# Control of Curlyleaf Pondweed (*Potamogeton crispus*) with Endothall Herbicide Treatments and the Response of the Native Plant Community in Suburban Lakes

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JONATHAN DUANE JAKA

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ADVISOR: DR. RAYMOND M. NEWMAN

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# Dedication

I dedicate this thesis to my wife Meghan.  Her support during the past few years was the most important part of my success in graduate school.

# Comprehensive Abstract

Aquatic plants play an integral role in maintaining a stable aquatic ecosystem. Aquatic plants positively affect lake ecosystems by stabilizing shorelines, reducing sediment resuspension, and maintaining balanced water chemistry (i.e. taking up phosphorous or releasing oxygen). In a "healthy" aquatic ecosystem, native aquatic plants are often diverse and abundant. Invasive aquatic plants can disrupt delicate ecosystem equilibrium and become abundant to a level that negatively affects the diversity of the native plant community. Invasions of non-native plants such as curlyleaf pondweed (*Potamogeton crispus*) can have wide-ranging negative effects on whole lake ecosystems. Herbicide treatments have been shown to successfully control invasive aquatic plants during treatment years. Low-dose early-season endothall herbicide treatments have successfully controlled curlyleaf pondweed within treatment years while having no measurable negative affects on the native plant community, but extant experiments lacked pre-treatment data making accurate assessments of the effectiveness and specificity of endothall treatments difficult.

I evaluated the efficacy of endothall herbicide treatments in Lakes Riley and Susan, Carver County, Minnesota, USA using pre- and post-treatment data, as well as a non-treated reference lake. Endothall treatments were conducted in the spring of 2013 and 2014 in both lakes. I compared several years of pre-treatment data to data collected during the years of treatment and a reference lake to ascertain the magnitude to which treatments were successful. There were significant declines in curlyleaf frequency of occurrence (FO), biomass and turion production in both treatment lakes during treatment years. Treatments reduced peak curlyleaf FO to 31.0% of pre-treatment values in Lake

Riley and 24.5% of pre-treatment values in Lake Susan. Curlyleaf pondweed dry plant biomass was reduced to 2.2% and 1.2% of pre-treatment values respectively in Lakes Riley and Susan. Turion production was reduced to less than 1% of pre-treatment values in Lake Riley and 2.6% of peak pre-treatment values in Lake Susan. Turion density in the sediments declined significantly only in Lake Susan where the density decreased to 8.6% of pre-treatment values after treatments. Noteworthy declines in turion density in the sediments in Lake Riley were observed, but treatments only decreased turion density in the sediments to 46.7% of pre-treatment values. Curlyleaf FO declined moderately from 2013 to 2014 in the untreated reference Mitchell Lake, but not as drastically as the treatment lakes. There were no significant declines in biomass, turion production, or turion density in the sediments in Mitchell Lake.

I also evaluated the native aquatic plant response to the herbicide treatments. Native plant FO increased in several of the most commonly occurring plant species in Lakes Riley and Susan. In Lake Riley, five out of the six most commonly occurring plant species increased in FO in post-treatment years although not significantly. In Lake Susan, three of the most commonly occurring plant species increased significantly in frequency from pre- to post- treatment, although two species declined significantly in frequency. Coontail (*Ceratophyllum demersum)* declined significantly from pre-treatment to post-treatment years but because it is a native plant that can act as an invasive, the decline in Lake Susan may have been a positive change for the native plant community as a whole. Total mean native plant biomass lake-wide all native species did not change significantly in either study lake, however a significant native plant decline was observed in the same years in the reference lake. Of the most commonly occurring native plant species in Lake

Riley, three of six species had slightly higher biomass in 2014 when compared to pre-treatment years, but the biomass of native plants was quite low in Lake Riley with the exception of coontail and the invasive Eurasian watermilfoil (*Myriophyllum spicatum*). In Lake Susan, six of the eight native plant species were slightly but not significantly more abundant in 2014 compared to pre-treatment years. Eurasian watermilfoil declined significantly from pre-treatment to post-treatment years in Lake Susan, although the declines may be more likely related to high milfoil weevil (*Euhrychiopsis lecontei*) abundance. Finally, mean plant rake ratings and littoral-wide plant rake ratings (relative species abundance) were also higher for most of the native plants in both study lakes when comparing 2014 to pre-treatment years, however the values fluctuated from year-to-year.

Generally, native plants seemed to respond positively to reductions in curlyleaf pondweed in both treatment lakes. However increases in treatment lakes were often small and likely hindered by poor summer water clarity. Eurasian watermilfoil increased dramatically in Lake Riley while it decreased in Lake Susan. Additional monitoring (and potentially follow up spot treatments) may be necessary to reduce the likelihood of reinvasion of curlyleaf pondweed. Water clarity and Eurasian watermilfoil abundance (Lake Riley) are issues that likely need to be addressed to further enhance the native plant communities of Lakes Riley and Susan.

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# Prologue

This thesis is divided into three chapters. Chapter I is a literature review of the role of aquatic plants in a lake ecosystem, the importance of a native plant community, the negative effects of invasive aquatic plants with a focus on curlyleaf pondweed (*Potamogeton crispus*), and control of invasive aquatic plants with a focus on the herbicide endothall. The purpose of Chapter I was to detail the importance of the native plant community and demonstrate and discuss efforts to control aquatic invasive plants as a management strategy for removal of the invasive as well as an overall lake management strategy.

In Chapter II, I evaluate the efficacy of curlyleaf pondweed control, using low-dose early-season endothall herbicide, in two treatment lakes compared to an untreated reference lake. Treatments were conducted in two consecutive years in both study lakes and plant data were compared to pre-treatment years. I also evaluated the native plant community response to endothall herbicide treatments as well as the response to the reduction in curlyleaf pondweed abundance, biomass, and turion density in the sediments.

Lastly, Chapter III summarizes the whole thesis and brings the two chapters together. It compares my results to a few key papers that helped guide my research. It also highlights the key results from the study and relates them to management implications.

# Chapter I

## Literature Review of the Importance of the Aquatic Plant Community, Effects of Invasive Aquatic Plants, and Use of Endothall Herbicide for Curlyleaf Pondweed Control

## *Introduction*

Aquatic plants play an integral role in maintaining a stable aquatic ecosystem. Aquatic plants stabilize shorelines, reduce sediment resuspension, and maintain water chemistry (i.e. uptake nutrients or release oxygen) (Canfield et al. 1985, Barko et al. 1998, Blindow et al. 2006). Additionally, plants provide forage for waterfowl and invertebrates, shelter for algae-eating zooplankton and juvenile fishes, and cover for predatory fishes (Krull 1970, S□ndergaard et al. 1996, Valley et al. 2004). These ecosystem services are most effective and resilient to disturbance when a moderately abundant and diverse aquatic plant community exists (Blindow 1992, Valley et al. 2004, Zimmer et al. 2009). Invasion of nonnative aquatic plants (e.g. *Potamogeton crispus*) often leads to dense monotypic aquatic plant communities with cascading negative effects throughout the aquatic ecosystem and often removes ecosystem services previously described (Boylen et al. 1999, Alpert et al. 2000, Schultz and Dibble 2012). Management strategies using herbicides have been relatively successful in controlling some invasive aquatic plants (Poovey et al. 2002, Johnson et al. 2012). Less is known about the response of the native plant community after the successful control of invasive plants.

### Role of Aquatic Plants in Lakes

Aquatic plants uptake phosphorous from lakes, thereby reducing the intensity of algal blooms. Lakes with greater maximum depth of vegetation generally have better water clarity (Dennison et al. 1993, Scheffer et al. 1993, Scheffer et al. 2001). Some aquatic plants remove nutrients directly from the water column (Thomaz and Cunha

2010). Some plants obtain all necessary nutrients to maintain optimal growth rates from their above sediment foliage alone (Madsen and Cedergreen 2002). The amount of phosphorous released into the water column by living vascular plants is likely so minuscule as to not be biologically important (Carpenter and Lodge 1986, Graneli and Solander 1988, Madsen and Cedergreen 2002), especially when compared to the amount of phosphorous being absorbed from the water column by living plants. On average, aquatic plants act as a phosphorous sink (Carpenter and Lodge 1986). Additionally, plants provide a platform for periphyton production. Periphyton attached to plants also removes phosphorus directly from the water column and provides forage for snails and other invertebrates (Jones et al. 2000).

The decomposition of aquatic plants is an important source of dissolved organic carbon (DOC) in many lakes and supplements terrestrial inputs of carbon (Wetzel and Søndergaard 1998). The DOC in a lake is highly important in supporting the production of the aquatic microorganism community. Additionally, submersed vegetation often aids in the oxygenation of the water column (Carpenter and Lodge 1986), although floating leaf and emergent species actually may decrease dissolved oxygen within a dense stand (Frodge et al. 1990). Photosynthetically active rooted plants also oxygenate their roots and rhizome structures, subsequently diffusing oxygen into the upper layers of the sediment (Carpenter and Lodge 1986). However, in highly productive eutrophic lakes, sediments quickly take up oxygen released from plants and a highly oxygenated zone within the sediments is not present (Carpenter and Lodge 1986).

Aquatic plants can also positively affect an aquatic ecosystem through a more physical means. Sediment resuspension can be the dominant process causing high turbidity in some lakes (Horppila and Nurminen 2003). Resuspension of sediments reduces light availability creating a negative feedback loop in which the growth of rooted aquatic vegetation is further hindered (Madsen et al. 2001). However, once a rooted plant population is established, sedimentation rates can increase through a reduction in current velocity within the littoral zone of a lake (Madsen et al. 2001). Rooted aquatic plants also inhibit erosion, which also reduces the amount of sediment available to be resuspended in the lake (Madsen et al. 2001). Conceptually, there seems to be an optimal threshold for both light availability and sedimentation in terms of plant growth (Barko and James 1998).

Aquatic plants provide critical habitat and physical structure to an aquatic ecosystem. A complex and diverse aquatic plant community provides refuge for juvenile fishes, algae-eating zooplankton and many other invertebrates (Thomaz and Cuhna 2010). Aquatic invertebrates taking refuge in vegetation provide sustenance for juvenile fishes that may also be taking refuge in the vegetation (Valley et al. 2004). Great numbers of game fish and other native minnows are often associated with the presence of aquatic vegetation, whereas bullheads and non-native carp (rough fish) are often associated with turbid lakes. As rough fish, they are able to tolerate the lower dissolved oxygen levels often found in eutrophic lakes devoid of plants (Drake and Pereira 2002). Dodson et al. (2005) found that loss of submersed aquatic vegetation due to land use change, resulted in dramatically lower zooplankton densities and diversity in their study lakes. Aquatic vegetation is a key component in the life cycle of many aquatic organisms that require the existence of plants to feed, reproduce and maintain water quality.

Many aquatic macrophytes have positive direct and indirect effects on waterfowl (Knapton and Petrie 1999, Krull 1970, and Stafford et al. 2010). Whether the waterfowl forage directly on the macrophytes themselves or on the invertebrates dwelling within the macrophyte stand (Krull 1970), there is often a positive correlation between aquatic macrophytes and waterfowl abundance (Knapton and Petrie 1999). Several studies have observed declines in waterfowl populations associated with declines in macrophyte populations (Hansel-Welch et al. 2003, Bajer et al. 2009) and conversely increasing waterfowl population numbers with increasing abundance of aquatic macrophytes (Blindow 1992, Leschisin et al. 1992). Some plants that are not known to be useful forage to waterfowl may provide food indirectly by supporting abundant invertebrate populations, which may be especially important to juvenile waterfowl (Krull 1970). Aquatic plants have been found to be particularly important as forage for diving ducks that require the addition of body fat before their migrations in the spring and fall (Knapton and Petrie 1999). Because of the aquatic plants and the animals living amongst them, waterfowl populations often thrive when a healthy and diverse plant community is present.

**Importance of Native vs. Invasive Plant Community**

Invasions of nonnative aquatic plants often have deleterious effects on ecosystem services provided by native plant communities (Bolduan et al. 1994, Boylen et al. 1999). Aquatic systems with abundant native macrophytes are often associated with higher water quality and consequently receive higher scores on Indices of Biotic Integrity (IBI), which equate to a “healthier” or more desirable aquatic ecosystem (Beck et al. 2010). Managing for a diverse native plant community is one of the most effective ways to reduce the invasibility of a lake (Alpert et al. 2000). Diversity in invertebrate populations is associated with diverse native plant communities which leads to cascading positive effects throughout the food web to fish, waterfowl and other semi-aquatic organisms (Theel et al. 2008). A healthy native plant community increases ecosystem resiliency and thereby protects the ecosystem from invasion and other disturbances (Zimmer et al. 2009).

A diverse native aquatic plant community in a lake must include several species occupying a moderate amount of the littoral zone to most effectively provide habitat and services for all aquatic organisms relying on the ecosystem (Rosine 1955, Krull 1970, Jones et al. 2000, Valley et al. 2004, Schultz and Dibble 2012). Native vegetation creates a complex habitat that is more beneficial to aquatic organisms as opposed to a system dominated by a single invasive species. Several species of waterfowl were found to have higher populations correlated to higher plant species richness (Leschisin et al. 1992). An individual bird may forage on several different species of aquatic plant depending on the stage of its life cycle (Krull 1970, S□ndergaard et al. 1998). Similarly, native plant communities with higher diversity generally harbor higher densities and more diverse communities of algae-eating zooplankton and macroinvertebrates, which may benefit fish and water clarity. Communities with high diversity of plants, compared to monotypic invaded plant communities, afford greater opportunities for a more diverse and abundant community of macroinvertebrates, which benefits the entire food web (Scultz and Dibble 2012).

Fish predation success and growth rates have been shown to be higher in stands of native aquatic plants compared to stands in heavily invaded lakes (Schultz and Dibble 2012), and when aquatic plants exist at a moderate level (Valley et al. 2004). Native plants at moderate levels strike a delicate balance in providing cover for both juvenile fish and predator fish (Valley et al. 2004). When plants become too dense, often associated with monotypic invaded communities, both predator and prey fish decline (Valley et al 2004).

### *Invasive Aquatic Plants*

When invasive aquatic vegetation grows dense monocultures it often displaces native vegetation and may not provide the same services provided by native plants that are being replaced (Madsen 1997). Invasions of dense aquatic macrophytes can reduce quality fish habitat and light penetration as well as decrease dissolved oxygen concentrations and macroinvertebrate density and diversity (Schultz and Dibble 2012). Aside from their environmental affects, invasive aquatic macrophytes can have strong negative economic effects by lowering recreational value of a water body, clogging water intakes and reducing real estate values (Pejchar and Mooney 2009, Zhang and Boyle 2010).

Dense surface growth negatively affects recreational use of lakes and real estate values as well as produces a heavy economic burden for management. Invasive plants can strongly impede swimming and boating (due to prop fowling), potentially leading to regional economic losses. In a study in several Vermont lakes, researchers found that property values could be significantly lower with the presence of a thoroughly established Eurasian watermilfoil (*Myriophyllum spicatum*) plant population (Zhang and Boyle 2010). Invasive aquatic plant growth has reached such excessive levels as to clog water intake pipes to a point where power plants must be shut down (Chapman et al. 1974, Zedler and Kercher 2004). Removal of invasive aquatic plants from such water intake and irrigation canal sites can cost several millions of dollars a year (Lovell and Stone 2005).

An introduction of an invasive aquatic plant often leads to negative effects for macroinvertebrate, fish and waterfowl communities. Extremely dense vegetation can increase rates of survival for juvenile fishes creating an overpopulated stunted fish population (Nichols and Shaw 1986). Overpopulation of stunted fish can negatively affect macroinvertebrate populations through feeding pressures that are greater than the ecosystem can sustain. Low macroinvertebrate populations can have negative effects on foraging and resting waterfowl populations (Krull 1970). Additionally, invasive plants are often not favored by native waterfowl for direct forage (Stafford et al. 2010).

Studies by Michaelan et al. (2010) showed that an increasing invasive species was strongly correlated with a reduction in species richness in native plants in several study lakes. Invasive aquatic vegetation displaces native aquatic plants in multiple ways, such as by forming dense mats at the surface such that adequate light may not reach native plants growing beneath in the water column (Nichols and Shaw 1986, Madsen et al. 1991). Invasive plants like curlyleaf pondweed (*Potamogeton crispus*), with a novel life cycle, begin growing much earlier in the season reaching the surface before native plant communities begin to establish (Nichols and Shaw 1986, Bolduan et al. 1994). Native plants can be also rapidly displaced through genetic hybridization causing a lack of genetically pure native plants (Huxel 1999). Eurasian watermilfoil is known to hybridize with native northern watermilfoil (*Myriophyllum sibiricum*) in shallow lakes in the Midwest (Moody and Les 2002) and may become more resilient to chemical treatment (LaRue et al. 2012). Once established, invasive plants, like curlyleaf pondweed or Eurasian watermilfoil, tend to dominate lake ecosystems, making native plant recovery nearly impossible without management intervention.

**Curlyleaf pondweed**

Curlyleaf pondweed is an invasive aquatic perennial plant native to Eurasia, Africa and Australia (Catling and Dobson 1985). The plant has been present in the United States since the 1850's and the state of Minnesota since the 1910's and has established in nearly all of the lower 48 states of the United States (Nichols and Shaw 1986, MN DNR 2015). There is some disagreement in the literature about how it was originally introduced into the United States, but several specimens were collected near fish hatcheries and may have been spread through stocking activities (Bolduan et al 1994).

Curlyleaf pondweed (a Monocotyledon) is a major nuisance plant that forms dense monospecific stands and out competes and displaces native plants (Bolduan et al. 1994). It reproduces mainly via a vegetative reproductive bud known as a turion. Turion production begins early in the life cycle and continues until just prior to senescence when the turions fall from the plant and either germinate shortly thereafter or becomes buried in the sediment (summer dormancy) and likely remain viable until at least the following growing season (Catling and Dobson 1985). Turions that germinate in the fall result in small, low-light tolerant plants, which overwinter under the ice in Minnesota lakes, and grow rapidly in the early spring, often matting at the surface at a time when native submerged aquatic plants would normally be entering the beginning of their growth cycle (Bolduan et al. 1994). Although curlyleaf pondweed does flower and produce seeds, this reproductive pathway has been found to be negligible in its overall reproduction in its introduced range (Sastroutomo 1981, Poovey et al. 2002). Because curlyleaf pondweed possesses the capability to thrive in low-light conditions, it is often thrives in disturbed and turbid lake systems (Bolduan et al. 1994).

Dense curlyleaf populations can cause major issues with recreational use of lakes in Minnesota. Boat motor propellers can be fouled when trying to navigate dense stands of curlyleaf pondweed (Catling and Dobson 1985). Swimming also becomes difficult when curlyleaf becomes dense in near shore and swimming areas. Recreational use of lakes can be so strongly impeded by dense growth, such that lake use can be reduced significantly enough to have negative economic effects at a local scale (Catling and Dobson 1985). Invasions of non-native plants may also reduce the number of anglers as a result of a lower quality sport fishery (Pejchar and Mooney 2009).

