Adaptive Management of Flowering Rush in the Detroit Lakes, MN

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ABSTRACT

Aquatic resource managers have limited resources to combat aquatic invasive plant species (AIS) infestations. Methodologies that control AIS with minimum resources should help managers allocate resources to other issues they face. Flowering rush (*Butomus umbellatus* L.) is spreading across the northern U.S. and southern Canada. Flowering rush relies on vegetative

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reproduction (primarily through rhizome buds) to colonize new sites and revegetate managed sites. Therefore, rhizome bud reduction should be a key goal in flowering rush management decisions. Management of flowering rush in Detroit Lakes, MN has shown that two diquat applications per growing season can reduce flowering rush biomass and bud density; however, in recent years, as new invaders arrived in the system (i.e., Zebra mussels) there were limited resources to address both AIS. Research was undertaken to determine if flowering rush could be controlled by single diquat applications (rather than two) in sites of low flowering rush prevalence. Treatment sites were designated as having very low, low, or high flowering rush prevalence (measured as percent frequency) with each receiving no, one, or two diquat treatments (0.37 mg L^{-1}), respectively. When compared to non-treated reference sites, flowering rush prevalence, biomass, and bud density in low prevalence sites did not increase after two years of single diquat applications while prevalence declined, and biomass and bud density remained constant in high prevalence sites. Total area infested by high prevalence levels of flowering rush declined over time even though total area infested increased during this study suggesting that adaptive management was sufficient to convert high prevalence sites to low prevalence sites. At peak infestation (2016), over 128 ha (316 ac) of flowering rush were being managed annually while in 2020 only 8 ha (20 ac) of flowering rush needed herbicide treatment in the Detroit Lakes. This adaptive management strategy suggests that single diquat applications are suitable to maintain control of sites with low flowering rush prevalence allowing resource managers to allocate resources elsewhere.

Key Words: Nuisance species, invasive species, *Butomus umbellatus*, diquat

INTRODUCTION

Flowering rush (*Butomus umbellatus* L.) is a rooted aquatic invasive plant in North America that is native to Eurasia. It was introduced to the Detroit Lakes, MN through the water garden industry in the early 1970's. After introduction, it had infested every major basin of the Detroit Lakes system by the 1990's (PRWD 2020a); predominantly in water less than four feet in depth (Marko et al. 2015; Madsen et al. 2016b). Flowering rush impaired the ecology of the Detroit Lakes by infesting fish spawning areas and displacing native vegetation and reduced recreational use areas for humans by infesting areas used for swimming, fishing, water skiing, and boating (Madsen et al. 2016b). Vegetative propagules called rhizome buds were likely the main vector of spread and colonization of new sites by flowering rush within the Detroit Lakes system.

Rhizome buds are vegetative structures that sprout from underground rhizomes (Marko et al. 2015). Rhizome buds allow flowering rush to persist in sites after management activities have reduced above and belowground biomass. Reduction of rhizome buds can be difficult as they are attached to flowering rush rhizomes by fragile stalks and can break away easily which can leave them in sediments to sprout after management activities have occurred. Reduction of rhizome buds must be a primary goal of resource managers expecting to attain long-term control of flowering rush.

In the late 1980's, resource managers in the Detroit Lakes started to utilize mechanical harvesters to try and slow the spread of flowering rush (PRWD 2020a). However, this did not control flowering rush and may have increased the rate of proliferation by spreading propagules (i.e., rhizome fragments and buds) that could colonize new sites within the system. By 1994, flowering rush had infested every major basin in the Detroit Lakes system (PRWD 2020a). In 2005, resource managers switched management strategies from mechanical to chemical control methods. Many herbicides were tested, but the herbicides did not reduce emergent and/or submersed flowering rush (bispyribac-sodium), would not work on submersed flowering rush in field sites (imazapyr and glyphosate), or lacked the contact time needed (2,4-D, triclopyr, imazamox, fluridone, endothall, flumioxazin) to control flowering rush in field locations of the Detroit Lakes (Poovey et al. 2012, 2013; Wersal et al. 2014; Madsen et al. 2016a). The contact herbicide diquat was the only herbicide that provided in-season reduction of flowering rush distribution, biomass, and rhizome bud number in Detroit Lakes field sites (Madsen et al. 2016a).

