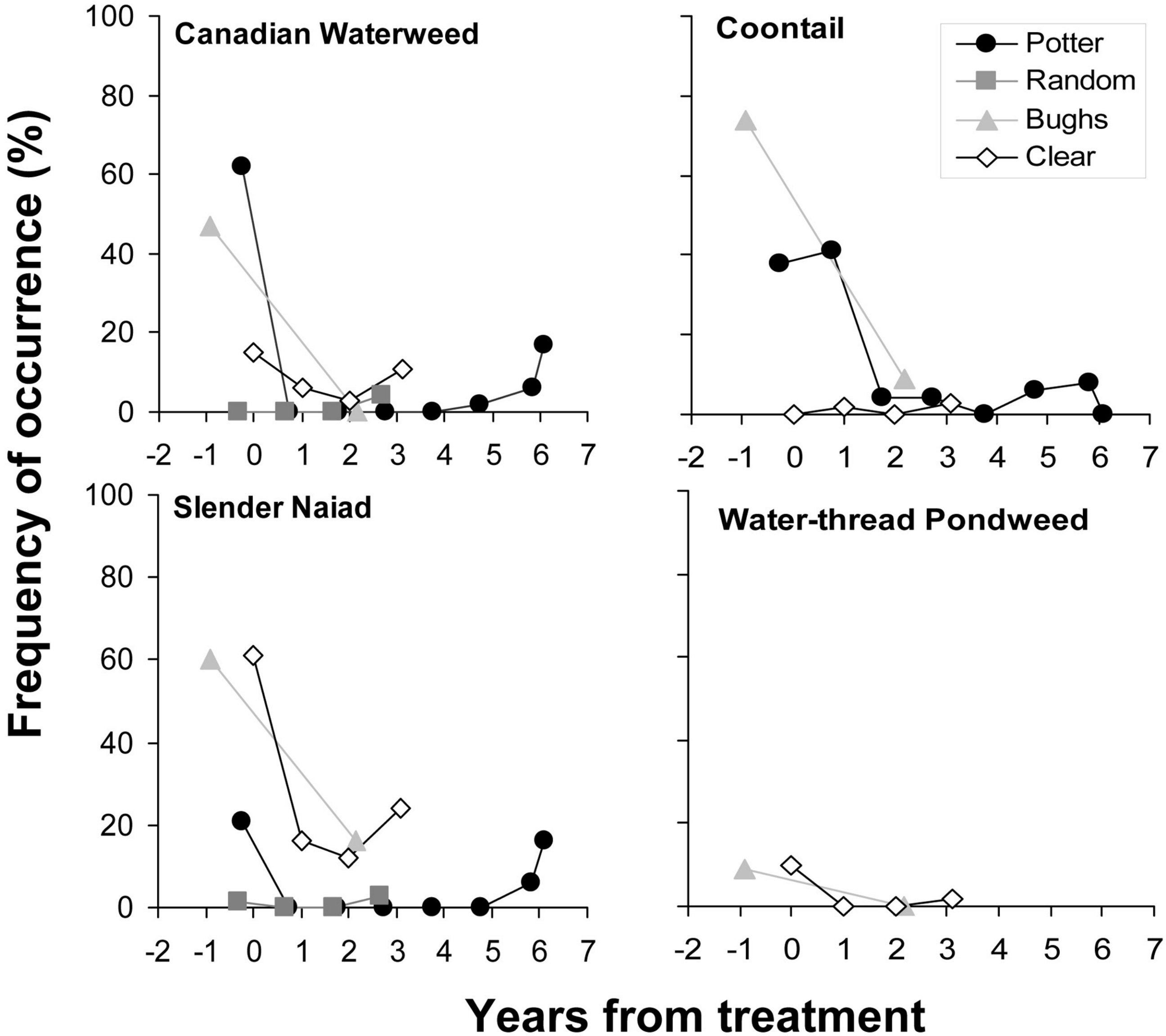

Figure 5.-Frequencies of occurrence for the native plant species that tended to decrease following treatment. Statistically significant changes are given in Table 2.

frequency of occurrence had reached 100%. The other exotic plant present, curly-leaf pondweed (*Potamogeton crispus* L.), decreased and was not found in surveys one year post-treatment, falling in the low end of the reference distribution (29th percentile). In the third year following treatment through the most recent survey, curly-leaf pondweed was detected at pretreatment levels.