The novel life cycle of curlyleaf pondweed gives it a strong competitive advantage over many native aquatic plant species. Curlyleaf exhibits rapid growth during early spring, when water temperatures begin to warm, allowing the plant to reach the surface as native plants have just begun to sprout (Nichols and Shaw 1986 and Bolduan et al. 1994). When the invasive plant reaches the surface and forms dense mats in the early season, native plants under the canopy are not able to receive the necessary amount of light needed for optimal growth. Typically, curlyleaf reaches its peak biomass in late spring or early summer compared to the peak biomass of native plants, which typically occurs in the middle or end of the summer (Bolduan et al. 1994, Johnson et al. 2012, and Jones et al. 2012). Senescence of curlyleaf pondweed generally occurs in early summer (Nichols and Shaw 1986). With the early senescence of curlyleaf, its decomposing biomass may result in a release of phosphorous into the water column that would be otherwise unavailable at this time if not for the presence of curlyleaf pondweed (Bolduan et al. 1994). Additional nutrients from decaying plant matter may contribute to phytoplankton production and foster degraded water quality and clarity (Rogers and Breen 1982, Bolduan et al. 1994, and Jones et al. 2012). The poor water clarity can perpetually foster an environment that is not conducive to the growth of native aquatic plants even though there is no longer a dense canopy of invasive plants shading out the native plants.

There are many negative effects of curlyleaf pondweed on native plant communities in shallow areas of lake systems (Catling and Dobson 1985, Nichols and Shaw 1986, Bolduan et al. 1994, Owens et al. 2007). Less studied is the response of a native plant community to the removal of curlyleaf. Curlyleaf can be controlled relatively well during treatment years with herbicide treatments (Poovey et al. 2002, Johnson et al. 2012, Jones et al. 2012). However, these studies did not have pre-treatment data regarding the native plant community. Without pre-treatment data, it's difficult to ascertain the degree to which treatments were truly effective and to measure the native aquatic plant response to said treatments.

### *Herbicide Treatments to Control Curlyleaf Pondweed*

Several herbicides, but primarily endothall and diquat, have been shown to successfully control curlyleaf pondweed during treatment years (Madsen et al. 2002, Poovey et al. 2002, Skogerboe and Getsinger 2002, Johnson et al. 2012). Diquat was found to be most effective at reducing shoot biomass at higher water temperatures, whereas endothall was more successful at lower temperature and lower doses (Poovey et al. 2002). Dose rate and water temperature play important roles in treatment timing when trying to enhance the native plant community. Endothall was chosen to treat my study lakes because it is effective at lower doses and lower water temperatures when native aquatic plants are typically not growing.

### Endothall

Endothall, dipotassium salt of endothall (7-oxabicyclo(2,2,1)heptane-2,3-dicarboxylic acid), is applied in liquid form as Aquathol® K. It is considered a broad-spectrum herbicide and is listed as effective against a wide variety of aquatic plant species both Monocotyledons and Dicotyledons (Skogerboe and Getsinger 2002). It is a contact herbicide that destroys plant tissue on contact as opposed to a systemic herbicide that is translocated through the plant (Skogerboe and Getsinger 2002). Endothall is suitable for treatment of curlyleaf pondweed because of its inherent ability to control curlyleaf at low concentrations and low water temperatures when native plants are not actively growing and therefore are not being affected by this herbicide (Poovey et al. 2002, Skogerboe and Getsinger 2002). Endothall was found to effectively control curlyleaf with application rates as low as 0.5 to 1.0 mg/L active ingredient (a.i.) (Skogerboe and Getsinger 2002). Effects on tested native aquatic plants were variable but 7 of the 9 native species tested did not demonstrate a significant decline six weeks after a treatment using an application rates up to 2.0 mg/L a.i. (Skogerboe and Getsinger 2002). Illinois pondweed (*Potamogeton illinoensis*), sago pondweed (*Stuckenia pectinata*), Eurasian watermilfoil, and curlyleaf pondweed were all found to be especially sensitive to endothall (Skogerboe and Getsinger 2002).

Timing of the application of endothall is very important in order to target only non-native species. Poovey et al. (2002) found that herbicide treatments at 16°C reduced curlyleaf biomass by 90% compared to an untreated reference, whereas treatments conducted in water temperatures of 23°C only reduced biomass by 60%. Conducting herbicide treatments in cooler water temperatures (spring or early season) affords an opportunity to selectively control for the invasive, leaving the native plant community relatively unharmed (Skogerboe and Getsinger 2002). Generally, native aquatic plants are not actively growing over temperature ranges at which endothall is most effective to control curlyleaf (10-16 ˚C) (Poovey et al. 2002).

Endothall has a relatively quick dissipation rate, which reduces its potential undesired effects on the native plant community. Yeo (1970) found that endothall concentrations between 0.5 to 1.0 mg/L dissipated to less than 50% of initial concentrations within 8 days in growth ponds that did not have a thermocline. The same concentrations were reduced to anywhere between 0% and 23% of initial concentrations at 12 days after treatment (Yeo 1970). Properly timed springtime herbicide treatments should have allowed the majority of herbicide to dissipate by the time native aquatic plants had begun actively growing.

A final factor that makes endothall a suitable candidate for curlyleaf pondweed control relates to the production of turions. Turions, which are produced as the plant reaches the surface, are the main reproductive structure of curlyleaf. Turions accumulate in the sediments and act as a seed bank for this species (Bolduan et al 1994). An early season endothall treatment can control curlyleaf pondweed before turions have been produced, thereby reducing production of viable turions and preventing accumulation of new turions in the sediments (Poovey et al. 2002, Johnson et al. 2012). This control method should aid in the management of curlyleaf pondweed in the following growing season by reducing available propagules.

***Summary***

Aquatic plant communities play a vital role in a healthy aquatic ecosystem. They directly and indirectly provide for a wide-array of ecological services affecting many aquatic and semi-aquatic organisms. Aquatic plants play an important role in ecosystem resiliency and the maintenance of a stable state. However, invasive aquatic plants, like curlyleaf pondweed, can severely disrupt ecosystem stability and resiliency. Curlyleaf pondweed often forms dense monotypic stands shading out native plants causing functional harm to the ecosystem and recreational aspects of a shallow lake system. Used in low-dose early-season treatments, endothall herbicide has been shown to be effective at selectively controlling curlyleaf pondweed while leaving native aquatic plants unharmed.

Recent studies have shown successful control of curlyleaf pondweed, while avoiding significant declines in the native plant community (Johnson et al. 2012, Jones et al. 2012). However, these studies did not have pre-treatment data, making it difficult to ascertain the magnitude at which treatments were successful at reducing curlyleaf and to determine whether treatments were able to enhance the native plant community or simply not harm it. In my research, I aimed to determine the degree to which curlyleaf pondweed could be controlled in a lake and to determine the native plant community response to herbicide treatments by comparing native plant communities before and after herbicide treatments.

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Chapter II

Efficacy of Curlyleaf Pondweed Herbicide Treatments and Native Aquatic Plant Response in Minnesota Suburban Lakes

### *Introduction*

A native aquatic plant community contributes to the overall function of a "healthy" lake aquatic ecosystem. A diverse aquatic plant community provides forage and shelter for invertebrates, fish, waterfowl and other semi-aquatic organisms (Valley et al. 2004). Aquatic plants help maintain water clarity and water quality by maintaining balanced water chemistry, inhibiting sediment resuspension and providing refuge for algae-eating zooplankton (von Donk and van de Bund 2002, Horppila and Nurminen 2003, Dodson et al. 2005). Diverse native aquatic plant assemblages provide the most opportunity for a wide range of ecosystem services and increase a system's resiliency (Dokulil et al. 2011). Invasive species such as curlyleaf pondweed (*Potamogeton crispus*) can become the dominant species in an ecosystem and diminish many of the previously discussed ecosystem services.

Curlyleaf pondweed is an invasive aquatic plant, introduced to North America from Eurasia, which can grow dense monotypic stands that displace many native species (Bolduan et al. 1994). Its reproduction is primarily vegetative via hardy propagules known as turions (Bolduan et al. 1994). In the northern United States, turions are typically produced in the spring and drop from the plant just before its senescence (Caitling and Dobson 1985). Curlyleaf that sprouts from turions in the fall, can grow slowly through the winter under ice cover and rapidly grow to the lake surface after ice out in early spring before much of the native plant growth begins (Bolduan et al. 1994). Turions that do not sprout in the same year they are produced can remain viable in a state of dormancy in sediment for two or more years (Nichols and Shaw 1986, Woolf and Madsen 2003). The novel life cycle of curlyleaf pondweed provides a competitive advantage over many native species, often leading to a displacement of the native plant community.

In my study, two lakes (Lake Riley (DOW# 10-000200) and Lake Susan (DOW# 10-001300); Carver County, Minnesota) had varying levels of curlyleaf pondweed infestations and were chosen as herbicide treatment lakes. A third untreated reference lake (Mitchell Lake, DOW# 27-007000, Hennepin County, Minnesota), also with relatively high abundance of curlyleaf pondweed, was selected as a reference. The treatment lakes had relatively low abundance and diversity of native aquatic plants and relatively high abundance of curlyleaf pondweed during the years previous to treatment compared to other lakes in the watershed. Curlyleaf pondweed had been observed increasing in frequency of occurrence (FO), biomass, and turion density in the sediments in both treatment lakes. Steady or dramatic increases of curlyleaf pondweed abundance and biomass may have deleterious effects on the native plant communities (Owens et al. 2007).

Endothall has been shown to reduce the biomass and turion production of curlyleaf pondweed in mesocosm tank studies at low doses (Poovey et al 2002, Skogerboe and Getsinger 2002). Additionally, curlyleaf pondweed frequency, biomass, and turion abundance have all been significantly reduced during treatment years using low-dose rates of endothall on a lake-wide scale (Johnson et al. 2012). Jones et al. (2012) found that treatments conducted in the same lakes did not have negative effects on native aquatic plants. However, the previous studies by Johnson et al. (2012) and Jones et al. (2012) had very little or no pre-treatment data to compare to treatment years. With little pre-treatment data it is difficult to ascertain the magnitude at which treatments were successful at controlling curlyleaf and to determine whether or not native plant communities were enhanced with curlyleaf pondweed control or simply unharmed. My study included several years of pre-treatment data.

Endothall herbicide treatments were used to control curlyleaf pondweed as a management tool to reduce the presence of the invasive species and to enhance the native plant communities in the treatment lakes. Low-dose endothall herbicide treatments in the early spring were chosen as the treatment method for this study because (1) treatments conducted in early spring cool waters would reduce the adverse effect on the native plant communities; (2) spring treatments target curlyleaf before most turions are produced, thereby reducing turion density in the sediments; and (3) endothall is effective at low-dose rates and dissipates relatively quickly compared to other herbicides (Yeo 1970). The effective removal of curlyleaf pondweed may produce an open niche that can be filled by native plant communities. The removal of curlyleaf may also allow greater water clarity to persist longer into the summer (Bolduan et al. 1994). Dense stands of decaying curlyleaf pondweed may add considerable nutrients to the water column when the plant senesces in early summer (Bolduan et al 1994).

In my study, the goal was to determine whether or not we can enhance native plant communities by controlling abundant populations of curlyleaf pondweed through early-season endothall herbicide treatments. The specific objectives of this study were to determine (1) the magnitude of the effect of herbicide treatments on curlyleaf pondweed; (2) if reductions of curlyleaf pondweed abundance, biomass and turion densities in the sediment will allow the native plant community to expand; and (3) how native plants respond to curlyleaf pondweed herbicide treatments.

### *Methods*

### Study Lakes

Three study lakes were chosen within the Riley Purgatory Bluff Creek Watershed in Carver and Hennepin Counties Minnesota, USA. Lakes Riley (DOW# 10-0002-00) and Susan (DOW# 10-0013-00) were chosen as treatment lakes and Mitchell Lake (DOW# 27-0070-00) as a reference. Lake restoration efforts have been ongoing in Lakes Riley and Susan since 2009. Lakes Riley and Susan underwent common carp (*Cyprinus carpio*) removal in 2009 and 2010 as part of lake restoration efforts (Bajer and Sorensen 2015). Carp levels have remained well below ecosystem-damaging thresholds since their removal in both lakes (P. Bajer, Personal Communication, 2014, Bajer and Sorensen 2015). Additionally, Lake Susan had six native plant species (*Chara spp., Myriophyllum sibiricum, Naja flexilis, Potamogeton zosteriformis, Valisneria americana, and Zosterella dubia*) transplanted into the lake in 2009, 2010 and 2011 to promote native plant revegetation (Knopik 2014) (see Table 1 for common names). Four out of the six native plant species that were transplanted into Lake Susan were still found at transplant sites in 2014 (see Appendix Figures 7 and 8), although transplanted species were rarely found during point intercept surveys.

Lakes Riley and Susan were chosen as treatment lakes because they had nuisance level abundance of curlyleaf pondweed that was increasing. Additionally, several years (2009 to 2012 in Lake Susan and 2010 to 2012 in Lake Riley) of plant data collected prior to herbicide treatments were available for use in the analysis. Due to increasing curlyleaf frequency and biomass (Figures 1 and 2), treatments were conducted in May 2013. Both lakes underwent lake-wide herbicide treatment in May 2013 and again in May 2014. Mitchell Lake was chosen as an untreated reference lake due to a high abundance of curlyleaf pondweed and no herbicide treatments previously or planned for the near future. Mitchell Lake is classified as a "Natural Environment Lake" by the Minnesota Department of Natural Resources (MN DNR) and as such had not been treated with chemical herbicides. Surveys were conducted in Mitchell Lake in 2013 and 2014. Lakes Riley and Susan are generally considered eutrophic to hyper-eutrophic whereas Mitchell Lake is mesotrophic (MN PCA). Lake area ranged from about 36 ha in Lake Susan to 120 ha in Lake Riley and the littoral zone area ranged from about 83% in Lake Susan to 38% in Lake Riley (MN DNR) (Table 2). All study lakes had relatively highly developed shorelines with many homes on the shorelines.

Three point intercept surveys were conducted per year in each lake. The first survey was conducted immediately before the herbicide treatments took place in May. A second survey was conducted several weeks after the treatments and was meant to capture the peak of curlyleaf pondweed biomass in June. Finally I conducted a point intercept survey in August, which was meant to capture the peak biomass of native plants in the study lakes. Results discussed in this thesis only discuss the most commonly occurring plants in the study lakes. A complete list of all plants observed during all surveys can be found in the appendix.

**Herbicide Treatments**

I began monitoring water temperatures in the early spring. When water temperatures were about 10 °C, I delineated dense areas of curlyleaf pondweed in both treatments lakes. Dense areas of curlyleaf were identified and confirmed in three ways: (1) by throwing a 14-tine double-headed garden rake from a boat periodically to determine presence or absence of curlyleaf pondweed (to determine edges of dense stands); (2) by monitoring vegetation with a Lowrance HDS 5 Gen2

Fishfinder/Chartplotter coupled with a Lowrance StructureScan HD Sonar Imaging System; and (3) reviewing curlyleaf pondweed delineations from previous years. Delineation shapefiles were created by navigating a boat around dense areas of curlyleaf pondweed while recording the track on a Garmin GPSmap76 Global Positioning System (GPS). The track was then uploaded to a personal computer using DNRGPS Version 6.1.0.5 software and I converted the data into a shapefile containing polygons with the densest areas of curlyleaf pondweed (to be the treatment area) using ArcGIS Desktop 10 Service Pack 5 software. Delineation shapefiles were sent to an herbicide applicator contractor to complete the treatments. The applicators conducted herbicide treatments in water temperatures between 10 and 16 °C in Lakes Riley and Susan in an effort to protect native plants and control curlyleaf pondweed before turion production began. Lake-wide early-season endothall treatments took place in both lakes in May 2013 and May 2014.

Lakes were treated with Aquathol K®, 40.3% dipotassium salt of endothall (7-oxabicyclo[2,2,1]heptane-2,3-dicarboxylic acid), at a target dose rate of 0.75 to 1.0 mg/L of the active ingredient (a.i.). Applications were conducted from a boat using depth adjustable application drop hoses to maximize the amount of the herbicide interacting with the curlyleaf and minimize drift out of treatment areas. The application boat followed a predetermined route programmed into a GPS, which was based on the delineation shapefiles that I provided. Endothall herbicide applications were conducted by Lake Restoration Inc. in Lake Riley on 7 May 2013 and Lake Susan on 3 May 2013. Treatments were conducted by PLM Lake and Land Management Corp. in Lake Riley on 20 May 2014 and in Lake Susan on 16 May 2014.

**Macrophyte Frequency of Occurrence**

The frequency of occurrence (FO) of native macrophytes and curlyleaf pondweed were obtained concurrently during point intercept surveys. The point intercept method was used to survey aquatic vegetation in all the study lakes (Madsen 1999). Surveys were created to have at least 120 survey points within the littoral zone (≤4.6m), although I surveyed outside the littoral zone as well to ascertain the maximum depth of plant growth. Points were randomly generated using ArcMap GIS software for each lake. After a shapefile of random points was generated in ArcMap, the points were transferred into the DNRGPS software and then loaded into a Garmin GPSmap76 GPS. The handheld GPS was used in the field to navigate a boat to each survey point.

At each survey point, a weighted double-headed garden rake attached to a rope was tossed and allowed to sink to the lake bottom. The rake (0.3 m wide) was dragged along the lake bottom for approximately 3.3 m, therefore sampling approximately one square meter at each point. Rake densities, species presence and relative density, and depth were recorded at each survey point. Plants were sorted to species in the field to obtain presence or absence at each point. The FO was calculated for an effective littoral zone by dividing the total number of sites sampled at a depth of ≤ 3.4 m by the number of occurrences for each plant species at a depth of ≤ 3.4 m. A depth of 3.4 m (rather than 4.6 m) was chosen based on the occurrence of the majority of the plants in the treatment lakes, in an attempt to better represent what was happening to the plant community during the study. The depth of 3.4 m was selected by comparing the depths at which plants were present, in all surveys and in both treatment lakes, and selecting the maximum depth in which 95% of the plants were present.

## Rake Density Ratings

Rake density ratings were determined at each of the point intercept survey points. When the double-sided garden rake was retrieved, a relative density rating of zero to five was given to the whole rake density based on the density of all plants on the rake. The plants were then sorted to species in the field and each plant species present was given its own rating of one to five based on each individual species' density. Rake density ratings were used in two separate calculations. Mean plant rake ratings (PRR) were calculated at points only where plants occurred. Mean PRR values only include values one through five. This method was used to observe changes in plant densities where plants were growing, with no correlation to FO. Additionally, relative rake ratings were used to calculate a mean littoral-wide plant rake rating (LWPRR) for the whole littoral area (≤3.4 m). This value was calculated by averaging all rake ratings (zero to five) throughout the littoral area, including areas with no plants that were assigned a zero. This calculation takes into consideration changes in density associated with FO throughout the littoral area.

## Macrophyte Biomass

Plant biomass sampling took place concurrently during point intercept surveys. I used a randomly selected subset of 80 point intercept sample points as biomass sampling points. A single headed garden rake was used to sample an area 0.3 m wide for a total survey area of 0.09 $m^2$. The garden rake, attached to adjustable length pole capable of reaching depths of 4.6m, was lowered to the lake bottom and rotated three times to ensure uprooting of plants (Johnson and Newman 2011). The rake was slowly rotated upon retrieval to make sure that all macrophytes were maintained on the rake. Biomass samples were placed in sealable plastics bags and stored in a cooler for transport to the laboratory.  At the laboratory, samples were stored at 5°C until they were processed and sorted.