Diquat was first identified as a potential flowering rush control option by Poovey et al. (2012) in a laboratory trial. Poovey et al. (2012) found that submersed injections of diquat (0.37 mg L^{-1}) could reduce aboveground flowering biomass with six hours of contact time. Madsen et al. (2012) determined that six hours of contact could be attained in most plots in the Detroit Lakes. However, diquat does not translocate from foliage to belowground plant biomass so it was thought to be unlikely that one application of diquat would reduce belowground biomass of flowering rush and disrupt the plant life cycle in a way that would provide long-term reduction.

In 2012, Madsen et al. (2016a) developed a chemical control protocol for flowering rush reduction in the Detroit Lakes whereby diquat was applied twice per growing season (one month between herbicide applications) at the maximum rate (0.37 mg L^{-1}) as submersed injections to areas infested with flowering rush. This protocol provided in-season reduction of flowering rush distribution by 60%, above and below ground biomass reduction by 99% and 82%, respectively, and rhizome bud density reduction by 83% in the Detroit Lakes (Madsen et al. 2016a) but did not determine if long-term control of flowering rush could be attained. Parsons et al. (2019) confirmed that this same protocol could provide long-term reduction of flowering rush from year to year. Furthermore, Turnage et al. (2020) confirmed that this protocol could provide selective control of flowering rush when intermixed with hardstem bulrush (*Schoenoplectus acutus* (Muhl. ex Bigelow) A. Love & D. Love) in Detroit Lakes field sites. Reduction of belowground flowering rush biomass and rhizome bud density by multiple diquat treatments is likely attained by forcing the plant to use up starch reserves (chemical energy) stored in the rhizome to regrow foliage after herbicide treatments rather than using those reserves for bud production or rhizome expansion.

After broad scale reduction of flowering rush biomass and density in the Detroit Lakes, resource managers and stakeholders wanted to reduce the number of herbicide applications to low density sites in order to save resources and reduce unnecessary herbicide input to the lakes but were hesitant to do so without confirmation that reduced herbicide applications could provide continued suppression of flowering rush. Additionally, zebra mussels (*Dreissena polymorpha*) invaded the Detroit Lakes in 2014 and forced resource managers to split their focus and resources from one invasive species to two (PRWD 2020b) which intensified the need to quickly reduce management costs of flowering rush while not sacrificing the progress that had been made in flowering rush management.

An adaptive management approach was requested by resource managers in the Detroit Lakes that would establish a series of management thresholds that would allow resource managers to rapidly determine the appropriate diquat treatment protocol for an infested site based on flowering rush prevalence within the site prior to treatment. Field trials were initiated in 2015 in the Detroit Lakes to determine if fewer diquat applications could reduce or maintain flowering rush prevalence and/or biomass within low density sites. The objective of these field trials was to determine action thresholds whereby resource managers could adapt management strategies for flowering rush based on percent frequency of the plant at infested sites.

MATERIALS AND METHODS

Site Description

The study was conducted in 2015 and repeated in 2016 in waterbodies of the Detroit Lakes chain in MN. The Detroit Lakes system consists of five mesotrophic to meso-eutrophic glacial kettle lake basins along the Pelican River in Becker County, MN. The basins (Big and Little Detroit Lake, Curfman Lake, Lake Sallie, and Lake Melissa) and river are surrounded by the city of Detroit Lakes, MN (46.81333o Lat., -95.84472o Long.). Flowering rush infested approximately 115 ha (284.5 ac) in the Detroit Lakes in 2015 and 128 ha (316.6 ac) in 2016. In 2015, there were 24 flowering rush sites across the Detroit Lakes system utilized for assessment of plant community response to diquat treatments; of these, nine were used to assess flowering rush biomass response. In 2016, the number of infested sites increased to 29 for the community assessment while the original biomass assessment sites were utilized for a second year.

A point intercept survey using a weighted plant rake and handheld GPS unit was conducted in flowering rush infested sites in June of each year. A second survey was conducted at eight WAT in September of each year, and a third survey was conducted at 52 WAT (June the following year; Madsen and Wersal 2018). Survey points were at least 25 m apart in each site. The prevalence of flowering rush was determined in each site during the June surveys and used to assign a diquat treatment protocol to each site; all sites except reference sites had been treated with the diquat protocol developed by Madsen et al. (2016a) the previous year. Three sites were used as reference sites. Sites with less than five percent flowering rush prevalence were not treated with diquat (very low prevalence sites). Sites with greater than five but 20 percent or less flowering rush prevalence were treated once per growing season (low prevalence sites). Those sites with greater than 20 percent flowering rush prevalence were treated twice (high prevalence sites). Thresholds were established based on cost-benefit expectations (very low prevalence sites), similar work conducted on other AIS (low prevalence sites), and stakeholder perceptions of nuisance infestations (low and high prevalence sites; Table 1).