The treatment affected several native taxa (Table 2; Fig. 5). Three natives had a significant negative response to treatment. Canadian waterweed and slender naiad (*Najas flexilis* (Willd.) Rostk. & W.L.E. Schmidt) were initially lost from the system, although they were again detected at low frequencies following 5 and 6 years, respectively. Compared to the reference distributions, shifts in Canadian waterweed represent the extreme low end of the range of potential natural variation (9th percentile), and slender naiad was beyond the range seen in reference lakes (21st percentile; Fig. 6). Coontail decreased the second year following treatment, falling within the low end of the range (25th percentile) of reference lake changes. It then remained between 0 and 6% frequency through the sixth year after treatment. The follow-up spot-treatments of 2,4-D may have contributed to low frequencies of coontail during 2001-2003; however, we believe this effect to be minimal on a whole-lake scale due to the small size of the spot-treatments (0.6, 3, and 12% of lake area). Frequencies of

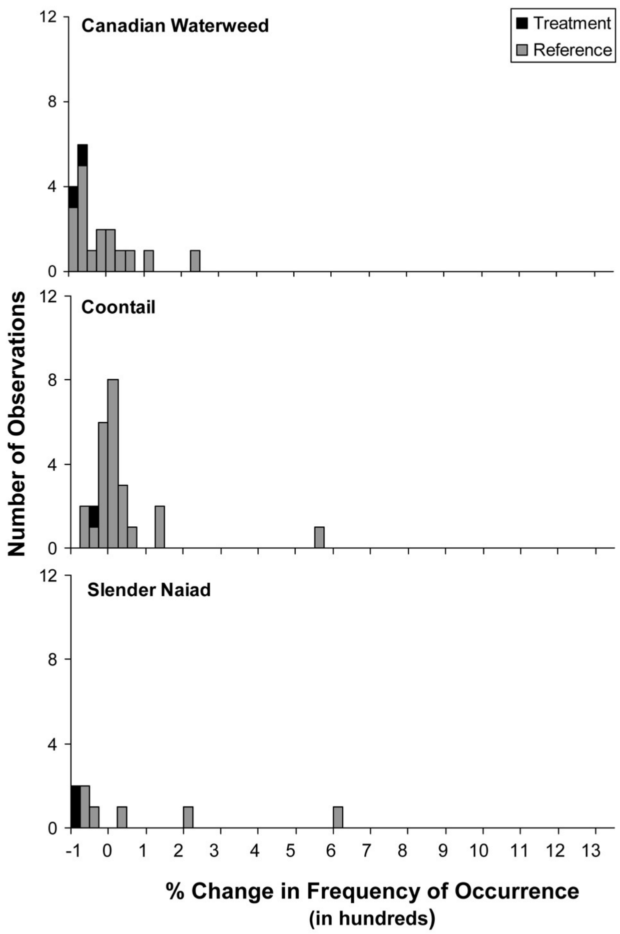

Figure 6.-Reference and treatment distribution of decreasing native species as percent change in frequency of occurrence. Percent change for treatment lakes is from pre-treatment to the average of 2 years post-treatment; for reference lakes it is the percent change of running 3-year trends (percent change from 1st year to the average of 2nd and 3rd years) from 1995 to 2004.