Samples were rinsed to remove sediment and excess debris and spun in a salad spinner to remove excess water prior to drying.  Samples were sorted by individual species and placed into paper bags that had been previously weighed.  While being sorted, any remaining root material was removed from individual plants and wet weight biomass was recorded.  The bagged plants were dried for at least 48 h at 105°C and reweighed.  Plant biomass was calculated as grams dry per square meter (g dry/$m^2$) by dividing the dry sample mass by the total sample area (0.09$m^2$).  Mean lake-wide littoral plant biomass was determined by taking the mean of all samples from depths of $\leq$ 3.4 m for each individual plant species.

**Turion Production**

Turions found in curlyleaf biomass samples were removed during the initial processing of biomass samples in the laboratory.  Turions were dried, enumerated and weighed separately from the remainder of the curlyleaf biomass.  Turion production per site was calculated by dividing the number of turions in a sample by the sample area.  I calculated the lake-wide mean turion production by averaging the turion production per site for all biomass sites in a given survey as turions/$m^2$.  Turion production was only calculated in the spring and early summer, as curlyleaf turions are rarely found late in the summer (Johnson et al. 2012).  I aimed to capture peak curlyleaf turion production, which tends to occur earlier in the growing season (Nichols and Shaw 1986, Bolduan et al. 1994, Johnson et al. 2012).

## Turion Density Sampling

To quantify the turion densities in the sediment and thus quantify the curlyleaf propagule bank, sediment samples were collected in October each year. Forty sampling sites were randomly selected from a subset of the point intercept survey points. Water depth and substrate type were recorded at each sampling point. A petite ponar (sample area = $0.0232m^2$; sample depth ~10 cm) was lowered from a boat at each sampling point and a sediment sample was collected (Johnson et al. 2012). The sediments retrieved by the ponar were passed through a 1 mm sieve to remove excess debris. Sprouted turions were counted and then discarded in the field once they had been enumerated and recorded. The remaining samples, including unsprouted turions, gravel and other organic material, were placed in a sealable plastic bag and stored in a cooler for transport back to the laboratory. Samples were again sieved at the laboratory in order to separate all turions from the remainder of the sample debris. Unsprouted turions were removed, counted and placed in sealable plastic bags with non-chlorinated well water and stored in a light free environment at a temperature of 5° C. Turion density in the sediments (turions/$m^2$) was derived by dividing the total number of turions at a given site by the sample area ($0.0232m^2$) (Johnson et al. 2012). Lake-wide littoral zone mean turion density was derived by averaging the number of turions at all 40 sampling sites.

## Turion Viability Analysis

Turion viability analyses were conducted in the laboratory with the unsprouted turions collected from the sediment turion density samples (Johnson et al. 2012). Unsprouted turions were maintained in a light free environment at 5° C for a period of about 30 days (cold dark incubations), at which time the turions were moved to an environment under a natural light spectrum for 12 hours a day at a temperature of approximately 20° C for another of 30 days (warm light incubations). Turions were scrutinized every 7 days during cold and warm incubations; if any turions had sprouted they were enumerated, recorded, and discarded. At the conclusion of the 60-day incubation period, turion viability was calculated by dividing the number of sprouted turions by the total number of turions found in a given sample. Total lake-wide turion viability was calculated by averaging the viabilities of samples containing turions (Johnson et al. 2012).

**Assessment of Water Quality**

During most surveys and intermittently throughout the field season, a set of water quality indicators were measured and recorded in the study lakes at the deepest part of the lakes. Dissolved oxygen (DO), temperature, and photosynthetically active radiation (PAR) were measured at intervals of 0.5 m until the bottom of the lake or a depth of 10 m. Dissolved oxygen and temperature were measured using a YSI 50B electronic meter and recorded in mg/L and °C respectively. Values for PAR were measured using a Li-Cor Li-189 Light Meter and a Li-Cor underwater quantum sensor and recorded in µmol photons/s/$m^2$. The depths at which 5% of light levels from surface remained were used to compare between years. Secchi depths (nearest 0.1 m) were also recorded during water quality assessments and compared between years.

**Statistical Analysis**

All statistical analysis was conducted using R statistical software version 2.15.2 (The R Foundation for Statistical Computing, 2012). Results were considered statistically significant when p values were $< 0.05$. An analysis of variance (ANOVA) was used to detect significant differences between years in dry plant biomass for both native aquatic plants and curlyleaf pondweed as well as difference in mean rake ratings, turion density, and turion production between years. A Tukey honest significant difference (HSD) test was used to determine whether or not differences between years were significant. To compare differences in pre-treatment years and post-treatment years for FO of native plants and curlyleaf pondweed, a chi-squared test was conducted using the previous year's data as expected values.

*Results*

**Water Quality Parameters**

Water clarity generally followed common eutrophic lake water clarity patterns (i.e. higher water clarity in the spring declining to relative poor clarity in the summer) with some annual variation in both treatment lakes (Table 3). Mitchell Lake is designated a mesotrophic lake and as such had better summer water clarity than Lakes Riley and Susan.

Lake Riley water clarity was observed at its highest in the spring of 2014 with a 6.6 m Secchi depth and a 6.5 m depth at which 5% of PAR remained (Table 3). Summer water clarity was also highest in 2014, with a Secchi depth of 1.3 m and a depth of 1.8 m at which 5% PAR remained in early July. Average summer Secchi depth for all years observed was 1.2 m and average summer depth at which 5% of PAR remained was 2.0 m in Lake Riley.

Lake Susan water clarity was observed at its highest in the spring of 2011 with a Secchi depth of 5.0 m (this depth represented the bottom of the lake) and a depth at which 5% of PAR remained of 5.0 m (Table 4). A similarly high clarity was observed in the spring of 2014. Summer water clarity was highest in 2011, with a Secchi depth of 2.3 m and a depth at which 5% of PAR remained of 3.0 m. Lake Susan average summer Secchi depth for all years observed was 1.0 m, with an average summer depth with ≥ 5% of PAR remaining of 1.5 m.

Mitchell Lake water clarity was observed at its highest in the spring of 2013 with a Secchi depth of 3.4 m and the depth at which 5% PAR remained was 3.8 m (Table 5). Summer water clarity was also observed at its highest in 2013 with a Secchi depth of 2.5

m and a depth at which 5% of PAR remained of 3.0 m. Average summer Secchi depth was 1.6 m and average summer depth of which 5% of PAR remained was 2.1 m in Mitchell Lake.

**Success of Curlyleaf Pondweed Control as an Outcome of Herbicide Treatments**

Overall curlyleaf pondweed control was successful in both treatment lakes during the treatment years. Curlyleaf FO and biomass declined significantly ($p < 0.05$) from pre-treatment to post-treatment years in both lakes (Figures 1 and 2). Turion production also declined significantly in both lakes, whereas turion density in the sediments declined significantly in Lake Susan only, however noteworthy declines were observed in Lake Riley as well.

*Curlyleaf Pondweed Frequency of Occurrence*

Data collected in years prior to treatment in Lakes Riley and Lake Susan showed an increase in curlyleaf pondweed FO through 2012 (Figure 1). Curlyleaf pondweed FO was dramatically lower in post-treatment years in both Lakes Riley and Susan. Curlyleaf peak FO decreased significantly ($p<0.05$) in both treatment lakes. Changes in curlyleaf peak FO were significant when comparing 2012 to 2013 and 2012 to 2014, whereas frequencies of occurrence were similar from 2013 to 2014. The peak FO observed in the untreated reference Mitchell Lake, decreased moderately from 2013 to 2014, however FO values were notably higher than the treatment lakes. Treatments reduced the FO of curlyleaf to 31.0% of peak pre-treatment values in Lake Riley and 24.5% of pre-treatment values in Lake Susan (Figure 1).

*Curlyleaf Pondweed Biomass*

Prior to treatment in 2013, Lakes Riley and Susan generally showed an increase in curlyleaf pondweed biomass through 2012 (Figure 2). Curlyleaf pondweed peak dry plant biomass decreased significantly ($p<0.05$) in both Lakes Riley and Susan after treatment. Declines in biomass were significant when comparing 2012 to 2013 and 2012 to 2014 for both lakes. Curlyleaf pondweed peak biomass in the untreated reference Mitchell Lake biomass declined slightly (from 29.7 ± 7.1 g dry/$m^2$ to 14.9 ± 5.1 g dry/$m^2$ (mean ±1 SE, used henceforth)), but the change was not significant. Treatments reduced curlyleaf biomass in Lake Riley to 2.2% of pre-treatment levels and to 1.2% of pre-treatment values in Lake Susan (Figure 2). Curlyleaf biomass was similar in 2013 and 2014 in the treatment lakes.

*Curlyleaf Pondweed Turion Production*

Curlyleaf pondweed turion production also declined significantly post herbicide treatment in both treatment lakes (Figure 3). Declines in peak turion productions were significant when comparing 2012 to 2013 and 2012 to 2014 in both lakes Riley and Susan. Lake Riley turion production declined to less than 1% of pre-treatment levels in 2014. In the same year, turion production in Lake Susan declined to 10.4% of the level observed in 2012 and turion production (in 2014) was 2.6% of production levels observed in 2011. Peak curlyleaf turion production in Lake Susan after treatment was 24.9 ± 13.6 turions per $m^2$ in 2012 (prior to treatments) although production was as high as 98.2 ± 50.4 turions per $m^2$ in 2011. Turion production levels were not significantly different when comparing 2013 to 2014 in either lake. Turion production did not change significantly in the untreated Mitchell Lake from 2013 (139.7 ± 50.1 turions per $m^2$) to 2014 (76.1 ± 30.4 turions per $m^2$).

*Curlyleaf Pondweed Turion Density in the Sediments*

In the years prior to treatment there was an increase in turion density in the sediments in both Lakes Riley and Susan (Figure 4). Noteworthy declines in sediment turion densities were observed in both treatment lakes in post-treatment years. Lake Susan turion density in the sediments decreased significantly from 2012 to 2014, but there were no significant declines in Lake Riley. Lake Riley 2014 turion density in the sediments values were 46.7% of values observed in 2012 and Lake Susan declined to 8.6% of pre-treatment levels during the same time. There were no significant changes observed in turion densities from 2013 and 2014 in either treatment lake. Turion density in the untreated Mitchell Lake did not significantly change from 2013 to 2014 when turion densities were 193.4 ± 43.9 turions per $m^2$ and 163.8 ± 33.5 turions per m2 respectively.

*Turion Viability Analysis*

Average turion viability per site also declined notably in both study lakes in treatment years. Turion viability was at or above 90% in pre-treatment years in both treatment lakes and declined dramatically post-treatment (Table 6 and 7). Mean viable turion density in the sediments declined dramatically as well in both lakes. Similar scale declines were noted in the untreated Mitchell Lake, where turion viability and mean viable turion density also declined dramatically from 2013 to 2014, along with declines in viable turion density in the sediments (Table 8).

## Native Plant Community Response to Curlyleaf Pondweed Control

### *Native Plant Frequency of Occurrence*

The mean FO of all pre-treatment years (2009 through 2012 in Lake Susan and 2011 and 2012 in Lake Riley) was compared to the mean of post-treatment years (2013 and 2014). Native plant post-treatment FO values were variable when compared to pre-treatment data. Of the most commonly occurring native plant species in Lakes Riley, the majority of the taxa (five out of six) increased from pre- to post-treatment years, however no plant species increased or decreased significantly (Figure 5). Coontail decreased slightly from pre-treatment years to post-treatment years. Canada waterweed, bushy pondweed, white water lily, narrow leaf pondweed, and sago pondweed all increased slightly. With the exception of coontail, these plant species increased by an average of 1.7% and were all observed at frequencies of less than 10% during all years. The invasive Eurasian watermilfoil increased significantly in Lake Riley in post-treatment years.

Of the most commonly occurring native species in Lake Susan, half of the taxa (four out of eight) increased (Figure 6). Three native plants (Canada waterweed, bushy pondweed, and American lotus) increased significantly in Lake Susan from pre-treatment to post-treatment years, however two natives (coontail and narrowleaf pondweed) decreased significantly. White water lily also increased slightly from pre- to post-treatment years, while yellow water lily and sago pondweed deceased slightly. Eurasian watermilfoil declined significantly post-treatment.

Of the most frequently occurring native species in Mitchell Lake, one species (flat-stem pondweed) increased significantly from 2013 to 2014 and no species significantly declined (Figure 7). Four species slightly increased and three species slightly declined in Mitchell Lake. With the exception of flat-stem pondweed, FO values were generally similar in Mitchell Lake when comparing 2013 to 2014.

*Native Mean Plant Rake Ratings*

Mean rake ratings were variable in Lake Riley throughout all years. The mean PRR (only where plants exist) of one native species (sago pondweed) increased significantly in Lake Riley from pre-treatment years to 2014. Overall, mean PRRs in Lake Riley were higher in 2014 compared to pre-treatment years, but the values fluctuated dramatically from pre-treatment years to 2013 and 2014 (Table 9). Canada waterweed and narrowleaf pondweed mean PRRs both increased significantly from pre-treatment years to 2013, however they declined significantly from 2013 to 2014. The mean PRR for Eurasian watermilfoil was significantly higher in 2014 compared to pre-treatment years and 2013.

Lake Riley mean LWPRR values (all sites sampled) were more variable when compared to Lake Riley PRR values. Similar to the PRR values, sago pondweed mean LWPRR increased significantly from pre-treatment years to 2014. The whole rake (all species combined) LWPRR decreased significantly from pre-treatment to 2013 but increased significantly from 2013 to 2014, and the whole rake LWPRR value in 2014 was similar to pre-treatment values. Generally, individual mean native LWPRRs were higher in 2014 compared to pre-treatment values, coontail being the only exception (Table 10). The mean LWPRR values for the majority of species in Lake Riley were higher in 2014 compared to pre-treatment values. Eurasian watermilfoil mean LWPRR increased significantly from pre-treatment values to 2014.

All mean PRR values for Lake Susan were higher in 2014 than in pre-treatment years (Table 11). Mean PRRs were significantly higher for four native species (American lotus, bushy pondweed, Canada waterweed, and white water lily) along with the whole rake rating in 2014 compared to pre-treatment years. Bushy pondweed, narrow leaf pondweed, and yellow water lily mean PRRs increased significantly from pre-treatment to 2013, but decreased significantly from 2013 to 2014. Eurasian watermilfoil mean PRR increased significantly from pre-treatment years to 2014.

Lake Susan mean LWPRR values were considerably more variable than PRR values. Six values were significantly different in 2014 compared to pre-treatment values (Table 12). Bushy pondweed and Canada waterweed mean LWPRRs increased significantly in 2014 compared to pre-treatment values. However, the whole rake mean LWPRR as well as coontail and narrowleaf pondweed mean LWPRRs decreased significantly in 2014 compared to pre-treatment years. Coontail decreased significantly from pre-treatment values to 2013, then increased from 2013 to 2014. In contrast to the Eurasian watermilfoil mean PRR, mean LWPRR decreased significantly in 2014 compared to pre-treatment years.

In Mitchell Lake, the mean PRR of star duckweed and white water lily decreased significantly from 2013 to 2014. Four out of the nine most commonly occurring native plant species mean PRRs increased slightly from 2013 to 2014 (Table 13). Similar to the mean PRR value, star duckweed mean LWPRR value decreased significantly from 2103 to 2014. The whole rake mean LWPRR value also decreased significantly, while flat-stem pondweed increased significantly during the same time. Generally, other mean LWPRR values in Mitchell Lake were similar in 2013 and 2014 (Table 14).

*Native Plant Biomass*

Total (all species combined) native dry plant biomass did not increase or decrease significantly after treatment in either treatment lake. However total native biomass was slightly higher in 2014 compared to pre-treatment years (Figure 8). Conversely, total biomass decreased significantly in untreated Mitchell Lake from 2013 to 2014.

In Lake Riley three (coontail, narrow leaf pondweed, and Canada waterweed) out of the six most commonly occurring species increased slightly from pre-treatment to post-treatment years, however not significantly. White water lily and sago pondweed biomass were slightly lower in 2014 when compared to pre-treatment years. Eurasian watermilfoil biomass increased significantly from 2013 to 2014. With the exception of coontail and Eurasian watermilfoil, all other species biomass values were quite low in Lake Riley (Figure 9).

Native dry plant biomass values in Lake Susan were mostly higher in 2014 than in pre-treatment years, but no increases were significant (Figure 10). Six (coontail, Canada waterweed, American lotus, white water lily, and sago pondweed) out of the eight most commonly occurring native plant species were slightly higher in 2014 compared to pre-treatment years. Yellow water lily and narrow leaf pondweed decreased slightly from pre-treatment years to 2014. Eurasian watermilfoil decreased significantly in 2014 when compared to pre-treatment years.

Total native plant biomass decreased significantly in Mitchell Lake from 2013 to 2014 (Figure 8). Driving the decline in the total native biomass was a significant decline in coontail biomass from 758 ± 124 g dry/$m^2$ in 2013 to 314 ± 45 g dry/$m^2$ in 2014. Of the seven remaining most commonly occurring plant species in Mitchell Lake, four

(Canada waterweed, northern watermilfoil, white water lily, and water stargrass) species declined slightly from 2013 to 2014 (Figure 11).

***Discussion***

Overall, the early-season endothall herbicide treatments were successful in controlling curlyleaf pondweed frequency of occurrence, biomass, and turion production, all of which declined by 90% or more in both lakes. A significant decline in turion density in the sediments was also observed in Lake Susan, but not in Lake Riley, although declines in Lake Riley were substantial. Reductions in turion densities in the sediments in Lake Riley were less than expected and this may reduce efficacy of treatments in the long-term as turion density in the sediments was never significantly reduced. Anecdotal observations at Lake Riley suggest that herbicide applications may not have been as precisely targeted in 2014 and perhaps turion densities were not reduced as greatly as they should have been.

Within treatment-year reductions of curlyleaf pondweed were similar to within year reductions demonstrated by Johnson et al. (2012) and Poovey et al. (2002). Turion viability and viable turion density in the sediments were also dramatically lower in treatment years compared to pre-treatment in both lakes. Because turion viability declined dramatically in the untreated reference lake as well, treatments may not have been a driving factor in reducing turion viability in the treated lakes; it may have been an unmeasured environmental factor such as colder than common water temperatures. Johnson et al. (2012) found that turion viability was significantly reduced in study lakes compared to untreated reference lakes, but he did not have pre-treatment data to compare.

Similar to results from Jones et al. (2012), the overall native plant community was not harmed by endothall herbicide treatments. There were no significant reductions of native plant FO in Lake Riley and the majority of native plants increased slightly during treatment years. In Lake Susan, FO decreased significantly in two native plant species after treatment, but increased significantly in three others. The significant decrease observed in coontail in Lake Susan, which occurred in almost 60% of sites prior to treatment, could be beneficial for the remaining native plant species and may open a niche for other native plants to grow. However, such a niche could also be beneficial for exotics if they are not managed properly. Overall, increases (both significant and otherwise) were more common than decreases when comparing pre-treatment FO to post-treatment years. Future studies should monitor native plant response on a longer temporal scale to determine whether increasing trends in native plant FO would continue.

Post-treatment total native plant biomass was either the same or higher in 2014 in both treatment lakes, whereas it declined significantly in the reference lake, indicating that treatments might have had a slight positive effect on the total native biomass. However, both my study and Jones et al. (2012) found that the herbicide treatments did not result in large or rapid increases in native aquatic plants.

Individual native plant species biomass did not significantly increase or decrease in either treatment lake. However, biomass of most native plant species was higher in 2014 than in pre-treatment years in both treatment lakes. The observed higher biomass values in 2014 are encouraging, however biomass response to treatments appeared to be moderate and was likely hindered by consistently low summer water clarity.