Plant Community Assessment

Species prevalence (percent frequency) from point intercept surveys in reference and treatment sites was analyzed from 2015 to 2016 and from 2015 to 2017 using a Cochran-Mantel-Haenszel test followed by a Fishers Exact test in the 'psych' and 'rcompanion' packages in the statistical software R (Madsen et al. 2016b; R Core Team 2020). Mean total, native, and non-native species richness at each survey event was analyzed in reference and treatment sites using a one way analysis of variance (ANOVA) procedure. Any differences detected in means were further separated using a Fishers Least Significant Difference (LSD) test (R Core Team 2020). All statistical analyses were conducted at the alpha=0.05 significance level.

Biomass Assessment

Prior to herbicide treatments in 2015 and 2016, nine flowering rush sites were selected for biomass sampling. Three sites were reference sites, three had lower flowering rush prevalence (five to ≤20 percent) and received one submersed diquat treatment per year, and three had higher prevalence (>20 percent) and received two diquat treatments per year (treatments were administered approximately one month apart). A 15-cm (six inch) diameter PVC coring device (0.018 m^2) was used to pull 40 sediment cores from each of the nine sites for a total of 360 cores per sampling effort (Madsen et al. 2007). Flowering rush tissues were removed from sediment cores, washed of dirt and debris, placed in labeled plastic bags, then shipped on ice to Mississippi State University (MSU). At MSU, samples were removed from plastic bags and separated into above and belowground biomass. Rhizome bud number was recorded and then above and belowground tissues were placed in separate labeled paper bags and dried in a forced air oven at 70C for three days. After drying, samples were weighed, and data recorded as g DW m^{-2}. Plots received diquat treatments (0.37 mg L^{-1}) in June (single and double applications) and July (only double applications). Biomass cores were pulled again at 8 and 52 WAT and processed in the same manner as pre-treatment samples.

Biomass and bud densities were analyzed with a mixed model analysis of variance (ANOVA) procedure using year as a random effect and number of diquat treatments as a fixed effect. If differences existed, a Fishers least significant difference (LSD) test was used to further separate treatment means. All statistical tests were conducted at the alpha=0.05 significance level (R Core Team 2020).

Infested Area Assessment

Binomial tests were used to assess changes infested area of flowering rush within each of the treatment categories (very low, low, and high prevalence sites). Binomial tests were conducted between the June 2015 and June 2016 survey periods. All statistical tests were conducted at the alpha=0.05 significance level (R Core Team 2020).

RESULTS AND DISCUSSION

Plant Community Assessment

A total of 23 species were recorded in the reference plots from 2015 to 2017 (Table 2). From 2015 to 2016 and from 2015 to 2017, flowering rush prevalence did not change in reference plots suggesting that flowering rush was near carrying capacity in these plots (Table 2). From 2015 to 2016, there were four species (coontail [*Ceratophyllum demersum* L.], whitestem pondweed

[*Potamogeton praelongus* Wulfen], sago pondweed [*Stuckenia pectinata* (L.) Borner], and common bladderwort [*Utricularia macrorhiza* Leconte]) that increased and three species (chara [*Chara* spp.], leafy pondweed [*Potamogeton foliosus* Raf.], and flatstem pondweed [*Potamogeton zosteriformis* Fernald]) that decreased in prevalence in the reference plots ($p<0.05$); prevalence of other species was not affected (Table 2). From 2015 to 2017, there was one species (coontail; $p<0.05$) that declined in prevalence while the presence of other species did not change in reference plots (Table 2). Mean total, native, and non-native species richness in the reference plots did not change from 2015 to 2017 (Figure 1).

There were 24 species recorded in plots receiving one diquat treatment from 2015 to 2017 (Table 3). From 2015 to 2016, flowering rush increased in prevalence by 10.9% but decreased in prevalence by 8.0% from 2015 to 2017 in sites receiving one diquat treatment ($p<0.05$; Table 3) suggesting that one application of diquat per year was enough to maintain flowering rush prevalence at a static level. From 2015 to 2016, there were four other species (northern watermilfoil [*Myriophyllum sibiricum* Kom.], curlyleaf pondweed [*Potamogeton crispus* L.], sago pondweed, and common bladderwort) that increased in prevalence while there were three species (star duckweed [*Lemna trisulca* L.], leafy pondweed, and Illinois pondweed [*Potamogeton illinoensis* Morong]) that declined in prevalence ($p<0.05$; Table 3); prevalence of other species were unchanged. From 2015 to 2017, there were two species (coontail and variable pondweed [*Potamogeton gramineus* L.]) that increased in prevalence and six species (chara, star duckweed, curlyleaf pondweed, Illinois pondweed, sago pondweed, and watercelery [*Vallisneria americana* Michx.]) that decreased in prevalence ($p<0.05$; Table 3); prevalence of other species was unchanged. Total, native, and non-native species richness were not affected in plots receiving one diquat application from 2015 to 2017 (Figure 1).