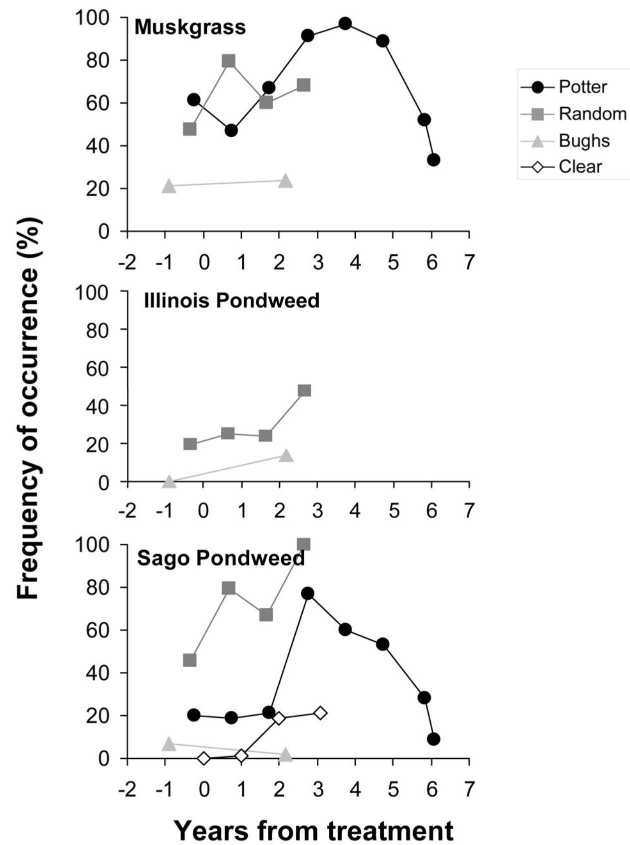

Figure 7.-Frequencies of occurrence for the native taxa that tended to increase following treatment. Statistically significant changes are given in Table 2.

occurrence of 2 native taxa increased for some length of time (Table 2; Fig. 7): muskgrass (*Chara* spp.) increased 4 seasons after treatment, and sago pondweed (*Stuckenia pectinata* (L.) Börner) increased 3 seasons post-treatment. The changes for muskgrass and sago were similar to reference distribution changes, in the 40th and 41st percentiles, respectively (Fig. 8). In the most recent survey (6 years post-treatment), both taxa had decreased relative to pre-treatment (Table 2).

The 2-month late summer Secchi depth average on Potter Lake decreased from 2.0 m pre-treatment to 1.0 m post-treatment ($P < 0.01$; Fig. 9), in the lower 17th percentile of changes in the reference lakes (Fig. 10). This decrease was the largest to occur in the 3 Wisconsin lakes for which pre- and post-treatment Secchi depths were available.

Random Lake

Prior to treatment (October 1999, 12.4 µg/L), Random Lake had an EWM infestation that had reached 83% frequency of occurrence. Following treatment, EWM was reduced to low frequencies (1-13%) for 2 years (2000-2001), falling below the first percentile of the reference lake changes (Fig. 4). Control from fluridone treatment ended in the third year (2002), when EWM rebounded to over 95% frequency (Fig. 3), an increase from pre-treatment (Table 2). Spot-treatments of 2,4-D and mechanical harvesting were then successfully used to control EWM for 3 years following reestablishment (2002-2004). With spot-treatments and harvesting, EWM coverage went from 95% frequency of occurrence in 2002 to 11% of lake area (9.3 ha) in 2004. Curly-leaf pondweed did not show a significant change the year after treatment but increased in the second and third years. This change