The majority of mean rake ratings (PRR and LWPRR) increased (several significantly) for most native plants species in both treatment lakes. When evaluating the mean rake rating data, it is important to note that this parameter is more subjective and often has a much lower sample size (specifically PRR values) than the other parameters discussed above. However, mean rake ratings of several native plant species increased significantly (particularly in Lake Susan) in post-treatment years, suggesting that native plants have become denser in areas where plants were found, due to the herbicide treatments or curlyleaf control. Meanwhile, no mean rake ratings in the reference lake increased significantly, but two native species mean rake ratings did decrease significantly. The whole rake mean PRR increased in both treatment lakes from pre-treatment years to 2014, while the whole rake rating declined in the untreated reference lake from 2013 to 2014. This again suggests that plants were denser in areas where plants were found in 2014 compared to pre-treatment years in both treatment lakes. Similarly, mean LWPRR values in both lakes were higher for most species in 2014 compared to pre-treatment years. The mean LWPRR values of several species increased significantly from pre-treatment years to post-treatment years in Lake Susan, indicating an increase in both density and distribution.

Native plant populations were likely hindered in both treatment lakes by poor summer water clarity (Knopik 2014). This is likely an issue that needs to be addressed in order to most efficiently test how native plants respond to early season herbicide treatments. Poor water clarity may be slowing the rate at which native plants are responding to the treatments and curlyleaf pondweed control. An additional factor that may also be hindering a native plant response to treatments in Lake Riley is the high density of Eurasian watermilfoil. Eurasian watermilfoil was reduced to very low levels in Lake Susan, which is likely attributable to a high density of milfoil weevils (*Euhrychiopsis lecontei*) (JaKa and Newman 2014) rather than the herbicide treatments. Milfoil weevil densities were low in Lake Riley (JaKa and Newman 2014) and Eurasian watermilfoil was observed at its highest abundance and density in 2014. Eurasian watermilfoil increased significantly in FO and in mean rake rating in post-treatment years in Lake Riley and is a dominant and aggressive species. The high density and abundance of Eurasian watermilfoil is likely negatively affecting the native plant community in Lake Riley.

Another factor that may be slowing the native plant response in both treatment lakes is local lakeshore homeowner herbicide treatments. Both Lake Riley and Lake Susan have heavily developed shorelines. Anecdotal evidence from both treatment lakes suggests that plant growth is chemically controlled on a routine basis by many homeowners based on at least 24 individual treatments taking place in front of homes in lake Riley and 10 treatments in Lake Susan during 2014. Counts of homeowner treatment signs on the shore were recorded when observed during surveys. I suspect some additional local treatments may have occurred based on conversations with homeowners who stated they would occasionally use herbicide pellets left over from previous years without obtaining additional permits. This anecdotal information suggests that local herbicide treatments may also be hindering the native plant response in the treatment lakes at smaller scales.

Under ideal conditions this experiment would have included additional years of post-treatment data. The temporal scale of this project limits the ability to draw broad conclusions regarding the native plant community. Increases in the native plant community, even when not significant, are encouraging. No significant changes being observed in total native plant biomass possibly demonstrates the slow rate of change or lag in the plant community over time. Other studies have shown similar patterns of slow plant community response when light continues to be a limiting factor in a restoration or management project (Hilt et al. 2006, Jeppesen et al. 2007). Significant changes were however observed in some of the finer scale parameters like individual plant species changes from pre- to post- treatment in this study. A similar study with additional post-treatment survey years may yield additional increases in the total native plant community.

Findings from this study agree with findings from other studies showing that curlyleaf pondweed can be successfully controlled (Johnson et al. 2012, Netherland et al. 2000, Poovey et al. 2002, Skogerboe and Getsinger 2002) with little to no measurable harm to the native plant community (Jones et al. 2012, Poovey et al. 2002), however substantial and significant enhancement of native plants was also not evident (Jones et al. 2012). Any management strategy targeting the removal of curlyleaf pondweed should incorporate plant community monitoring for additional invasions, as spot treatments would likely be necessary. Treatments were successful, however Johnson et al. (2012), showed that viable turions could remain at low levels after four to five consecutive treatment years. The treatments in my study resulted in increased native plant populations in varying amounts in all the parameters measured. The native plant response however, was likely hindered by low summer water clarity in both treatment lakes. The water clarity issue may need to be addressed in both study lakes in order to observe significant increases in the overall native plant community at a larger scale and a higher rate of change.

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***Tables*** *Chapter II*

Table 1. List of plant species discussed in this thesis, found in Lakes Riley and Susan and Mitchell Lake (taxonomic authority Crow and Hellquist 2000). *Denotes a non-native invasive species.

| Scientific Name | Common Name | Abbreviated Code |
|---:|---:|---:|
| *Ceratophyllum demersum* | Coontail | Cdem |
| *Chara spp.* | Muskgrass | Char |
| *Elodea canadensis* | Canada waterweed | Ecan |
| *Lemna trisulca* | Star duckweed | Ltri |
| *Myriophyllum sibiricum* | Northern watermilfoil | Msib |
| **Myriophyllum spicatum* | Eurasian watermilfoil | Mspi |
| *Najas flexilis* | Bushy pondweed | Nflex |
| *Nuphar variegata* | Yellow water lily | Nvar |
| *Nymphaea odorata* | White water lily | Nodo |
| *Nelumbo lutea* | American lotus | Nlut |
| **Potamogeton crispus* | Curlyleaf pondweed | Pcri |
| *Potamogeton pusillus* | Narrowleaf pondweed | Ppus |
| *Potamogeton zosteriformis* | Flat-stem pondweed | Pzos |
| *Stuckenia pectinata* | Sago pondweed | Spec |
| *Vallisneria americana* | Wild celery | Vame |
| *Zosterella dubia* | Water stargrass | Zdub |

Table 2. General lake characteristics of the study lakes. Littoral area is depth ≤ 4.6m, as described by the Minnesota Department of Natural Resources.

| Study Lake | Area (ha) | Littoral Area (ha) | Max Depth (m) |
|:---:|---:|---:|---:|
| Mitchell Lake | 46.1 | 44 | 5.8 |
| Lake Riley | 119.8 | 45 | 14.9 |
| Lake Susan | 35.5 | 30 | 5.2 |

Table 3. Lake Riley depth at which 5% of photosynthetically active radiation (PAR) remains and Secchi depths.

| Lake Riley | Depth 5% Light (m) | Secchi Depth (m) |
|---|---|---|
| 7/19/11 | 3.8 | 2.2 |
| 8/25/11 | 1.3 | 0.8 |
| | | |
| 6/8/12 | 3.0 | 2.2 |
| 6/26/12 | 1.8 | 1.3 |
| 7/9/12 | 1.8 | 1.0 |
| 8/14/12 | 1.0 | 0.6 |
| | | |
| 6/5/13 | 4.5 | 4.0 |
| 6/18/13 | 2.3 | 2.0 |
| 7/29/13 | 1.3 | 0.9 |
| 8/14/13 | 1.3 | 0.6 |
| 8/27/13 | 2.3 | 0.9 |
| | | |
| 5/16/14 | 5.0 | 2.4 |
| 5/29/14 | 6.5 | 6.6 |
| 6/5/14 | 2.5 | 2.1 |
| 6/18/14 | 2.0 | 1.4 |
| 7/2/14 | 1.8 | 1.3 |
| 7/30/14 | 2.5 | 1.7 |
| 8/12/14 | 3.0 | 2.1 |

Table 4. Lake Susan depth at which 5% of photosynthetically active radiation (PAR) remains and Secchi depths.

| Lake Susan | Depth 5% Light (m) | Secchi Depth (m) |
|---|---|---|
| 5/14/10 | 4.5 | 3.3 |
| 6/2/10 | 4.0 | 3.5 |
| 6/30/10 | 2.0 | 1.5 |
| 7/6/10 | 2.0 | 1.2 |
| 7/21/10 | 1.8 | 1.0 |
| 7/27/10 | 1.8 | 0.9 |
| 8/13/10 | 0.8 | 0.7 |
| | | |
| 5/19/11 | 5.0 | 5.0 |
| 6/1/11 | 3.3 | 3.2 |
| 6/7/11 | 4.0 | 2.7 |
| 6/14/11 | 3.8 | 2.5 |
| 6/27/11 | 3.0 | 2.3 |
| 7/7/11 | 3.3 | 1.7 |
| 8/3/11 | 1.4 | 0.9 |
| 8/10/11 | 0.9 | 0.8 |
| | | |
| 6/7/12 | 1.3 | 0.7 |
| 6/13/12 | 1.3 | 0.8 |
| 6/28/12 | 0.8 | 0.4 |
| 7/3/12 | 1.3 | 0.8 |
| 7/12/12 | 0.9 | 0.5 |
| 8/6/12 | 0.8 | 0.5 |
| | | |
| 5/6/13 | 1.5 | 1.3 |
| 5/21/13 | 2.0 | 1.8 |
| 6/17/13 | 3.3 | 2.4 |
| 6/26/13 | 1.0 | 1.6 |
| 7/16/13 | 1.3 | 2.0 |
| 8/12/13 | 1.5 | 1.0 |
| 8/27/13 | 1.8 | 1.0 |
| | | |
| 5/13/14 | 4.0 | 2.5 |
| 5/29/14 | 4.5 | 4.9 |
| 6/5/14 | 4.3 | 3.3 |
| 6/17/14 | 2.0 | 1.5 |
| 6/30/14 | 0.8 | 0.9 |
| 7/15/14 | 1.3 | 0.9 |
| 7/28/14 | 1.0 | 0.9 |
| 8/6/14 | 1.8 | 0.9 |
| 8/26/14 | 1 | 0.7 |

Table 5. Mitchell Lake depth at which 5% of photosynthetically active radiation (PAR) remains and Secchi depths.

| Mitchell Lake | Depth 5% Light (m) | Secchi Depth (m) |
|---|---|---|
| 6/6/13 | 3.8 | 3.4 |
| 6/21/13 | 3.0 | 2.5 |
| 8/22/13 | 1.5 | 1.1 |
| | | |
| 5/20/14 | 3.8 | 1.9 |
| 6/12/14 | 3.8 | 2.7 |
| 8/8/14 | 1.8 | 1.1 |

Table 6. Lake Riley turion density in the sediments and turion viability determined during turion viability trials.

| Lake Riley | Turions/$m^2$ | SE | Viability | Viable Turions/$m^2$ |
|---|---|---|---|---|
| October 2011 | 45 | 20 | 96% | 43 |
| October 2012 | 132 | 34 | 99% | 131 |
| October 2013 | 56 | 12 | 71% | 40 |
| October 2014 | 61 | 21 | 33% | 20 |

Table 7. Lake Susan turion density in the sediments and turion viability determined during turion viability trials. *Significant ($p < 0.05$) change from 2012 (pre-treatment).

| Lake Susan | Turions/$m^2$ | SE | Viability | Viable Turions/$m^2$ |
|---|---|---|---|---|
| October 2010 | 24 | 13 | 90% | 22 |
| October 2011 | 51 | 23 | 98% | 50 |
| October 2012 | 87 | 41 | 98% | 85 |
| October 2013 | 18 | 9 | 65% | 12 |
| October 2014 | *8 | 5 | 67% | 5 |

Table 8. Mitchell Lake turion density in the sediments and turion viability determined during turion viability trials in the laboratory.

| Mitchell Lake | Turions/$m^2$ | SE | Viability | Viable Turions /$m^2$ |
|---|---|---|---|---|
| October 2013 | 191 | 44 | 77% | 147 |
| October 2014 | 164 | 33 | 45% | 74 |

Table 9. Lake Riley mean plant rake ratings (PRR) for whole rake and the most common individual species. Approximately 147 sites sampled. Pre-treatment (Pre) is the mean of 2011 and 2012. *Denotes a significant ($p < 0.05$) change from the previous year. +Denotes a significant change from pre-treatment years. *Next to a common name highlights a significant change from pre-treatment to 2014.

| Lake Riley | Pre | SE | n | 2013 | SE | n | 2014 | SE | n |
|---|---|---|---|---|---|---|---|---|---|
| Whole rake | 2.50 | 0.12 | 110 | 2.65 | 0.12 | 86 | 2.83 | 0.13 | 110 |
| Bushy Pondweed | 1.75 | 1.25 | 2 | 2.50 | 0.29 | 4 | 2.63 | 0.38 | 8 |
| Canada waterweed | 0.70 | 0.20 | 5 | *3.00 | 0.00 | 1 | *1.29 | 0.18 | 7 |
| Coontail | 2.23 | 0.12 | 103 | 2.52 | 0.12 | 81 | 2.36 | 0.14 | 80 |
| Eurasian watermilfoil* | 1.30 | 0.13 | 51 | 1.26 | 0.10 | 19 | *+2.29 | 0.14 | 73 |
| Narrowleaf pondweed | 1.25 | 0.20 | 16 | *3.67 | 0.49 | 6 | *1.41 | 0.15 | 17 |
| Sago pondweed* | 0.60 | 0.10 | 5 | 0.00 | 0.00 | 0 | +1.64 | 0.20 | 11 |
| White water lily | 2.50 | 2.00 | 2 | 2.67 | 0.88 | 3 | 2.25 | 0.75 | 4 |

Table 10. Lake Riley mean littoral-wide plant rake ratings (LWPRR) for whole rake and the most common individual species. Pre-treatment (Pre) is the mean of 2011 and 2012. *Denotes a significant ($p < 0.05$) change from the previous year. +Denotes a significant change from pre-treatment years. *Next to a common name highlights a significant change from pre-treatment to 2014.

| Lake Riley | Pre | SE | n | 2013 | SE | n | 2014 | SE | n |
|---|---|---|---|---|---|---|---|---|---|
| Whole rake | 2.10 | 0.14 | 144 | *1.52 | 0.13 | 150 | *2.09 | 0.14 | 149 |
| Bushy Pondweed | 0.02 | 0.02 | 144 | 0.07 | 0.03 | 150 | 0.14 | 0.05 | 149 |
| Canada waterweed | 0.02 | 0.01 | 144 | 0.02 | 0.02 | 150 | 0.06 | 0.06 | 149 |
| Coontail | 1.59 | 0.12 | 144 | 1.36 | 0.12 | 150 | 1.27 | 0.12 | 149 |
| Eurasian watermilfoil | 0.46 | 0.07 | 144 | *0.16 | 0.04 | 150 | *+1.12 | 0.12 | 149 |
| Narrowleaf pondweed | 0.14 | 0.04 | 144 | 0.15 | 0.06 | 150 | 0.16 | 0.04 | 149 |
| Sago pondweed* | 0.02 | 0.01 | 144 | 0 | 0 | 150 | *+0.12 | 0.04 | 149 |
| White water lily | 0.03 | 0.03 | 144 | 0.05 | 0.03 | 150 | 0.06 | 0.03 | 149 |

Table 11. Lake Susan mean plant rake ratings (PRR) for whole rake and the most common individual species. Approximately 71 sites sampled. Pre-treatment (Pre) is the mean of 2009, 2010, 2011, and 2012. *Denotes a significant ($p < 0.05$) change from the previous year. +Denotes a significant change from pre-treatment years. *Next to a common name highlights a significant change from pre-treatment to 2014.

| Lake Susan | Pre | SE | n | 2013 | SE | n | 2014 | SE | n |
|---|---|---|---|---|---|---|---|---|---|
| Whole Rake* | 2.65 | 0.17 | 58 | *4.10 | 0.19 | 39 | *+3.36 | 0.21 | 44 |
| American lotus* | 0.93 | 0.12 | 23 | *2.61 | 0.31 | 18 | +2.11 | 0.24 | 18 |
| Bushy pondweed | 0.25 | 0.25 | 1 | *2.50 | 0.50 | 2 | *1.45 | 0.16 | 11 |
| Canada waterweed* | 0.77 | 0.08 | 32 | *2.25 | 0.33 | 12 | +2.57 | 0.20 | 30 |
| Coontail | 1.86 | 0.17 | 54 | 2.04 | 0.16 | 27 | 2.27 | 0.18 | 37 |
| Eurasian watermilfoil* | 0.41 | 0.06 | 41 | 0.00 | 0.00 | 0 | +1.00 | 0.00 | 5 |
| Narrowleaf pondweed | 0.91 | 0.10 | 50 | *3.19 | 0.28 | 21 | *1.29 | 0.18 | 7 |
| Sago pondweed* | 0.56 | 0.08 | 16 | *1.60 | 0.24 | 5 | +1.67 | 0.49 | 6 |
| White water lily* | 0.95 | 0.19 | 16 | *2.67 | 0.55 | 9 | +2.11 | 0.24 | 18 |
| Yellow water lily | 1.11 | 0.23 | 16 | *4.43 | 0.20 | 7 | *3.33 | 0.42 | 6 |

Table 12. Lake Susan mean littoral-wide plant rake ratings (LWPRR) for whole rake and the most common individual species. Pre-treatment (Pre) is the mean of 2009, 2010, 2011, and 2012. *Denotes a significant ($p < 0.05$) change from the previous year. +Denotes a significant change from pre-treatment years. *Next to a common name highlights a significant change from pre-treatment to 2014

| Lake Susan | Pre | SE | n | 2013 | SE | n | 2014 | SE | n |
|---|---|---|---|---|---|---|---|---|---|
| Whole Rake* | 2.47 | 0.19 | 66 | 2.29 | 0.27 | 70 | +1.78 | 0.21 | 83 |
| American lotus | 0.33 | 0.07 | 66 | 0.67 | 0.16 | 70 | 0.46 | 0.11 | 83 |
| Bushy pondweed* | 0 | 0 | 66 | 0.07 | 0.05 | 70 | +0.19 | 0.06 | 83 |
| Canada waterweed* | 0.37 | 0.06 | 66 | 0.39 | 0.12 | 70 | *+0.93 | 0.15 | 83 |
| Coontail* | 1.52 | 0.17 | 66 | *0.79 | 0.13 | 70 | *+1.01 | 0.15 | 83 |
| Eurasian watermilfoil* | 0.26 | 0.04 | 66 | *0.67 | 0.16 | 70 | +0.06 | 0.03 | 83 |
| Narrowleaf pondweed* | 0.69 | 0.09 | 66 | 0.96 | 0.19 | 70 | *+0.11 | 0.04 | 83 |
| Sago pondweed | 0.14 | 0.04 | 66 | 0.11 | 0.05 | 70 | 0.12 | 006 | 83 |
| White water lily | 0.23 | 0.07 | 66 | 0.34 | 0.13 | 70 | 0.46 | 0.11 | 83 |
| Yellow water lily | 0.27 | 0.08 | 66 | 0.44 | 0.16 | 70 | 0.24 | 0.1 | 83 |

Table 13. Mitchell Lake mean plant rake ratings (PRR) for whole rake and the most common individual species. Approximately 160 sites sampled. *Denotes a significant ($p < 0.05$) change from the previous year.