Twenty-five species were recorded in plots that received two diquat treatments from 2015 to 2017 (Table 4). Flowering rush decreased in prevalence both years (10.3% and 12.8%, respectively; $p<0.05$; Table 4) compared to 2015 levels. From 2015 to 2016, four species (curlyleaf pondweed, sago pondweed, common bladderwort, and watercelery) increased in prevalence and two species (leafy pondweed and flatstem pondweed) decreased in prevalence in sites that received two diquat treatments ($p<0.05$); prevalence of other species were unchanged (Table 4). From 2015 to 2017, variable pondweed that increased in prevalence and four species (star duckweed, northern watermilfoil, sago pondweed, and watercelery) decreased in prevalence ($p<0.05$); prevalence of other species were unchanged (Table 4). Total, native, and non-native species richness were unchanged in plots that received two diquat applications from 2015 to 2017 (Figure 1).

Total species richness was the same in reference and treatment plots in 2015 but declined in treatment plots in 2016 and 2017 ($p<0.05$) when compared to reference plots, which suggests that diquat treatments reduced total species richness over time (Figure 1). There was no difference in native species richness of reference plots nor plots that received a single diquat treatment from 2015 to 2017; however, sites that received two diquat treatments consistently had fewer native species than reference plots ($p<0.05$; Figure 1). There was no difference in non-native species (flowering rush and curlyleaf pondweed) richness between reference and treatment plots from 2015 to 2017 (Figure 1).

Biomass Assessment

Flowering rush aboveground biomass decreased by 40.5% by eight WAT in reference plots but recovered by 52 WAT ($p=0.0007$; Figure 2). Aboveground biomass of flowering rush in plots that received one diquat treatment was always lower than reference plot biomass and was unchanged at eight and 52 WAT which suggests a single diquat treatment was sufficient to maintain flowering rush biomass at low levels in these sites (p=0.0007; Figure 2). Aboveground biomass of flowering rush in plots treated with diquat twice was the same as reference plots at zero WAT, was reduced 100% at eight WAT compared to reference plots but had recovered to reference plot biomass levels by 52 WAT (p=0.0007; Figure 2).

Belowground flowering rush biomass was unchanged in reference plots at eight and 52 WAT (p=0.0496; Figure 2). Belowground biomass of flowering rush in plots treated once with diquat was 95.8% lower than reference plot biomass at zero WAT but the same as reference plot biomass at eight and 52 WAT (p=0.0496; Figure 2); there was no change in belowground flowering rush biomass in plots that received one diquat treatment at zero, eight, and 52 WAT (Figure 2). Belowground biomass of flowering rush in sites treated twice with diquat was not different from reference plot biomass at any time (Figure 2); however, by eight WAT, belowground biomass in these plots declined 96.7% from zero WAT levels and had recovered to zero WAT levels by 52 WAT (p=0.0496; Figure 2).

Flowering rush rhizome bud density remained unchanged in reference plots and plots that received a single diquat treatment eight and 52 WAT compared to zero WAT bud densities (Figure 2). Rhizome bud density of flowering rush in plots that received one diquat application was consistently lower than bud density of reference plots at zero, eight, and 52 WAT (88.9%, 98.0%, and 97.3%, respectively; p=0.0257; Figure 2) but remained unchanged in these plots over time. Flowering rush rhizome bud density in plots treated with diquat twice was the same as reference plot bud density at zero WAT, was reduced 86.5% at eight WAT compared to reference plots but recovered to reference plot densities by 52 WAT (p=0.0257; Figure 2). At eight WAT, rhizome bud density in plots that received two diquat applications was reduced by

88.3% of bud density in the same plots at zero WAT but had recovered to zero WAT density by 52 WAT (p=0.0257; Figure 2).