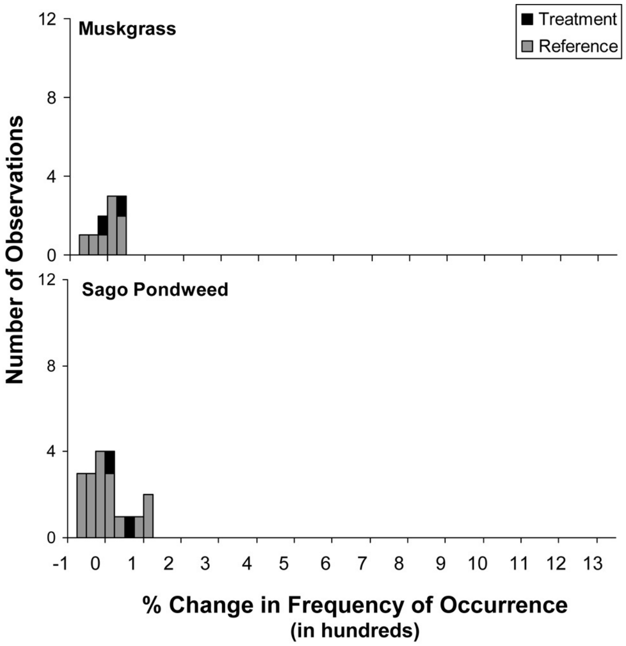

Figure 8.-Reference and treatment distribution of increasing native species as percent change in frequency of occurrence. Percent change for treatment lakes is from pre-treatment to the average of 2 years post-treatment; for reference lakes it is the percent change of running 3-year trends (percent change from 1st year to the average of 2nd and 3rd years) from 1995 to 2004.

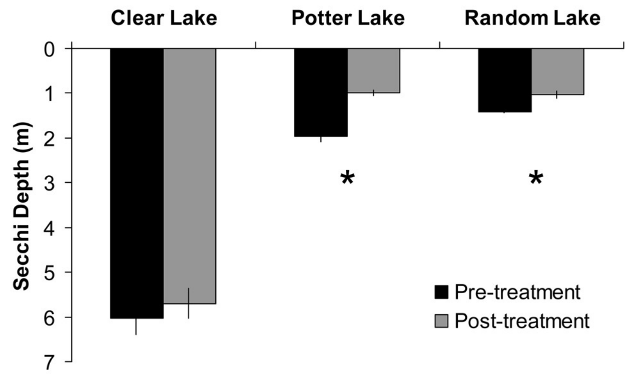

Figure 9.-Mean July/August Secchi depth readings averaged for 2 years pre-treatment and 2 years post-treatment (± standard error). T-tests indicate significant reductions in Secchi on Potter ($P <$ 0.01) and Random ($P < 0.01$) following treatment (Clear Lake, $P =$ 0.39). Data not available for Bughs Lake.

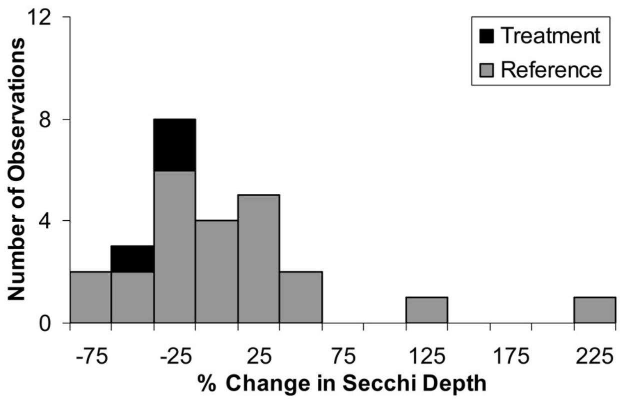

Figure 10.-Reference and treatment distribution of Secchi depth as percent change. Percent change for treatment lakes is from pre-treatment to the average of 2 years post-treatment; for reference lakes it is the percent change of running 3-year trends (percent change from 1st year to the average of 2nd and 3rd years).

represents the extreme high end of the reference distribution (99th percentile).