| Mitchell Lake | 2013 | SE | n | 2014 | SE | n |
|---|---|---|---|---|---|---|
| Whole Rake | 3.98 | 0.10 | 141 | 3.71 | 0.11 | 125 |
| Coontail | 3.42 | 0.12 | 135 | 3.76 | 0.47 | 120 |
| Canada waterweed | 1.80 | 0.58 | 5 | 1.33 | 0.33 | 3 |
| Star duckweed | 1.51 | 0.07 | 74 | *1.09 | 0.04 | 65 |
| Northern watermilfoil | 2.88 | 0.17 | 49 | 2.43 | 0.21 | 37 |
| White water lily | 2.51 | 0.22 | 37 | *1.40 | 0.12 | 40 |
| Narrowleaf pondweed | 1.45 | 0.16 | 11 | 1.43 | 0.17 | 14 |
| Flat-stem pondweed | 2.18 | 0.21 | 22 | 2.32 | 0.17 | 37 |
| Sago pondweed | 1.25 | 0.25 | 4 | 2.00 | 0.27 | 8 |
| Water stargrass | 1.91 | 0.31 | 11 | 2.07 | 0.25 | 14 |

Table14. Mitchell Lake mean littoral-wide plant rake ratings (LWPRR) for whole rake and the most common individual species. *Denotes a significant ($p < 0.05$) change from the previous year.

| Mitchell Lake | 2013 | SE | n | 2014 | SE | n |
|---|---|---|---|---|---|---|
| Whole Rake | 3.46 | 0.14 | 162 | *2.96 | 015 | 157 |
| Coontail | 2.85 | 0.14 | 162 | 2.87 | 0.38 | 157 |
| Canada waterweed | 0.06 | 0.03 | 162 | 0.03 | 0.02 | 157 |
| Star duckweed | 0.69 | 0.07 | 162 | *0.45 | 0.05 | 157 |
| Northern watermilfoil | 0.87 | 0.12 | 162 | 0.57 | 0.10 | 157 |
| White water lily | 0.57 | 0.10 | 162 | 0.36 | 0.06 | 157 |
| Narrowleaf pondweed | 0.10 | 0.03 | 162 | 0.13 | 0.04 | 157 |
| Flat-stem pondweed | 0.03 | 0.07 | 162 | *0.55 | 0.09 | 157 |
| Sago pondweed | 0.03 | 0.02 | 162 | 0.10 | 0.04 | 157 |
| Water stargrass | 0.13 | 0.04 | 162 | 0.18 | 0.05 | 157 |

*Figures* Chapter II

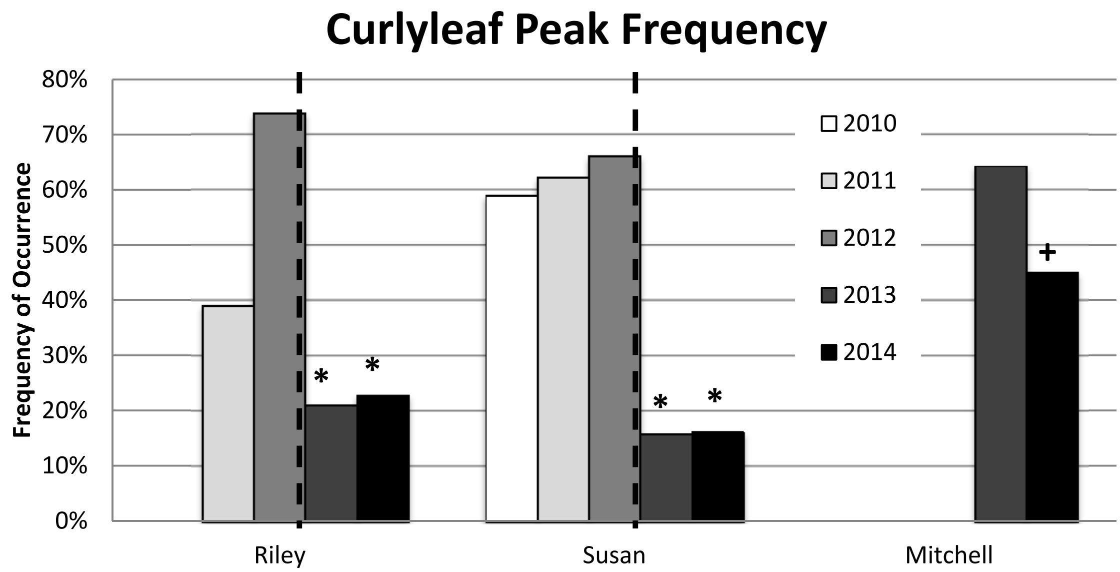


Figure 1. Curlyleaf pondweed frequency of occurrence in all study lakes. *Denotes a significant ($p < 0.05$) change when compared to 2012. +Denotes a significant decrease from 2013 to 2014. Dashed line represents initiation of herbicide treatments.

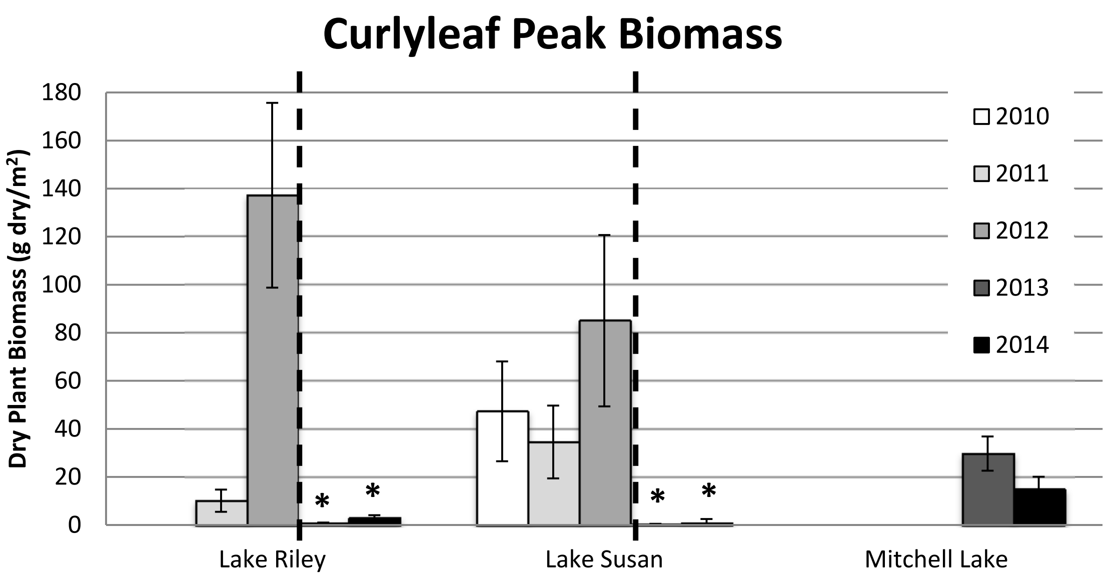


Figure 2. Curlyleaf pondweed biomass (g dry/$m^2$ ±1 SE) in all study lakes. *Denotes a significant ($p < 0.05$) change when compared to 2012. Dashed line represents initiation of herbicide treatments.

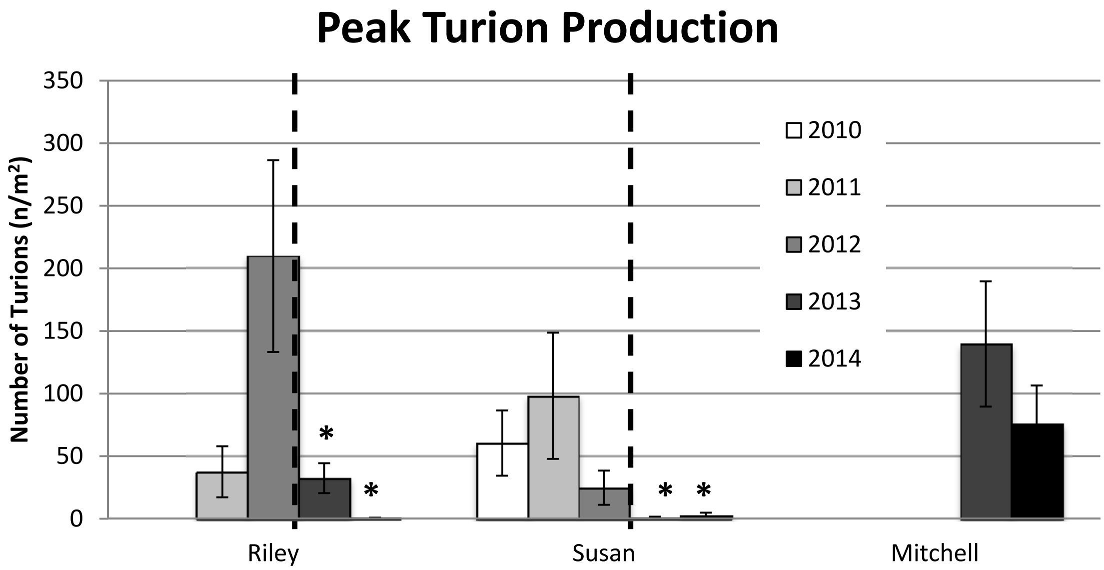


Figure 3. Curlyleaf pondweed turion production (n/m$^2$ ± 1 SE) in all study lakes. *Denotes a significant (p <0.05) change when compared to 2012. Dashed line represents initiation of herbicide treatments.

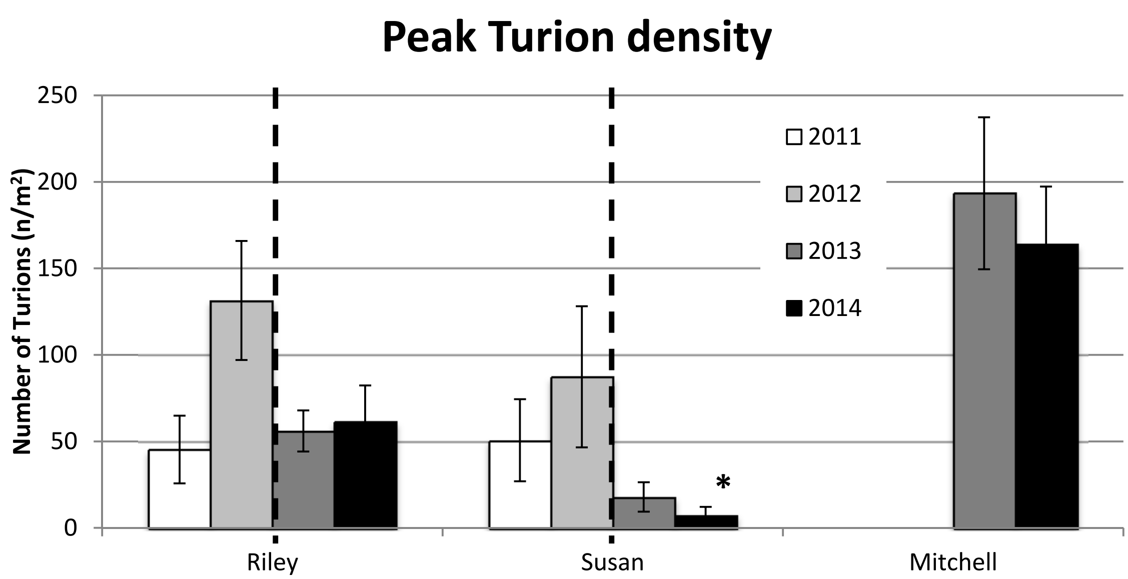


Figure 4. Curlyleaf pondweed turion density in the sediments (n/m$^2$ ± 1 SE) in all study lakes. *Denotes a significant (p <0.05) change when compared to 2012. Dashed line represents initiation of herbicide treatments.

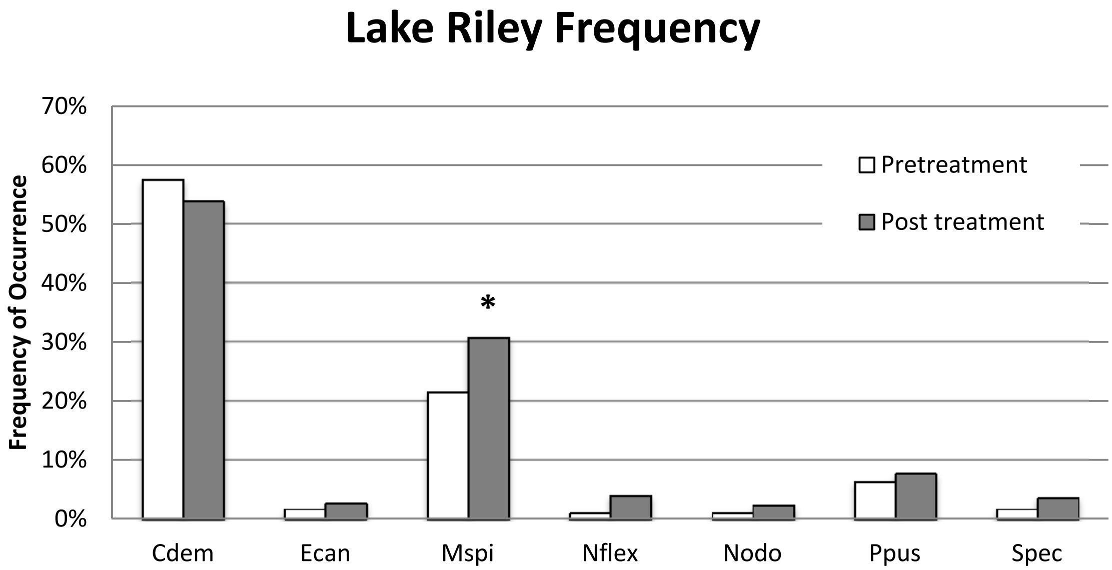


Figure 5. A comparison of pre-treatment years (mean of 2011 and 2012) and post-treatment years (mean of 2013 and 2014) frequency of occurrence for the most commonly occurring species in Lake Riley. *Denotes a significant ($p < 0.05$) change. Pre-treatment is the mean of 2011 and 2012 and post-treatment is the mean of 2013 and 2014. See Table 1 for definition of abbreviation codes.

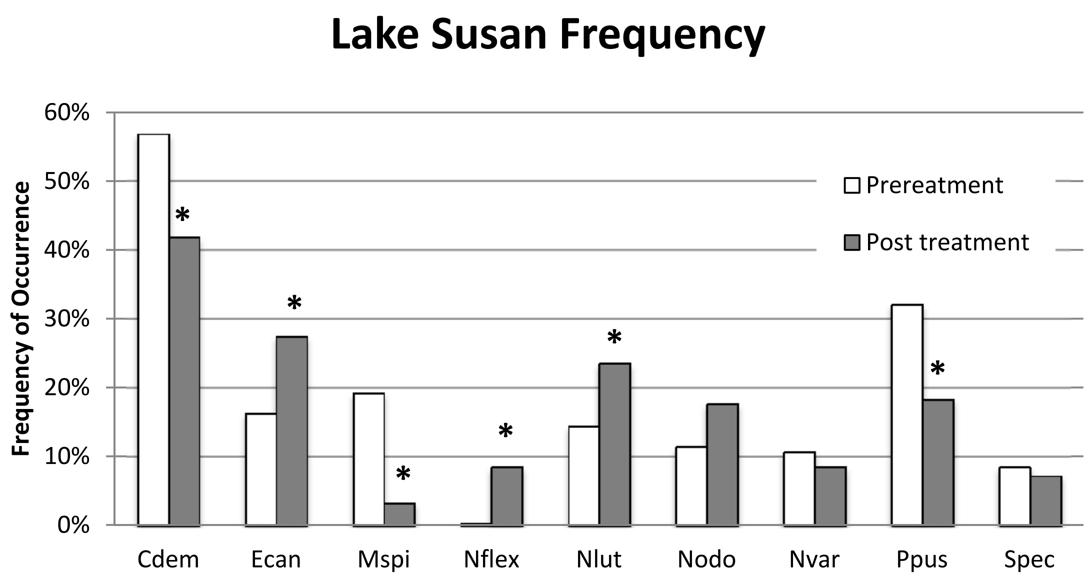


Figure 6. A comparison of pre-treatment years (mean of 2009 through 2012) and post-treatment years (mean of 2013 and 2014) frequency of occurrence for the most commonly occurring species in Lake Susan. *Denotes a significant ($p < 0.05$) change. Pre-treatment is the mean of 2009 through 2012 and post-treatment is the mean of 2013 and 2014. See Table 1 for definition of abbreviation codes.

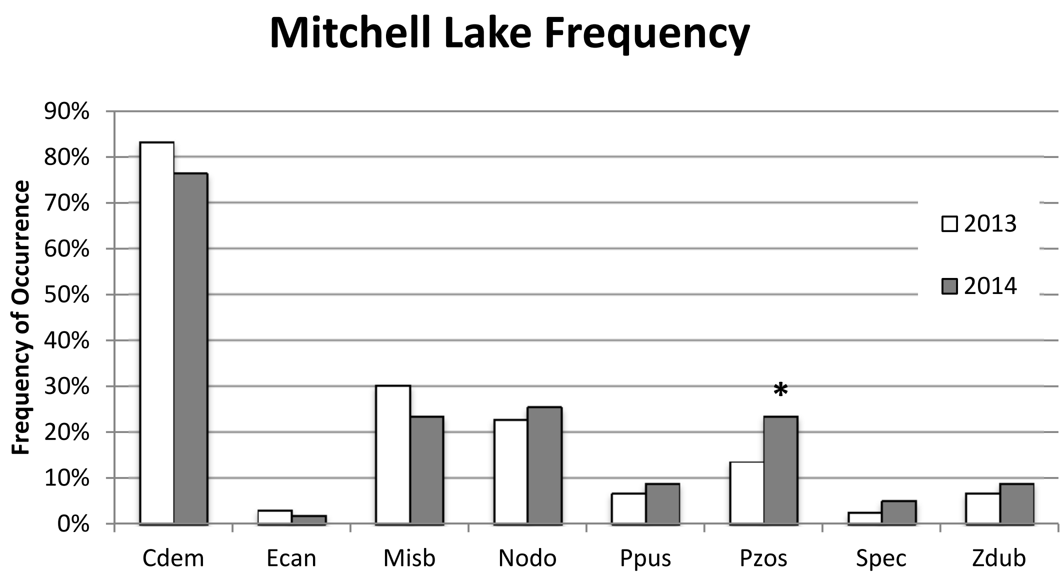


Figure 7. Mitchell Lake frequency of occurrence for the most commonly occurring species. *Denotes a significant ($p < 0.05$) change. See Table 1 for definition of abbreviation codes.

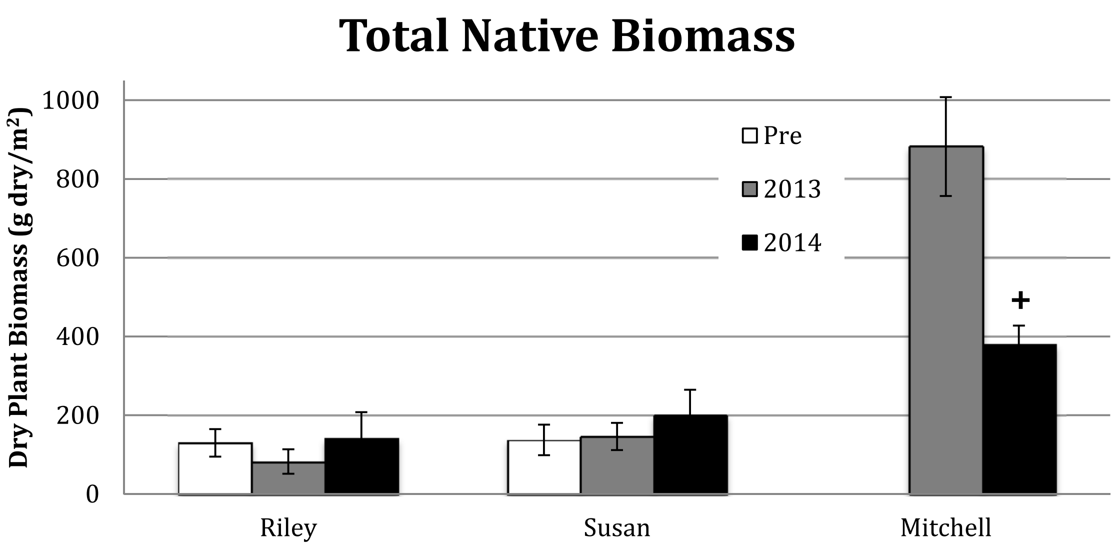


Figure 8. Total (all native species) mean native plant biomass (g dry/$m^2 \pm 1$ SE) for all study lakes. Year Pre is the mean of all pre-treatment years. +Denotes a significant ($p < 0.05$) change from 2013 to 2014.