Flowering rush prevalence (Table 2), total species richness (Figure 1), flowering rush biomass, and flowering rush rhizome bud density (Figure 2) remained unchanged in reference plots 52 WAT which suggests these sites were at an ecological equilibria; from 2015 to 2017 only one species (coontail) declined in prevalence in these sites (Table 2). Sites that received either single or sequential diquat treatments did not exhibit a reduction in species richness over time (Figure 1) or flowering rush biomass or rhizome bud density 52 WAT (Figure 2) while flowering rush prevalence was decreased from 2015 to 2017 (Table 3).

Infested Area Assessment

Flowering rush infested 115.1 ha (284.5 ac) of lake bed in 2015 and 128.1 ha (316.6 ac) in 2016 and of these areas, 12 ha (29.6 ac) was used for reference sites while the rest were utilized as treatment sites. In 2015, there was one very low prevalence flowering rush site which covered 1.7 ha (4.1 ac) that was not treated, seven low prevalence sites of 48.7 ha (120.4 ac) which received one diquat treatment, and thirteen high prevalence sites of 52.8 ha (130.4 ac) that received two diquat treatments. In 2016, one very low prevalence flowering rush site that covered 8.1 ha (20.1 ac) of habitat that did not receive diquat treatments, ten low prevalence sites which covered 58.7 ha (145.0 ac), and fourteen high prevalence sites which covered 42.7 ha (105.4 ac) of habitat. From 2015 to 2016, there was a 6.5 ha (16.0 ac) increase in the amount of very low prevalence flowering rush habitat, a 9.9 ha (24.6 ac) increase in the amount of low prevalence infested areas, and a 10.1 ha (25 ac) decrease in high prevalence sites.

While overall area infested by flowering rush increased by 13.0 ha (32.1 ac) from 2015 to 2016, there was a 14.4% reduction in the proportion of high prevalence flowering rush sites ($p<0.0001$; binomial test; R Core Team 2020). There was a 5.4% increase in the proportion of very low prevalence sites ($p<0.0001$; binomial test; R Core Team 2020). There was no change in the proportion of low prevalence sites. Results of binomial tests suggest that the adaptive management protocol was converting high prevalence sites to low prevalence sites, and low prevalence to very low prevalence sites.

The protocol developed by Madsen et al. (2016a) would have required 3,860 L (1,019.6 gal) of diquat in 2015 and 3,978 L (1,050.8 gal) of diquat in 2016 to treat all of the flowering rush treatment sites. By utilizing an adaptive strategy, diquat use was reduced 25% to 2,886 L (762.4 gal) in 2015 and 34% to 2,617 L (691.6 gal) in 2016 when compared to the amount of diquat that would have been required by the previous protocol.

Prior to operational scale treatments, flowering rush infested over 80 ha (200 ac) of water in the Detroit Lakes (DL-Online 2020). At peak infestation, over 128 ha (316 ac) of flowering rush were being managed annually in the Detroit Lakes while in 2020 only 8 ha (20 ac) needed herbicide treatment (DL-Online 2020). Development of this adaptive management strategy was beneficial to resource managers as it allowed them to conserve management resources but not sacrifice management goals. This adaptive management protocol allowed for the further reduction of flowering rush prevalence in infested sites, did not allow flowering rush biomass or rhizome bud number to increase in infested sites, reduced overall diquat use by 25 to 34% in the Detroit Lakes system, and did not negatively affect the native plant community.

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TABLES AND FIGURES

Table 1. Action thresholds for adaptive management of flowering rush based on percent frequency of the plant in infested sites.

Frequency (%)	**Classification**	**No. Diquat Applications**	**Diquat Rate**
0-5	Very Low	0	NA
>5 to ≤20	Low	1	0.37 mg L^{-1}
>20	High	2	0.37 mg L^{-1}

Table. 2. Change in percent frequency of occurrence for plant species in non-treated reference plots in the Detroit Lakes system from 2015 to 2017; an ‘*’ indicates a statistically significant change in frequency of occurrence via Cochran-Mantel-Haenszel test and subsequent Fisher’s Exact test at the alpha=0.05 level of significance.