Natives susceptible to fluridone were originally present in low abundances (Fig. 5). Spiny naiad (*Najas marina* L.) decreased significantly following treatment (Table 2) and was undetected in transect surveys for at least 3 years. Two native species, muskgrass and sago pondweed, increased significantly immediately following treatment (Table 2; Fig. 7), representing the 91st and 60th percentile of the reference distribution (Fig. 8). Floating-leaf pondweed (*Potamogeton natans* L.) and Illinois pondweed (*P. illinoensis* Morong) did not immediately increase, but had increased after 3 years.

The 2-month late summer Secchi depth average on Random Lake decreased from 1.4 m to 1.0 m following treatment ($P < 0.01$; Fig. 9), falling at the 24th percentile of observed reference lake changes (Fig. 10).

Bughs Lake

Prior to treatment (May 2000, 6 µg/L), EWM was present in the lake at a frequency of 59%. No formal post-treatment survey occurred until the third year after treatment, although informal surveys indicated that EWM was never eradicated. Reductions of approximately 95% of EWM biomass were estimated the summer of treatment (July 2000; Gansberg, pers. observ.; Table 2; Fig. 3). By August the following year (2001, second growing season after treatment) an informal survey indicated EWM was again present at most of its pre-treatment distribution (Cason 2002). Based on these estimations, the temporary reduction of EWM in Bughs Lake falls below the first percentile of the reference distribution (Fig. 4).

Successful treatments with 2,4-D occurred following the lake-wide re-establishment of EWM in 2002, 2003 and 2004. A 7.3-ha 2,4-D treatment took place in spring of 2002 on the entire area of EWM regrowth (72% of the lake). With these treatments, EWM steadily declined, and treatment sizes decreased to 5.7 ha in 2003 and 3 ha in 2004. Application rates were at or near 112 kg/ha. The only formal post-treatment survey, taken after the first 2,4-D follow-up treatment, indicated EWM was present at 6% frequency of occurrence.

Overall, the presence of native species in Bughs Lake was greatly reduced following the fluridone treatment (Table 2; Fig. 5). Significant reductions occurred for coontail, Canadian waterweed, and slender naiad. Other species potentially affected, but sparsely distributed and not statistically analyzed, included: water-thread pondweed (*Potamogeton diversifolius* Raf.; 9% to 0% frequency) and northern watermilfoil (*Myriophyllum sibiricum* Kom.; 4% to 0% frequency). The only species to increase on Bughs Lake was Illinois pondweed (Table 2; Fig. 7).

There were no pre-treatment Secchi data available for Bughs Lake. Water clarity was <1 m the summer after treatment (2000), with blue-green algal blooms present (Gansberg, pers. observ.). In February of 2001, dissolved oxygen levels in the lake became low enough to threaten fish survival, and an aerator was installed to prevent a winter fish kill. Post-treatment Secchi depths indicated that water transparency was 1.2 m in June of the second season after treatment (2001), and 4 m in June of the following year (2002).

Clear Lake

Prior to treatment (October 2000; 10 µg/L), EWM in Clear Lake was present at 45% frequency. Following treatment, EWM was still relatively widespread (28% frequency; Fig. 3), so a second treatment took place in fall 2001 (10 µg/L). Although we did not detect EWM in formal surveys for 2 years following the second treatment (which falls below the first percentile of the reference distribution; Fig. 4), small areas of regrowth were observed and spot-treated with 2,4-D both of those years (0.01 ha, 0.1 ha, respectively).

Plant species that had an initial negative response to treatment included (Table 2; Fig. 5): Canadian waterweed, slender naiad, water-thread pondweed, and leafy pondweed (*Potamogeton foliosus* Raf. subsp. *foliosus*). Of these, Canadian waterweed and leafy pondweed had returned to pre-treatment levels by the summer of 2003, the third year after initial treatment. The changes for Canadian waterweed and slender naiad fell in the 18th and 24th percentile of the reference distribution, respectively (Fig. 6). Water star-grass (*Heteranthera dubia* (Jacq.) MacMill) increased the year after treatment before returning to pre-treatment levels. Increases also occurred in 5 species by the third year after the initial treatment (Table 2; Fig. 7): stoneworts, Robbin's pondweed (*Potamogeton robbinsii* Oakes), flat-stem pondweed (*P. zosteriformis* Fernald), sago pondweed and water celery.