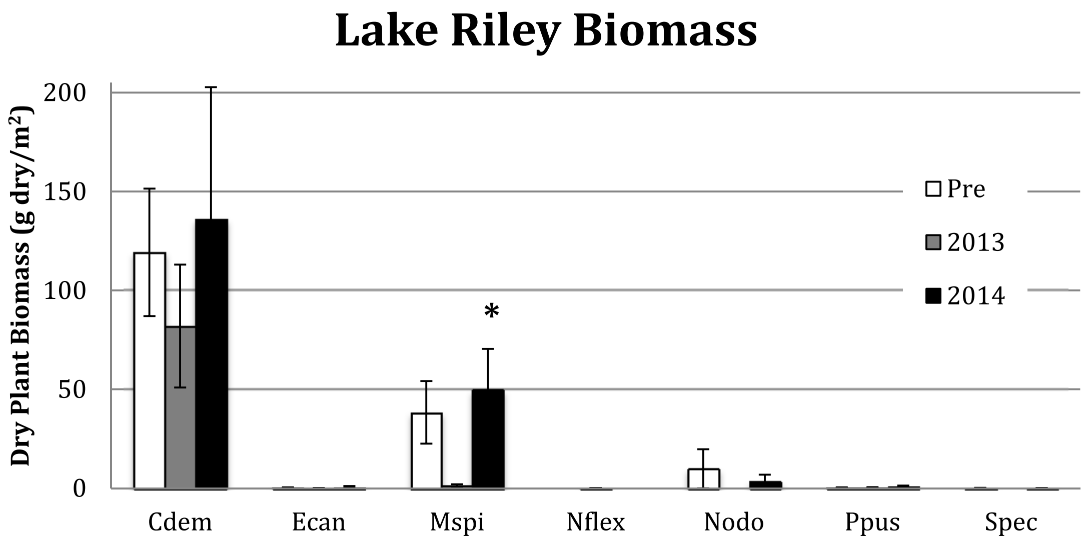


Figure 9. A comparison of pre-treatment years with 2013, and 2014 in Lake Riley biomass (g dry/$m^2$ ± 1 SE) for the most commonly occurring species. Year Pre is the mean of all pre-treatment years. *Denotes a significant ($p < 0.05$) change from the previous year. See Table 1 for definition of abbreviation codes.

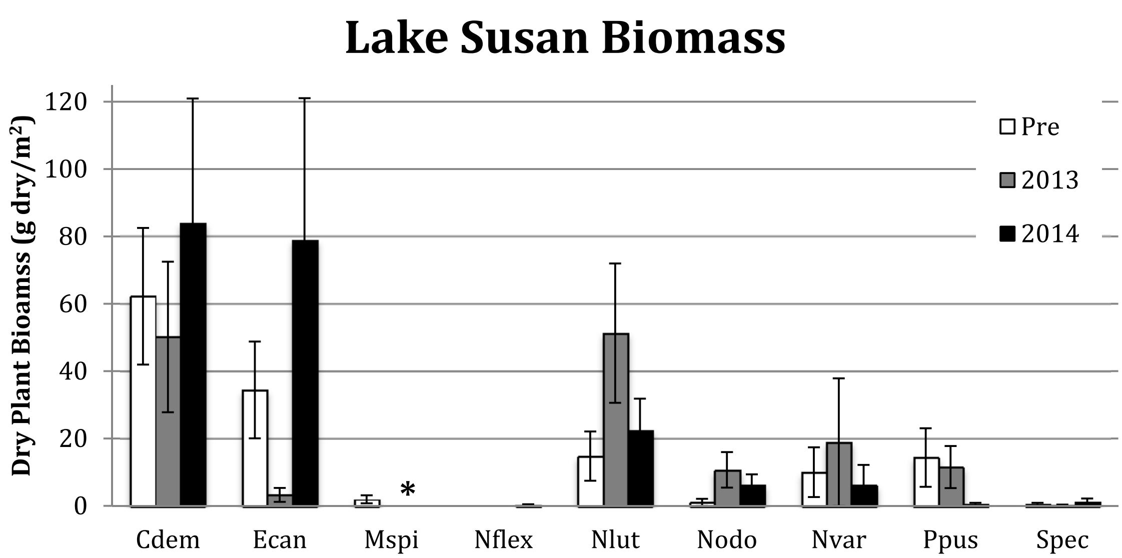


Figure 10. A comparison of pre-treatment years with 2013, and 2014 in Lake Susan biomass (g dry/$m^2$ ± 1 SE) for the most commonly occurring species. Year Pre is the mean of all pre-treatment years. *Denotes a significant ($p < 0.05$) change when compared to pre-treatment years. See Table 1 for definition of abbreviation codes.

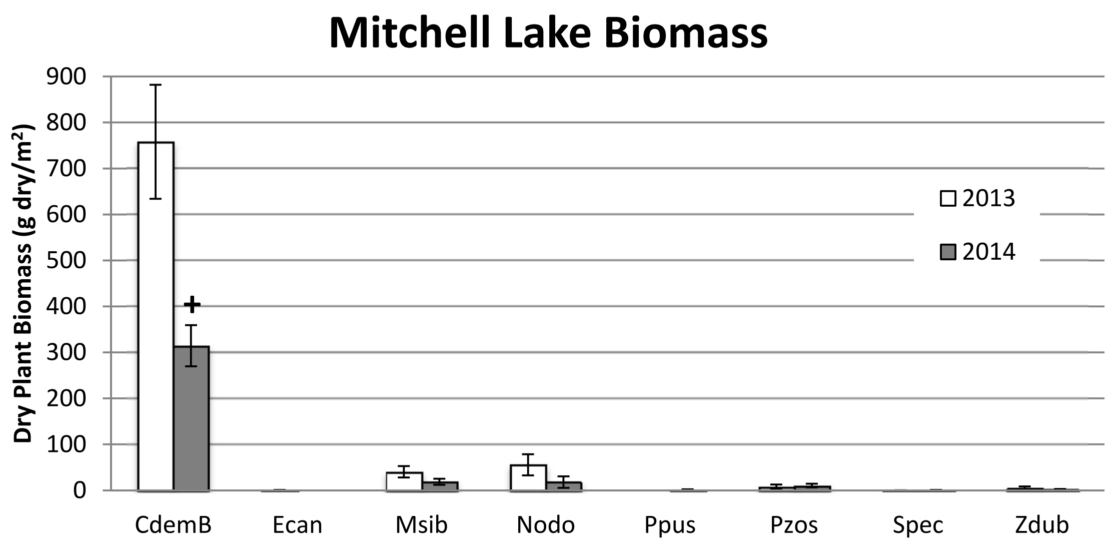


Figure 11. Mitchell Lake biomass (g dry/$m^2$ ± 1 SE) for the most commonly occurring species in 2013 and 2014. +Denotes a significant ($p < 0.05$) change. See Table 1 for definition of abbreviation codes.

# Chapter III

## Concluding Remarks:

Native aquatic plants play an important role in a healthy aquatic ecosystem. They provide direct functions such as reducing wave action and stabilizing sediments and they provide indirect benefit to semi-aquatic and aquatic organisms as shelter and forage base. Native aquatic plants help to increase stability and resiliency in an aquatic ecosystem (Hilt 2010). A reduction in nutrients in the water column creates a positive feedback loop that aids in the maintenance of a clear water state, especially in shallow lakes (Carpenter and Lodge 1986, Scheffer 2001). Invasive species such as curlyleaf pondweed that form dense monotypic stands can displace native aquatic plants, disrupt recreation activities and create negative feed back loops in an aquatic ecosystem (Nichols and Shaw 1986, Bolduan et al. 1994). Lakes not infested with invasive aquatic plants often have higher scores on indices of biotic integrity (Beck et al. 2010) and support a more healthy aquatic system (Valley et al. 2004). Endothall herbicide treatments, especially those conducted during early season, have been suggested to reduce curlyleaf pondweed frequency of occurrence (FO), biomass, and turion production (Poovey et al. 2002, Johnson et al. 2012), while having minimal impact on the native plant community (Skogerboe and Getsinger 2002, Jones et al. 2012). However, these studies lacked pre-treatment data making assessments of the effectiveness and specificity of endothall treatments difficult.

I evaluated the magnitude to which early season endothall treatments controlled curlyleaf pondweed using several years of pre-treatment data and two years of post-treatment data in two metro lakes and one un-treated reference lake. Curlyleaf pondweed FO, biomass, turion production and turion density in the sediments were steadily increasing prior to treatments. Treatments significantly reduced the FO, biomass, and turion production in both treatments lakes, while no significant changes occurred in the reference lake except for a decline in FO. However, while turion production declined significantly following treatments, the decline in turion density in the sediment was not always significant. This suggests that endothall treatments may need to extend beyond two years as also suggested by Johnson et al. (2012) who documented that viable turions could remain at low levels after four to five consecutive treatment years.

I also evaluated the response of the native plant community to early season endothall herbicide treatments and curlyleaf pondweed control. The native plant community generally responded positively, although many observed increases were not statistically significant. Jones et al. (2012) also found that native plant biomass increased substantially in several treatment lakes, but none were significant. There were several significant increases in biomass, FO, and mean rake ratings in Lakes Riley and Susan. The FO of Canada waterweed, bushy pondweed and American lotus increased significantly after herbicide treatments in Lake Susan. Sago pondweed PRR and LWPRR increased significantly in Lake Riley after herbicide treatments and several other species also increased as also did the invasive Eurasian watermilfoil. The PRR values of most native species increased significantly in Lake Susan after herbicide treatments. Bushy pondweed and Canada waterweed mean LWPRR values increased significantly after herbicide treatments as well.

The native plant community response is likely hindered by low water clarity in both treatment lakes. The Lake Riley mean summer Secchi depth from 2011 through 2014 was 1.2 m and the mean summer Secchi depth in 2010 through 2014 in Lake Susan was 1.0 m. Similarly to what Johnson et al. (2012) and Jones et al. (2012) observed, control of curlyleaf pondweed did not seem to increase water clarity in summer. Lakes

Riley and Susan will likely require additional management strategies to further enhance the native plant community. The plant community in both study lakes would likely benefit from aluminum sulfate (Alum) treatments, which are typically associated with a significant increase in spring and summertime water clarity.

Johnson et al. (2012) showed that with complementary years of spot treatments, curlyleaf pondweed can be maintained at reduced levels using less intense management compared to initial treatment years. Additional monitoring and possible spot treatments will likely be necessary in both treatment lakes. Additional herbicide treatments targeting the control of Eurasian watermilfoil is also necessary in Lake Riley, especially if Alum treatments will be part of the future management strategy. Alum treatments would likely benefit all aquatic macrophytes, including invasive species, which makes continued monitoring and Eurasian watermilfoil control increasingly important.

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Appendix
Supplemental Information

Appendix Table 1. List of plant species found in Lakes Riley and Susan and Mitchell Lake (taxonomic authority Crow and Hellquist 2000).

| Common Name | Scientific Name | Abbreviation |
|---|---|---|
| Coontail | *Ceratophyllum demersum* | Cdem |
| Muskgrass | *Chara spp.* | Char |
| Canada waterweed | *Elodea canadensis* | Ecan |
| Lesser duckweed | *Lemna minor* | Lmin |
| Star duckweed | *Lemna trisulca* | Ltri |
| Northern watermilfoil | *Myriophyllum sibiricum* | Msib |
| Eurasian watermilfoil | *Myriophyllum spicatum* | Mspi |
| Bushy pondweed | *Najas flexillis* | Nfle(x) |
| American lotus | *Nelumbo lutea* | Nlut |
| Yellow water lily | *Nuphar variegata* | Nvar |
| White water lily | *Nymphaea odorata* | Nodo |
| Curlyleaf pondweed | *Potamogeton crispus* | Pcri |
| Narrowleaf pondweed | *Potamogeton pusillus* | Ppus |
| Flat-stem pondweed | *Potamogeton zosteriformis* | Pzos |
| White water buttercup | *Ranunculus longirostris* | Rlon |
| Great duckweed | *Spirodella polyrhiza* | Spol |
| Sago pondweed | *Stuckenia pectinata* | Spec |
| Common bladderwort | *Utricularia vulgaris* | Uvul |
| American celery | *Vallisneria americana* | Vame |
| Water stargrass | *Zosterella dubia* | Zdub |
| Horned pondweed | *Zannichellia palustris* | Zpal |

Appendix Table 2. Frequncies of occurrence for all plant species in all surveys in Lake Riley. Frequencies were calculated using a littoral zone of 4.6 m. Definitions for abbreviations can be found in Appendix Table 1.

| | Cdem | Chara | Ecan | Lmin | Mspi | Nfle | Nodo | Pcri | Ppus | Spec | Zpal |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **Jun 11** | 48.5% | 0.0% | 2.3% | 0.0% | 31.0% | 0.0% | 0.6% | 32.2% | 3.5% | 5.8% | 0.6% |
| **Aug 11** | 45.1% | 0.0% | 1.1% | 0.0% | 6.9% | 0.0% | 0.6% | 0.5% | 6.3% | 2.3% | 1.7% |
| | | | | | | | | | | | |
| **May 12** | 39.6% | 0.0% | 2.4% | 0.0% | 50.3% | 0.0% | 1.8% | 62.1% | 3.6% | 5.3% | 1.2% |
| **Jun 12** | 54.9% | 0.6% | 6.3% | 0.0% | 55.4% | 0.6% | 1.1% | 25.1% | 9.1% | 5.1% | 0.0% |
| **Aug 12** | 53.6% | 0.0% | 1.7% | 0.0% | 28.7% | 1.1% | 1.1% | 2.8% | 4.4% | 0.6% | 0.0% |
| | | | | | | | | | | | |
| **May 13** | 32.0% | 0.0% | 0.0% | 0.0% | 23.0% | 0.0% | 0.0% | 53.4% | 0.0% | 0.0% | 0.0% |
| **Jun 13** | 53.5% | 0.0% | 2.4% | 0.0% | 42.9% | 0.0% | 0.0% | 18.2% | 6.5% | 0.6% | 0.0% |
| **Aug 13** | 46.1% | 0.6% | 0.6% | 0.0% | 10.7% | 2.2% | 1.7% | 1.1% | 3.4% | 0.0% | 0.0% |
| | | | | | | | | | | | |
| **May 14** | 37.2% | 0.0% | 0.0% | 0 | 10.6% | 0.0% | 0.0% | 26.7% | 7.2% | 0.6% | 0.0% |
| **Jun 14** | 43.8% | 1.1% | 0.6% | 1.1% | 37.5% | 1.1% | 2.3% | 19.9% | 4.0% | 2.3% | 1.1% |
| **Aug 14** | 45.3% | 1.7% | 3.9% | 0 | 40.8% | 4.5% | 2.2% | 2.8% | 10.1% | 6.1% | 0.0% |

Appendix Table 3. Frequncies of occurrence for all plant species in all surveys in Lake Susan. Frequencies were calculated using a littoral zone of 4.6 m. Definitions for abbreviations can be found in Appendix Table 1.

| | Cdem | Ecan | Lmin | Ltri | Mspi | Nfle | Nlut | Nodo | Nvar | Pcri | Pnod | Ppus | Pzos | Rlon | Spec | Vame | Zdub | Zpal |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Jun 09** | 43.0% | 0.0% | 3.0% | 0.0% | 35.0% | 0.0% | 0.0% | 10.0% | 5.0% | 17.0% | 0.0% | 15.0% | 0.0% | 0.0% | 1.0% | 0.0% | 0.0% | 6.0% |
| **Aug 09** | 37.0% | 2.0% | 0.0% | 0.0% | 32.0% | 0.0% | 7.0% | 8.0% | 5.0% | 6.0% | 0.0% | 17.0% | 0.0% | 0.0% | 3.0% | 0.0% | 0.0% | 1.0% |
| **Jun 10** | 36.6% | 3.8% | 0.0% | 0.0% | 20.6% | 7.6% | 7.6% | 0.0% | 9.2% | 28.2% | 0.0% | 32.8% | 0.0% | 0.0% | 6.1% | 0.0% | 0.0% | 0.0% |
| **Aug 10** | 29.8% | 1.5% | 0.0% | 0.0% | 3.8% | 3.8% | 3.8% | 0.0% | 6.9% | 1.5% | 0.0% | 18.3% | 0.0% | 0.0% | 9.9% | 0.0% | 0.0% | 0.0% |
| **Sep 10** | 32.1% | 1.5% | 0.0% | 0.0% | 8.4% | 6.1% | 6.1% | 0.0% | 3.8% | 6.1% | 0.0% | 6.1% | 0.0% | 0.0% | 5.3% | 0.0% | 0.0% | 0.0% |
| **May 11** | 43.9% | 11.2% | 0.0% | 0.0% | 10.2% | 0.0% | 0.0% | 7.1% | 1.0% | 29.6% | 0.0% | 14.3% | 0.0% | 0.0% | 10.2% | 0.0% | 0.0% | 0.0% |
| **Jun 11** | 53.2% | 26.6% | 0.0% | 0.0% | 13.8% | 5.3% | 5.3% | 3.2% | 6.4% | 41.5% | 0.0% | 35.1% | 0.0% | 0.0% | 7.4% | 0.0% | 0.0% | 0.0% |
| **Aug 11** | 38.8% | 23.1% | 5.0% | 0.8% | 9.9% | 0.8% | 10.7% | 8.3% | 5.0% | 7.4% | 0.0% | 30.6% | 0.0% | 0.0% | 5.8% | 0.0% | 0.0% | 0.0% |
| **May 12** | 27.9% | 14.4% | 0.0% | 0.0% | 17.3% | 0.0% | 6.7% | 1.9% | 7.7% | 40.4% | 0.0% | 20.2% | 0.0% | 0.0% | 1.9% | 0.0% | 0.0% | 0.0% |
| **Jun 12** | 34.2% | 18.9% | 0.0% | 0.0% | 9.9% | 0.0% | 9.9% | 6.3% | 5.4% | 23.4% | 0.0% | 21.6% | 0.0% | 0.0% | 8.1% | 0.0% | 0.0% | 0.0% |
| **Aug 12** | 23.0% | 10.7% | 0.8% | 0.0% | 0.8% | 0.0% | 10.7% | 5.7% | 7.4% | 2.5% | 0.8% | 6.6% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% |
| **May 13** | 7.5% | 5.8% | 0.0% | 0.0% | 0.8% | 0.0% | 0.0% | 0.0% | 0.0% | 38.3% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% |
| **Jun 13** | 19.0% | 6.3% | 0.0% | 0.0% | 0.8% | 0.0% | 8.7% | 8.7% | 6.3% | 7.9% | 0.0% | 7.9% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% | 0.0% |
| **Aug 13** | 20.6% | 9.2% | 0.0% | 0.8% | 0.0% | 1.5% | 14.5% | 6.9% | 5.3% | 0.8% | 0.8% | 16.0% | 0.0% | 0.0% | 3.8% | 0.0% | 0.0% | 0.0% |
| **May 14** | 22.3% | 16.5% | 0.0% | 0.0% | 0.0% | 1.0% | 0.0% | 0.0% | 2.9% | 29.1% | 0.0% | 9.7% | 0.0% | 1.0% | 0.0% | 0.0% | 0.0% | 0.0% |
| **Jun 14** | 26.4% | 21.6% | 0.0% | 0.0% | 1.6% | 0.8% | 8.0% | 14.4% | 5.6% | 8.8% | 0.8% | 10.4% | 0.0% | 0.0% | 3.2% | 0.0% | 0.0% | 0.8% |
| **Aug 14** | 25.7% | 20.8% | 0.0% | 0.7% | 3.5% | 7.6% | 12.5% | 12.5% | 4.2% | 2.1% | 0.0% | 4.9% | 0.7% | 0.0% | 4.2% | 0.7% | 1.4% | 1.4% |