Common Name	**Scientific Name**	**2015 - 2016***	**2015 - 2017**
Flowering rush	*Butomus umbellatus* L.	15.7	-4.3
Coontail	*Ceratophyllum demersum* L.	22.9*	-17.4*
Chara	*Chara* L. spp.	-20.0*	-7.2
Water moss	*Drepanocladus* (Mull. Hal.) G. Roth spp.	4.3	-8.7
Elodea	*Elodea canadensis* Michx.	1.4	-1.4
Star duckweed	*Lemna trisulca* L.	0.0	7.2
Northern watermilfoil	*Myriophyllum sibiricum* Kom.	8.6	0.0
Nitella	*Nitella* C. A. Agardh spp.	4.3	-4.3
White waterlily	*Nymphaea odorata* Aiton	1.4	0.0
Yellow pondlily	*Nuphar lutea* (L.) Sm.	-4.3	-7.2
Curlyleaf pondweed	*Potamogeton crispus* L.	4.3	-14.5
Leafy pondweed	*Potamogeton foliosus* Raf.	-24.3*	5.8
Illinois pondweed	*Potamogeton illinoensis* Morong	-10.0	-2.9
Whitestem pondweed	*Potamogeton praelongus* Wulfen	20.0*	-1.4
Richardson's pondweed	*Potamogeton richardsonii* (Benn.) Rydb.	4.3	2.9
Robbin's pondweed	*Potamogeton robbbinsii* Oakes	-1.4	1.4
Flatstem pondweed	*Potamogeton zosteriformis* Fernald	-18.6*	10.1

White water buttercup	*Ranunculus longirostris* Godr.	-8.6	-4.3
Hardstem bulrush	*Schoenoplectus acutus* (Muhl. ex Bigelow) A. Love & D. Love	8.6	-1.4
Sago pondweed	*Stuckenia pectinata* (L.) Borner	22.9*	-2.9
Cattail	*Typha* L. spp.	2.9	-2.9
Common bladderwort	*Utricularia macrorhiza* Leconte	34.3*	1.4
Watercelery	*Vallisneria americana* Michx.	8.6	-5.8

Table 3. Change in percent frequency of occurrence for plant species in plots receiving one diquat treatment in the Detroit Lakes system from 2015 to 2017; an ‘*’ indicates a statistically significant change in frequency of occurrence via Cochran-Mantel-Haenszel test and subsequent Fisher’s Exact test at the alpha=0.05 level of significance.

Common Name	**Scientific Name**	**2015 - 2016***	**2015 - 2017**
Flowering rush	*Butomus umbellatus* L.	10.9*	-8.0*
Coontail	*Ceratophyllum demersum* L.	-1.1	13.8*
Chara	*Chara* L. spp.	-8.2	-9.2*
Water moss	*Drepanocladus* (Mull. Hal.) G. Roth spp.	-0.5	-5.7
Elodea	*Elodea canadensis* Michx.	-1.1	1.9
Star duckweed	*Lemna trisulca* L.	-10.3*	-4.2*
Northern watermilfoil	*Myriophyllum sibiricum* Kom.	12.0*	-3.8
Nitella	*Nitella* C. A. Agardh spp.	1.1	-1.1
White waterlily	*Nymphaea odorata* Aiton	0.0	0.0
Yellow pondlily	*Nuphar lutea* (L.) Sm.	-1.1	0.0
Curlyleaf pondweed	*Potamogeton crispus* L.	13.0*	-8.4*
Leafy pondweed	*Potamogeton foliosus* Raf.	-23.9*	-1.1
Variable pondweed	*Potamogeton gramineus* L.	-1.1	16.5*
Illinois pondweed	*Potamogeton illinoensis* Morong	-9.8*	-14.2*
Floating pondweed	*Potamogeton nataus* L.	0.0	-0.4
Whitestem pondweed	*Potamogeton praelongus* Wulfen	2.2	0.4
Richardson's pondweed	*Potamogeton richardsonii* (Benn.) Rydb.	-1.1	-1.9

Robbin's pondweed	*Potamogeton robbbinsii* Oakes	-1.1	0.0
Flatstem pondweed	*Potamogeton zosteriformis* Fernald	-1.6	0.8
White water buttercup	*Ranunculus longirostris* Godr.	-0.5	0.0
Hardstem bulrush	*Schoenoplectus acutus* (Muhl. ex Bigelow) A. Love & D. Love	0.0	0.8
Sago pondweed	*Stuckenia pectinata* (L.) Borner	26.1*	-11.5*
Common bladderwort	*Utricularia macrorhiza* Leconte	4.9*	2.7
Watercelery	*Vallisneria americana* Michx.	5.4	-12.3*

Table 4. Change in percent frequency of occurrence for plant species in plots receiving two diquat treatments in the Detroit Lakes system from 2015 to 2017; an '*' indicates a statistically significant change in frequency of occurrence via Cochran-Mantel-Haenszel test and subsequent Fisher's Exact test at the alpha=0.05 level of significance.