Pre-treatment mean Secchi depth on Clear Lake was 6.0 m. This did not change significantly following treatment, when mean Secchi depth was 5.7 m ($P = 0.39$; Fig. 9). This change is in the 28th percentile of reference lake changes (Fig. 10).

Discussion

All fluridone treatments in Wisconsin significantly and temporarily reduced Eurasian watermilfoil coverage. Significant nuisance relief lasted 1 to 4 growing seasons, with initial whole-lake treatments augmented by follow-up chemical spot treatments or manual removal. In 3 of the treatments, EWM subsequently returned to pre-treatment levels or higher (Fig. 3), and two lakes (Potter and Random) have been retreated with fluridone. The widespread manner and rapid rate with which EWM returned in these 3 lakes strongly suggests regeneration rather than new introductions. EWM control on Bughs Lake, which had only one year of EWM relief from fluridone, has steadily improved with 2,4-D treatments. Post-fluridone control on Random Lake had also improved with a combination of 2,4-D treatments and harvesting.

We documented inconsistent field responses of curly-leaf pondweed, an exotic *Potamogeton* susceptible to fluridone (SePRO 2002). Curly-leaf decreased in Bughs, increased in Random, and initially decreased but returned to pre-treatment levels in Potter. Other field applications of fluridone for EWM have been associated with either increases or no change in curly-leaf pondweed (Welling *et al.* 1997, Getsinger *et al.* 2002, Madsen *et al.* 2002). These differences may be due to treatment timing, rapid regrowth from turions, or competitive release following the removal of EWM (Madsen *et al.* 2002).

Generally, effects on nontarget vegetation in Wisconsin lakes correspond with those reported by SePRO and mesocosm experiments (McCowen *et al.* 1979, Netherland *et al.* 1997, Nelsen *et al.* 1998, Poovey *et al.* 2004) designed to assess sensitivity of specific taxa to fluridone. Coontail, Canadian waterweed, slender naiad, spiny naiad, leafy pondweed, and water-thread pondweed (Table 2; Fig. 5) decreased in all treatment lakes in which they were present. Significant decreases occurred in both high (14 µg/L) and low concentration (6 µg/L) treatments. Treatment dosages were reduced over time to minimize negative impacts to natives (Netherland *et al.* 1997, Smith and Pullman 1997, Madsen *et al.* 2002); however, susceptible natives were affected in all treatments.

Natives that increased in at least one instance (for some length of time) included muskgrass, sago pondweed, stoneworts, Illinois pondweed, floating-leaf pondweed, Robbin's pondweed, flat-stem pondweed, and wild celery (Table 2; Fig. 7). EWM has been shown to competitively exclude native freshwater vegetation (Lind and Cottam 1969, Titus and Adams 1979, Madsen *et al.* 1991, McFarland and Rogers 1998), but with wide-scale removal of this highly competitive, lightlimiting invasive, surviving native vegetation may respond by increasing growth, abundance and distribution (Hauxwell *et al.* 2004). Without vigilant monitoring and retreatment, however, these taxa may again be competitively excluded upon return of EWM (Hauxwell *et al.* 2004).

Due to the staggered timing of fluridone treatments (1997-2001), these consistent changes observed for individual species across lakes are likely due to treatment and not variable environmental influences. In some cases, the magnitude of changes observed in treated lakes is greater than observed in a 10-year span of interannual variation for 4 reference lakes. The large decreases, outside the range seen in untreated lakes (first quartile of the reference distribution) for all treatments containing EWM, Canadian waterweed, coontail, and slender naiad (Fig. 4 and 6), strongly suggest a direct effect of the fluridone treatment. We observed large increases outside the range seen in untreated lakes (fourth quartile of the reference distribution) for 1 of 2 treatments with curly-leaf pondweed, and 1 of 2 treatments with muskgrass (Fig. 4 and 8). The second treatment for each of those species, as well as both treatments containing sago pondweed, were within the normal range of variation (inner quartiles), and therefore may not be directly or solely due to treatment.