Appendix Table 4. Frequncies of occurrence for all plant species in all surveys in Mitchell Lake. Frequencies were calculated using a littoral zone of 4.6 m. Definitions for abbreviations can be found in Appendix Table 1.

| | Cdem | Chara | Ecan | Lmin | Ltri | Msib | Mspi | Nfle | Nlut | Nodo | Nvar | Pcri | Ppus | Pzos | Rlon | Spec | Spol | Typh | Uvul | Zdub |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Jun 13** | 58.2% | 0.0% | 1.1% | 0.5% | 37.0% | 28.8% | 1.1% | 0.0% | 0.0% | 14.7% | 0.0% | 66.8% | 22.8% | 7.6% | 0.0% | 0.0% | 2.7% | 0.5% | 0.0% | 0.0% |
| **Jun 13** | 62.0% | 0.0% | 2.7% | 0.0% | 35.3% | 35.9% | 3.3% | 0.0% | 0.0% | 15.2% | 0.5% | 59.2% | 28.3% | 9.2% | 7.1% | 0.0% | 6.0% | 0.0% | 0.0% | 4.9% |
| **Aug 13** | 76.2% | 5.4% | 2.7% | 0.0% | 41.6% | 27.0% | 4.3% | 0.5% | 0.5% | 22.2% | 0.0% | 1.6% | 6.5% | 11.9% | 4.9% | 2.2% | 16.8% | 0.0% | 0.5% | 5.9% |
| | | | | | | | | | | | | | | | | | | | | |
| **May14** | 64.9% | 4.0% | 1.1% | 1.7% | 50.0% | 12.1% | 2.9% | 0.6% | 0.0% | 0.6% | 0.0% | 33.3% | 13.8% | 7.5% | 5.2% | 0.0% | 0.0% | 0.0% | 0.6% | 1.7% |
| **Jun 14** | 74.9% | 2.3% | 3.4% | 0.0% | 55.4% | 20.0% | 1.1% | 0.0% | 0.6% | 20.6% | 0.0% | 48.6% | 29.1% | 22.3% | 5.1% | 3.4% | 3.4% | 0.0% | 0.6% | 8.6% |
| **Aug 14** | 72.3% | 0.0% | 1.7% | 0.6% | 38.2% | 22.0% | 1.2% | 0.0% | 0.6% | 24.3% | 0.0% | 0.6% | 8.1% | 21.4% | 1.7% | 4.6% | 5.8% | 0.0% | 0.0% | 8.1% |

Appendix Table 5. Mean plant biomass (g dry/m$^2$) for all plant species in all surveys in Lake Riley. Biomass was calculated using a littoral zone of 4.6 m. Definitions for abbreviations can be found in Appendix Table 1.

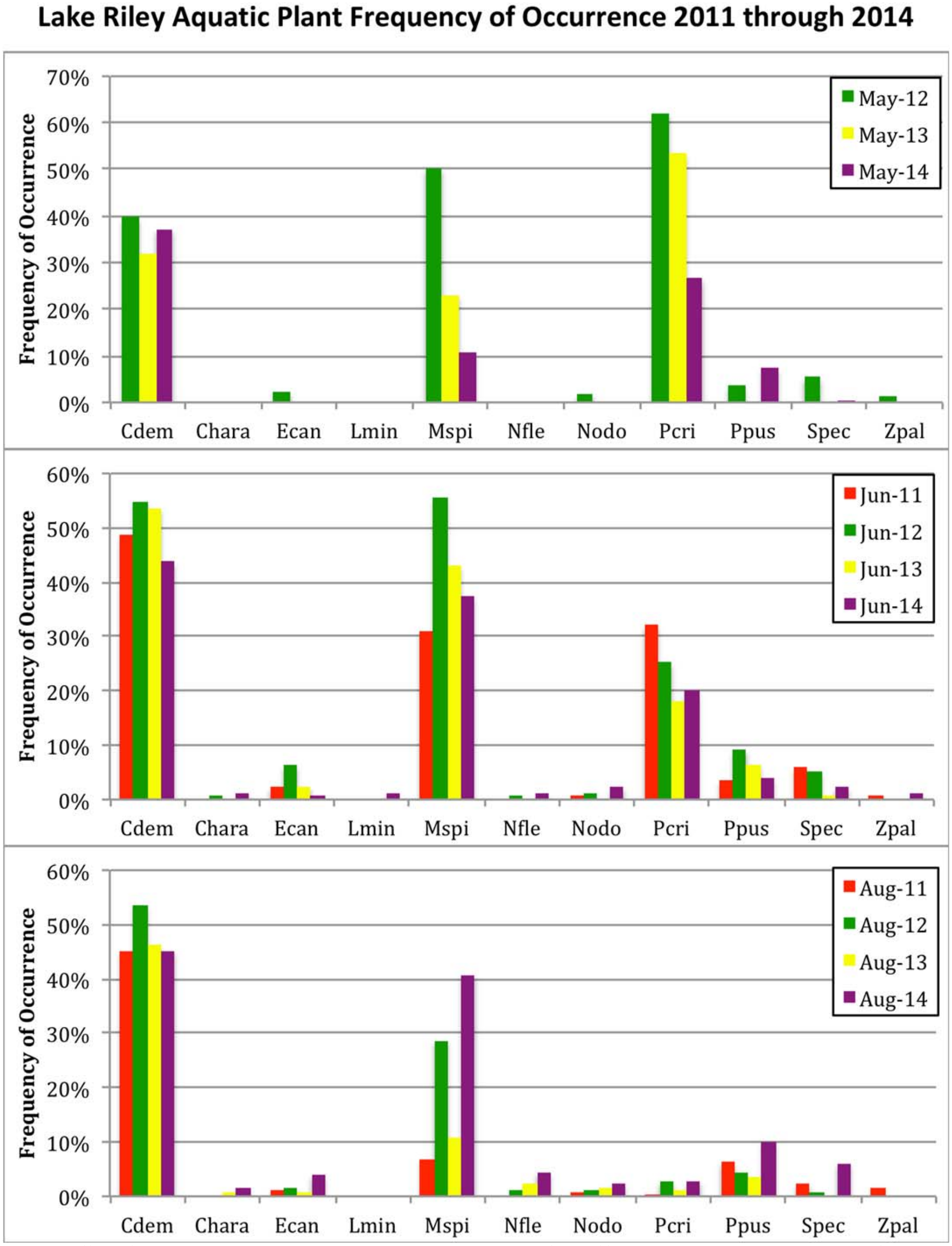
| | Cdem | Chara | Ecan | Mspi | Nfle | Nodo | Pcri | Ppus | Spec | Zpal |
|---|---|---|---|---|---|---|---|---|---|---|
| **Jun 11** | 26.83 | - | 0.04 | 9.59 | - | - | 8.10 | 0.47 | 0.09 | 0.03 |
| **SE** | 8.38 | - | 0.03 | 4.67 | - | - | 3.76 | 0.29 | 0.09 | 0.03 |
| **Aug 11** | 56.82 | - | 0.08 | 36.86 | - | - | - | 0.17 | - | - |
| **SE** | 2.96 | - | 0.01 | 2.93 | - | - | - | 0.02 | - | - |
| | | | | | | | | | | |
| **May 12** | 69.71 | - | 0.40 | 52.04 | - | - | 120.09 | 0.26 | 1.01 | - |
| **SE** | 29.60 | - | 0.40 | 25.77 | - | - | 34.38 | 0.26 | 0.72 | - |
| **Jun 12** | 173.42 | - | 0.58 | 124.12 | - | 0.91 | 2.85 | 2.14 | 3.38 | - |
| **SE** | 50.53 | - | 0.32 | 54.28 | - | 0.91 | 0.84 | 1.67 | 2.98 | - |
| **Aug 12** | 199.65 | 0.04 | 0.39 | 25.11 | - | 15.47 | 0.39 | 0.21 | 0.14 | - |
| **SE** | 77.55 | 0.04 | 0.26 | 17.70 | - | 15.47 | 0.19 | 0.21 | 0.14 | - |
| | | | | | | | | | | |
| **May 13** | 19.78 | - | 0.95 | 0.29 | - | - | 7.28 | - | - | - |
| **SE** | 7.87 | - | 0.52 | 0.21 | - | - | 1.63 | - | - | - |
| **Jun 13** | 38.95 | - | - | 22.72 | - | - | 0.67 | 0.22 | - | - |
| **SE** | 11.98 | - | - | 8.44 | - | - | 0.23 | 0.12 | - | - |
| **Aug 13** | 66.22 | - | 0.06 | 1.02 | 0.06 | - | 0.02 | 0.22 | - | - |
| **SE** | 25.14 | - | 0.06 | 0.58 | 0.04 | - | 0.02 | 0.18 | - | - |
| | | | | | | | | | | |
| **May 14** | 60.60 | 0.70 | 0.14 | 0.79 | - | - | 2.00 | 0.01 | - | - |
| **SE** | 24.88 | 0.70 | 0.14 | 0.41 | - | - | 0.81 | 0.01 | - | - |
| **Jun 14** | 60.02 | - | 0.08 | 22.66 | - | - | 2.77 | 0.02 | 0.29 | - |
| **SE** | 22.16 | - | 0.06 | 10.19 | - | - | 1.00 | 0.02 | 0.18 | - |
| **Aug 14** | 117.72 | - | 0.55 | 43.16 | - | 2.98 | - | 0.75 | 0.04 | - |
| **SE** | 58.27 | - | 0.42 | 17.95 | - | 2.98 | - | 0.41 | 0.04 | - |

Appendix Table 6. Mean plant biomass (g dry/m$^2$) for all plant species in all surveys in Lake Susan. Biomass was calculated using a littoral zone of 4.6 m. Definitions for abbreviations can be found in Appendix Table 1.

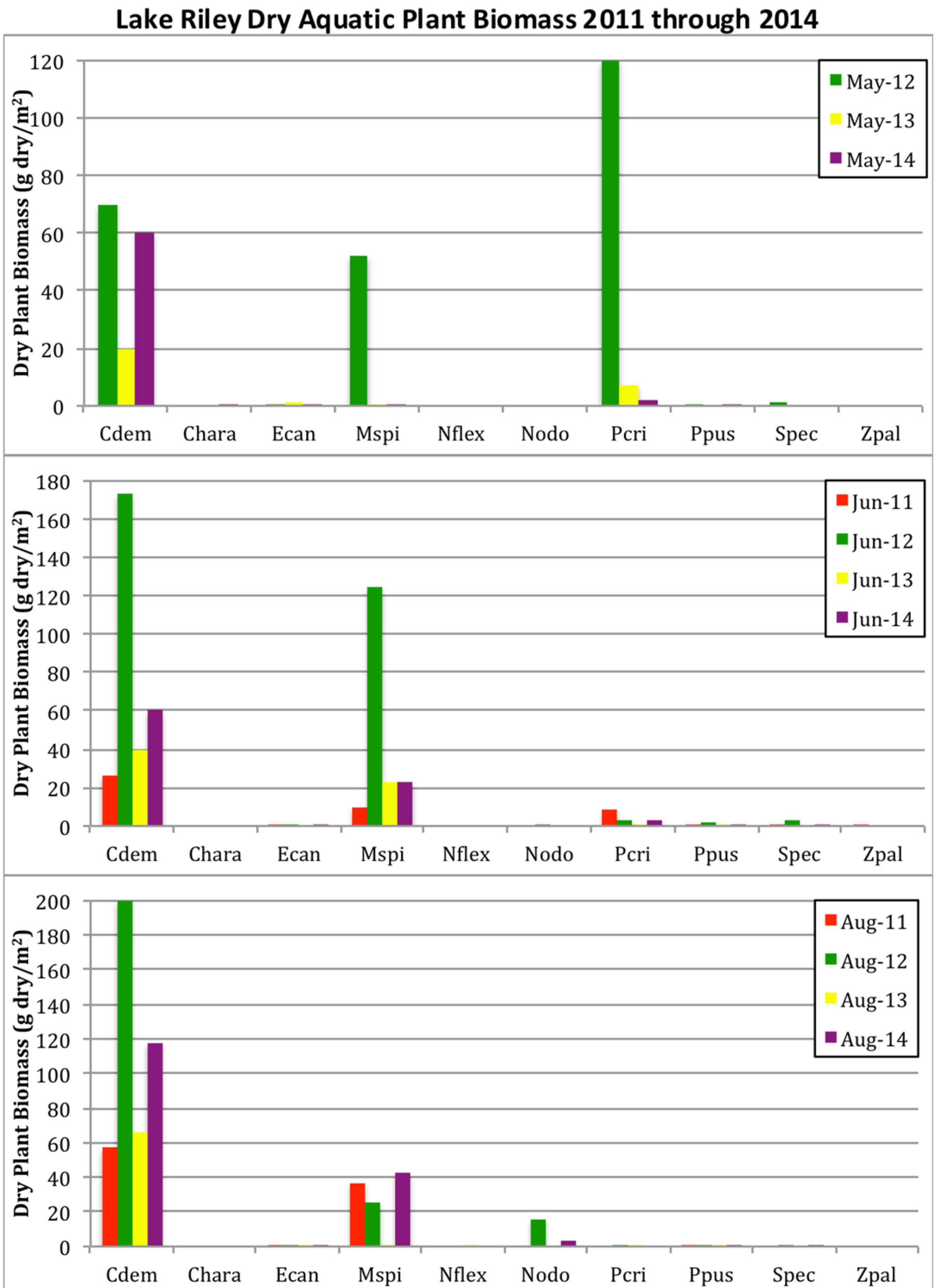
| | Cdem | Ecan | Ltri | Msib | Mspi | Nfle | Nlut | Nodo | Nvar | Pcri | Ppus | Pzos | Rlon | Spec | Zpal |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Jun 10** | 105.2 | 0.06 | - | - | 5.43 | - | - | 1.77 | 6.16 | 31.16 | 31.08 | - | - | 0.73 | - |
| **SE** | 38.11 | 0.06 | - | - | 2.44 | - | - | 1.77 | 5.74 | 14.05 | 15.31 | - | - | 0.73 | - |
| **Aug 10** | 98.63 | 0.03 | - | - | 0.63 | - | 1.87 | - | 1.94 | 0.09 | 10.76 | - | - | 2.52 | - |
| **SE** | 30.54 | 0.03 | - | - | 0.39 | - | 1.71 | - | 1.42 | 0.07 | 6.66 | - | - | 1.55 | - |
| **Sep 10** | 106.1 | - | - | - | 4.67 | - | 2.95 | 0.48 | 1.81 | 0.07 | - | - | - | - | - |
| **SE** | 36.72 | - | - | - | 3.90 | - | 2.50 | 0.48 | 1.81 | 0.05 | - | - | - | - | - |
| **May 11** | 35.27 | 0.41 | - | - | 2.72 | - | - | 0.15 | - | 2.62 | 0.14 | - | - | - | - |
| **SE** | 21.46 | 0.33 | - | - | 2.59 | - | - | 0.15 | - | 1.79 | 0.10 | - | - | - | - |
| **Jun 11** | 59.47 | 12.54 | - | - | 1.03 | - | - | - | 11.71 | 19.31 | 1.96 | - | - | 0.13 | - |
| **SE** | 26.70 | 7.69 | - | - | 0.72 | - | - | - | 9.66 | 8.88 | 0.95 | - | - | 0.09 | - |
| **Aug 11** | 73.60 | 73.21 | - | - | 1.43 | - | 1.33 | - | 10.75 | 1.00 | 28.01 | - | - | - | - |
| **SE** | 30.36 | 30.59 | - | - | 1.28 | - | 1.33 | - | 8.38 | 0.71 | 17.40 | - | - | - | - |
| **May 12** | 59.76 | 17.93 | - | - | 1.11 | - | 1.51 | - | 2.39 | 50.03 | 2.75 | - | - | 0.34 | - |
| **SE** | 30.68 | 15.57 | - | - | 1.05 | - | 1.10 | - | 2.39 | 21.99 | 2.75 | - | - | 0.26 | - |
| **Jun 12** | 29.59 | 6.27 | - | - | 0.94 | - | 15.84 | - | 15.14 | 5.95 | 5.48 | - | - | 1.15 | - |
| **SE** | 22.36 | 3.27 | - | - | 0.88 | - | 11.38 | - | 15.14 | 3.40 | 3.54 | - | - | 1.06 | - |
| **Aug 12** | 2.81 | 5.08 | 0.03 | - | 0.40 | - | 27.67 | 1.83 | 4.62 | 0.13 | 5.82 | - | - | - | - |
| **SE** | 1.86 | 3.87 | 0.03 | - | 0.40 | - | 15.77 | 1.83 | 4.62 | 0.09 | 5.76 | - | - | - | - |
| **May 13** | 1.24 | 0.08 | - | - | - | - | - | - | - | 6.47 | - | - | - | - | - |
| **SE** | 0.70 | 0.05 | - | - | - | - | - | - | - | 1.86 | - | - | - | - | - |
| **Jun 13** | 1.35 | 0.38 | 0.01 | 0.01 | - | - | - | 1.16 | 0.60 | 0.20 | 0.30 | - | - | - | - |
| **SE** | 0.49 | 0.19 | 0.01 | 0.01 | - | - | - | 0.73 | 0.60 | 0.14 | 0.14 | - | - | | - |
| **Aug 13** | 25.07 | 1.63 | - | - | - | - | 25.64 | 5.34 | 9.46 | 0.19 | 5.75 | - | - | 0.10 | - |
| **SE** | 11.50 | 1.06 | - | - | - | - | 10.73 | 2.70 | 9.46 | 0.18 | 3.18 | - | - | 0.10 | - |
| **May 14** | 22.86 | 1.67 | - | - | - | 0.01 | - | - | - | 1.29 | 0.18 | - | 0.01 | - | - |
| **SE** | 10.57 | 1.00 | - | - | - | 0.01 | - | - | - | 0.69 | 0.13 | - | 0.01 | - | - |
| **Jun14** | 23.15 | 15.46 | - | - | - | 0.05 | 1.25 | 1.10 | 5.78 | 1.02 | 0.11 | - | - | 0.11 | - |
| **SE** | 11.32 | 11.17 | - | - | - | 0.05 | 0.89 | 0.58 | 5.42 | 0.81 | 0.06 | - | - | 0.11 | - |
| **Aug 14** | 52.47 | 49.24 | 0.01 | - | - | 0.18 | 14.01 | 3.92 | 3.80 | - | 0.33 | 0.02 | - | 0.70 | 0.06 |
| **SE** | 23.49 | 26.67 | 0.01 | - | - | 0.12 | 5.98 | 1.96 | 3.80 | - | 0.25 | 0.02 | - | 0.65 | 0.06 |

Appendix Table 7. Mean plant biomass (g dry/$m^2$) for all plant species in all surveys in Mitchell Lake. Biomass was calculated using a littoral zone of 4.6 m. Definitions for abbreviations can be found in Appendix Table 1.