Common Name	**Scientific Name**	**2015 - 2016***	**2015 - 2017**
Flowering rush	*Butomus umbellatus* L.	-10.3*	-12.8*
Coontail	*Ceratophyllum demersum* L.	1.3	-2.1
Chara	*Chara* L. spp.	4.2	-5.8
Water moss	*Drepanocladus* (Mull. Hal.) G. Roth spp.	-1.3	-2.9
Elodea	*Elodea canadensis* Michx.	-0.3	0.8
Star duckweed	*Lemna trisulca* L.	3.5	-7.4*
Northern watermilfoil	*Myriophyllum sibiricum* Kom.	4.5	-16.0*
Nitella	*Nitella* C. A. Agardh spp.	0.0	0.4
White waterlily	*Nymphaea odorata* Aiton	0.0	0.0
Yellow pondlily	*Nuphar lutea* (L.) Sm.	-3.5	-1.2
Curlyleaf pondweed	*Potamogeton crispus* L.	10.0*	-2.5
Leafy pondweed	*Potamogeton foliosus* Raf.	-32.8*	2.1
Variable pondweed	*Potamogeton gramineus* L.	0.0	3.7*
Illinois pondweed	*Potamogeton illinoensis* Morong	0.0	-1.2
Whitestem pondweed	*Potamogeton praelongus* Wulfen	1.0	2.9
Richardson's pondweed	*Potamogeton richardsonii* (Benn.) Rydb.	4.5	-6.6
Robbin's pondweed	*Potamogeton robbbinsii* Oakes	-0.3	0.0

Flatstem pondweed	*Potamogeton zosteriformis* Fernald	-12.9*	2.1
Widgeongrass	*Ruppia cirrhosa* (Petagna) Grande	-0.3	0.0
White water buttercup	*Ranunculus longirostris* Godr.	-0.6	0.0
Hardstem bulrush	*Schoenoplectus acutus* (Muhl. ex Bigelow) A. Love & D. Love	-2.3	0.0
Sago pondweed	*Stuckenia pectinata* (L.) Borner	30.2*	-15.2*
Cattail	*Typha* L. spp.	-0.6	0.8
Common bladderwort	*Utricularia macrorhiza* Leconte	2.3*	-1.6
Watercelery	*Vallisneria americana* Michx.	18.6*	-12.3*