We would expect decreases in water clarity following treatment to be driven by the magnitude of impact to the plant community, because nutrients released into the water column are available for immediate uptake by planktonic algae. Additionally, resuspension of sediments will increase as a function of the area of susceptible vegetation (Søndergaard and Moss 1998). Compared to our constructed reference distribution (Fig. 10), 2 of the 3 changes in water clarity are in the outer range of natural variability (first quartile). The lake with the largest decrease in water clarity (Potter) also had the largest initial decrease in vegetation, and the fewest successional increases of the 2 lakes exhibiting a decrease in Secchi. The other lake (Random) had a significant decrease in clarity despite having lost only one susceptible species following treatment, and having had 3 other species increase. Clear Lake did not exhibit a significant decrease in water clarity, and also had more fluridone-tolerant natives that increased following treatment than the other lakes. Bughs Lake, for which limited Secchi data were available, had the highest decrease in native plant frequency of all lakes, and anecdotal accounts of water quality and post-treatment Secchi measurements indicate that the clarity on Bughs Lake declined. Decreases in Secchi depth, as well as increased chlorophyll *a* and total phosphorus, occurred in another (partial-lake) fluridone treatment with large decreases in vegetation (O'Dell *et al.* 1995).

This resulting increase in algal densities following herbicide treatment has the potential to not only immediately decrease water clarity, but also to inhibit subsequent growth and reestablishment of native macrophyte populations (Kemp *et al.* 1983, Chambers and Kalff 1985, Barko *et al.* 1986, Scheffer *et al.* 1993, Jeppesen *et al.* 1997, Hauxwell *et al.* 2001) either through light limitation (Gasith and Hoyer 1998) or reduced nutrient availability (Carpenter *et al.* 1998), particularly in eutrophic lakes (Valley *et al.* 2006). Although this was not observed on any of the 4 Wisconsin lakes, a temporary state shift is a potential outcome of large-scale vegetation removal, especially in shallow lakes (Scheffer *et al.* 1993). Further water quality problems may result if the high biological oxygen demand associated with decomposing vegetation results in substantially reduced dissolved oxygen concentrations. This, combined with winter ice cover may have been the cause of low dissolved oxygen levels that threatened fish survival on Bughs Lake.

Conclusions

The results from the 4 Wisconsin lakes treated with fluridone indicate that 1 to 4 seasons of EWM relief might be expected, contingent on aggressive post-treatment monitoring and follow-up chemical or manual control. The impact on native species depended largely on the composition of the plant community prior to treatment and the sensitivity of those plants to various rates of fluridone; those lakes that had high frequencies of susceptible plants (naiads, coontail, milfoils, and Canadian waterweed) also had the largest negative impacts to the native plant community. Lakes with low frequencies of susceptible species may result in increases in tolerant taxa upon removal and continued suppression of EWM. Plant community composition and overall changes in distribution may contribute to water clarity reductions following whole-lake treatments.

These results indicate that whole-lake treatments as previously employed in Wisconsin may optimally result in temporary shifts in native plant communities, with probable subsequent return of EWM. Under less ideal conditions (lakes with high abundance of susceptible taxa), negative impacts to native plant communities and water quality occur. A systematic approach to determine which lakes are good potential candidates for whole-lake fluridone treatment is needed. This approach should consider, among other criteria, the dominant natives in the plant community and their sensitivity to set rates of fluridone, and the potential impacts that could result due to shifts in water quality. Use of such a systematic management approach will help limit negative impacts on native plant communities, as well as help predict the conditions that lead to reductions in water clarity.

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