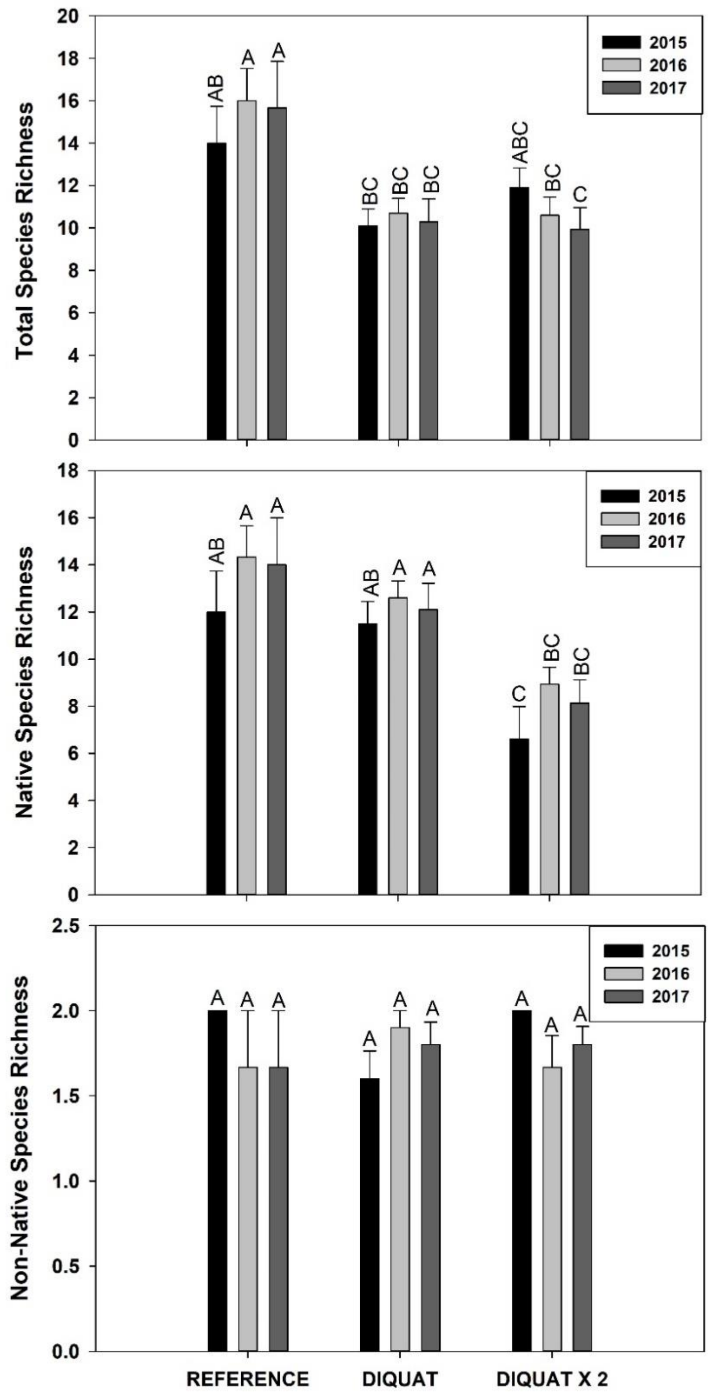

Figure 1. Mean total, native, and non-native species richness in reference and treated plots; bars sharing the same letter are not different at the alpha = 0.05 significance level; error bars are one standard error of the mean.

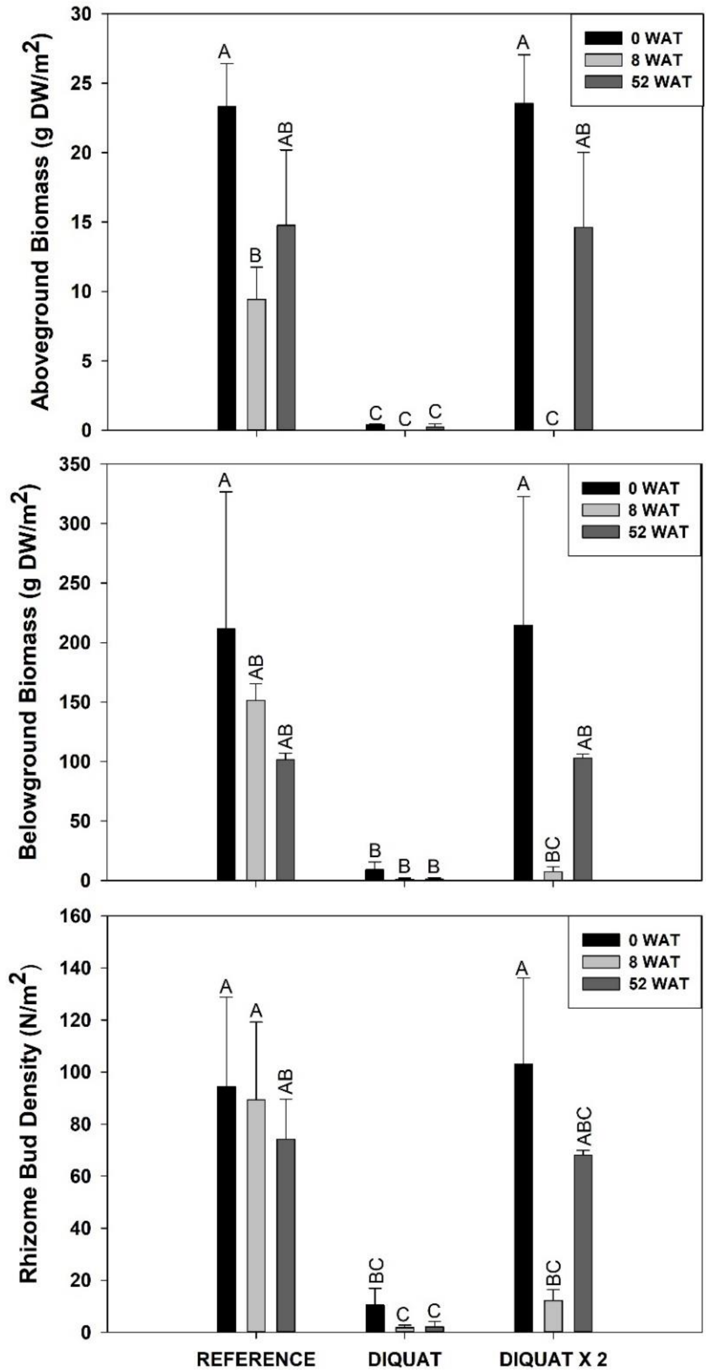

Figure 2. Flowering rush biomass and rhizome bud density in reference and diquat treated plots; bars sharing the same letter are not different at the alpha = 0.05 significance level; error bars are one standard error of the mean.