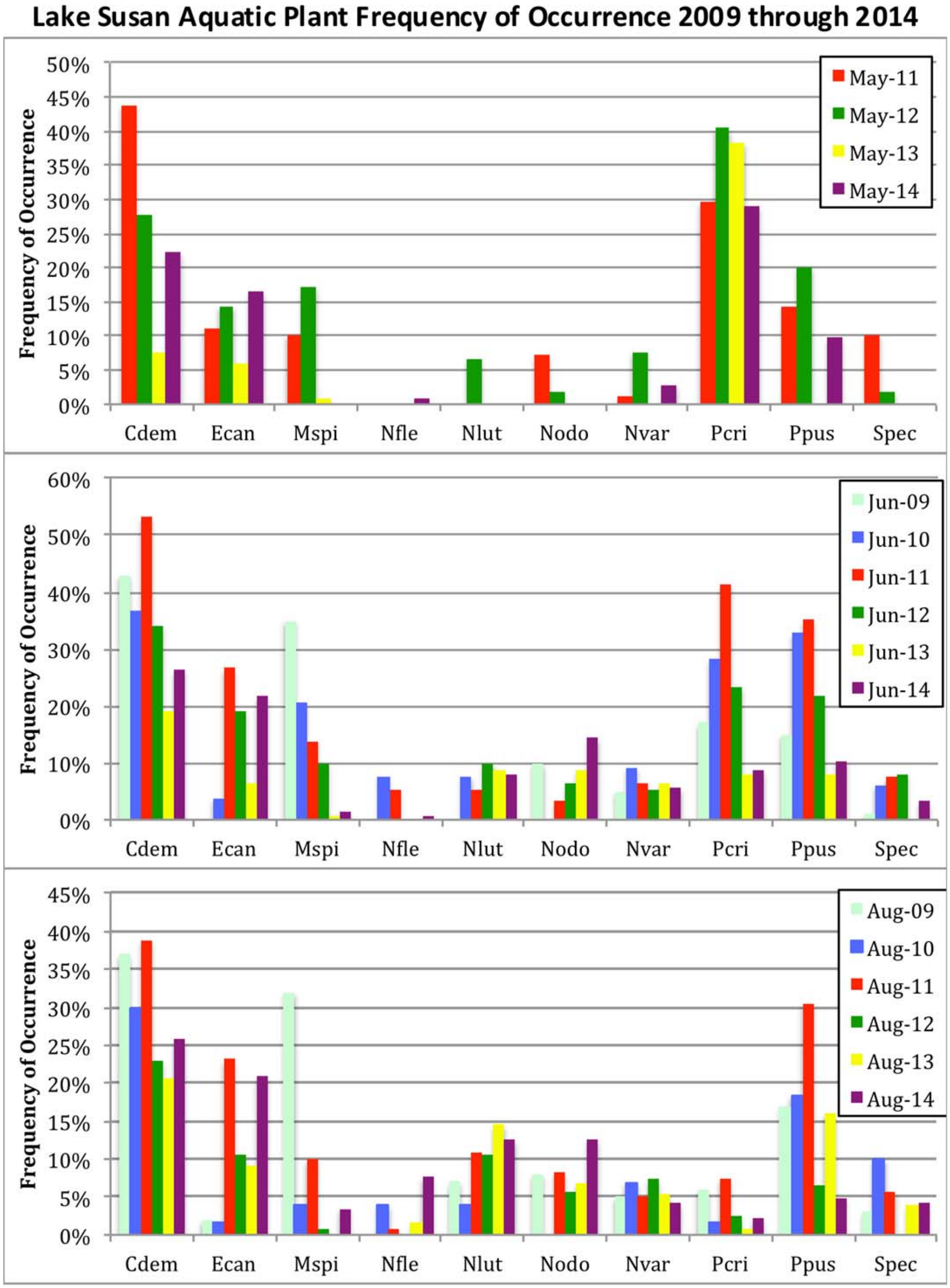
| | Cdem | Chara | Ecan | Ltri | Msib | Mspi | Nfle | Nlut | Nodo | Pcri | Ppus | Pzos | Rlon | Spec | Spol | Uvul | Zdub |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| **Jun 13** | 187.95 | 0.20 | 0.02 | 6.54 | 5.01 | 0.36 | - | - | - | 27.47 | 1.78 | 0.24 | - | - | - | - | 0.67 |
| **SE** | 38.36 | 0.11 | 0.02 | 1.76 | 2.61 | 0.29 | - | - | - | 6.44 | 0.70 | 0.11 | - | - | - | - | 0.67 |
| **Jun 13** | 187.03 | 0.11 | 0.09 | 10.19 | 13.22 | 0.12 | 0.01 | - | 6.07 | 20.70 | 8.92 | 1.07 | 1.29 | - | - | - | 1.56 |
| **SE** | 30.63 | 0.08 | 0.05 | 5.89 | 4.54 | 0.12 | 0.01 | - | 3.40 | 10.78 | 4.70 | 0.56 | 0.69 | - | - | - | 0.63 |
| **Aug 13** | 704.75 | - | 0.16 | 11.55 | 37.60 | 1.47 | - | | 51.64 | - | - | 7.78 | 2.37 | 0.02 | - | - | 4.42 |
| **SE** | 117.44 | - | 0.09 | 3.67 | 11.47 | 1.25 | - | | 21.34 | - | - | 3.96 | 1.49 | 0.02 | - | - | 3.41 |
| **May 14** | 283.44 | - | 0.05 | 9.29 | 1.29 | - | 0.02 | - | - | 5.72 | 1.25 | 0.30 | 0.18 | - | - | 0.02 | 0.04 |
| **SE** | 54.92 | - | 0.03 | 2.24 | 0.62 | - | 0.02 | - | - | 1.91 | 0.46 | 0.15 | 0.09 | - | - | 0.02 | 0.02 |
| **Jun 14** | 349.88 | 0.11 | 0.08 | 11.72 | 5.66 | - | - | 0.40 | 1.74 | 13.97 | 6.10 | 2.83 | 1.15 | 0.43 | - | 0.06 | 1.15 |
| **SE** | 50.94 | 0.11 | 0.05 | 2.47 | 2.50 | - | - | 0.40 | 1.74 | 4.79 | 2.30 | 1.22 | 0.61 | 0.29 | - | 0.06 | 0.91 |
| **Aug 14** | 287.48 | - | - | 7.74 | 16.93 | 0.64 | - | 5.34 | 16.45 | 0.02 | 1.22 | 9.56 | 0.35 | 0.15 | 0.04 | - | 1.63 |
| **SE** | 42.21 | - | - | 2.25 | 6.07 | 0.64 | - | 5.34 | 11.42 | 0.02 | 0.88 | 3.75 | 0.19 | 0.12 | 0.03 | - | 1.09 |



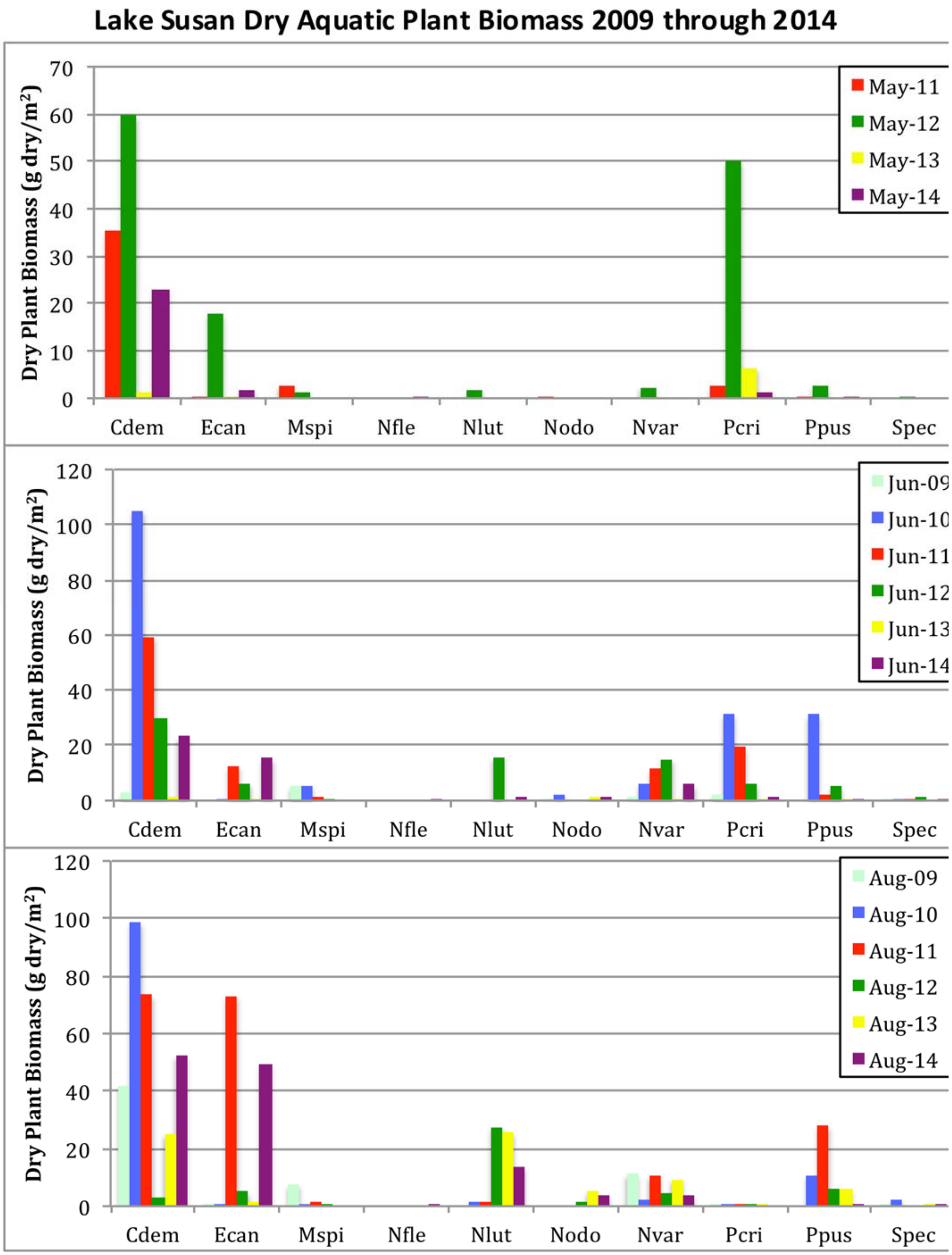

Appendix Figure 1. Frequency of occurrence for Lake Riley surveys May, June and August 2011, 2012, 2013 and 2014. Frequencies were calculated using 4.6 m littoral zone depth. Definitions for abbreviations can be found in Appendix Table 1.



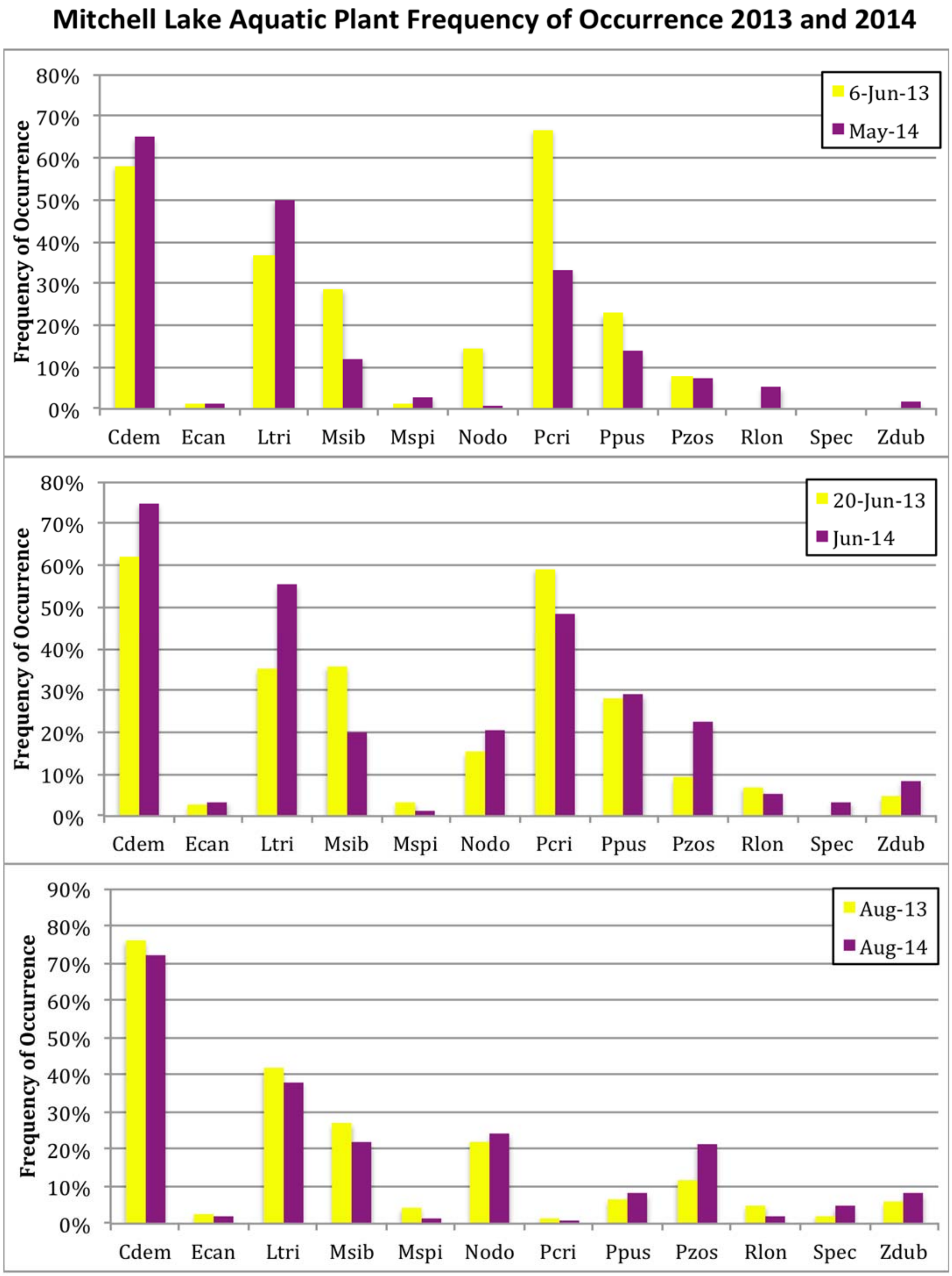
Appendix Figure 2. Dry aquatic plant biomass (g dry/$m^2$) for Lake Riley surveys May, June and August 2011, 2012, 2013 and 2014. Biomass was calculated using 4.6 m littoral zone depth. Definitions for abbreviations can be found in Appendix Table 1.



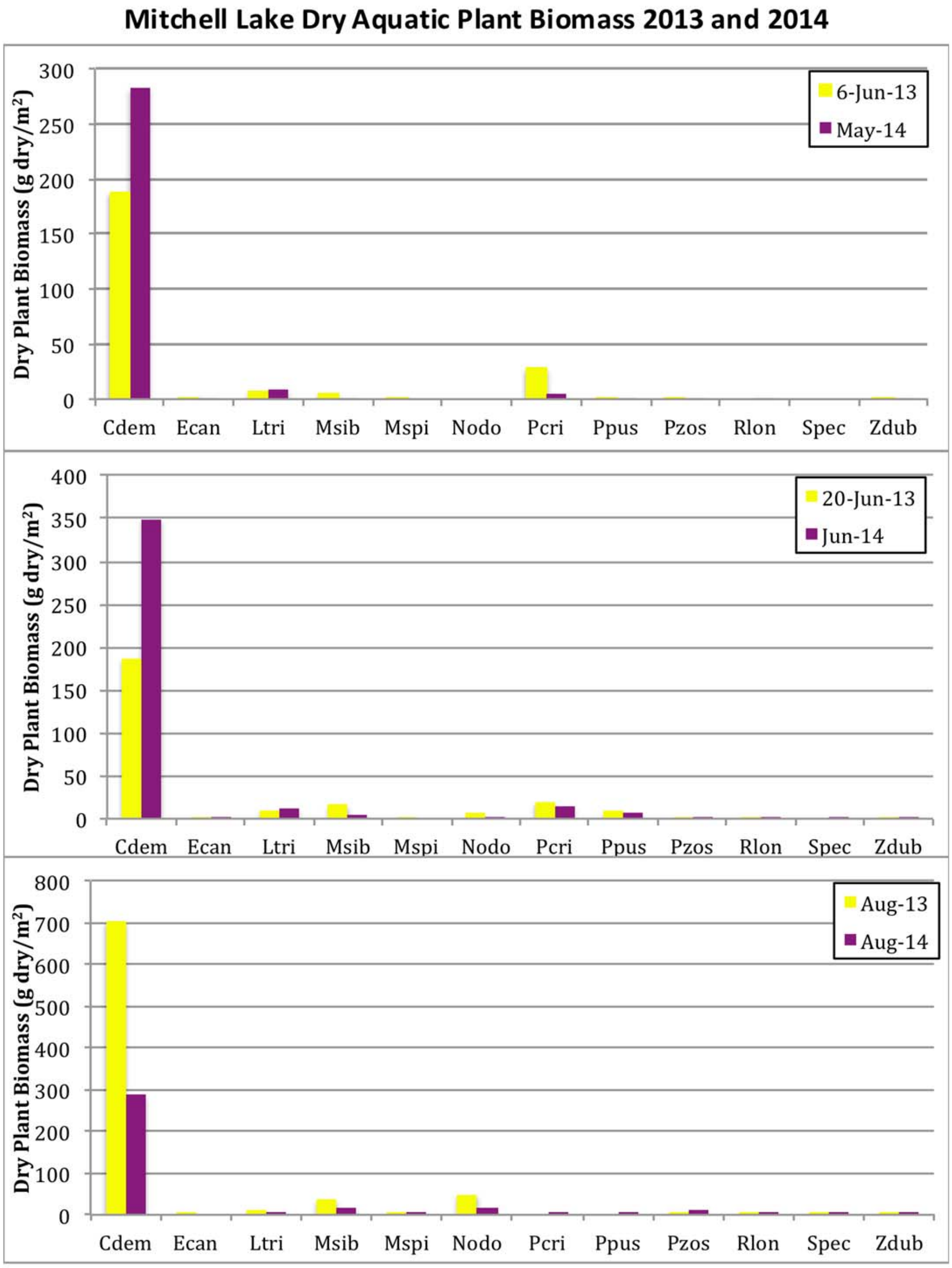

Appendix Figure 3. Frequency of occurrence for Lake Susan surveys May, June and August 2009, 2010, 2011, 2012, 2013 and 2014. Frequencies were calculated using 4.6 m littoral zone depth. Definitions for abbreviations can be found in Appendix Table 1.



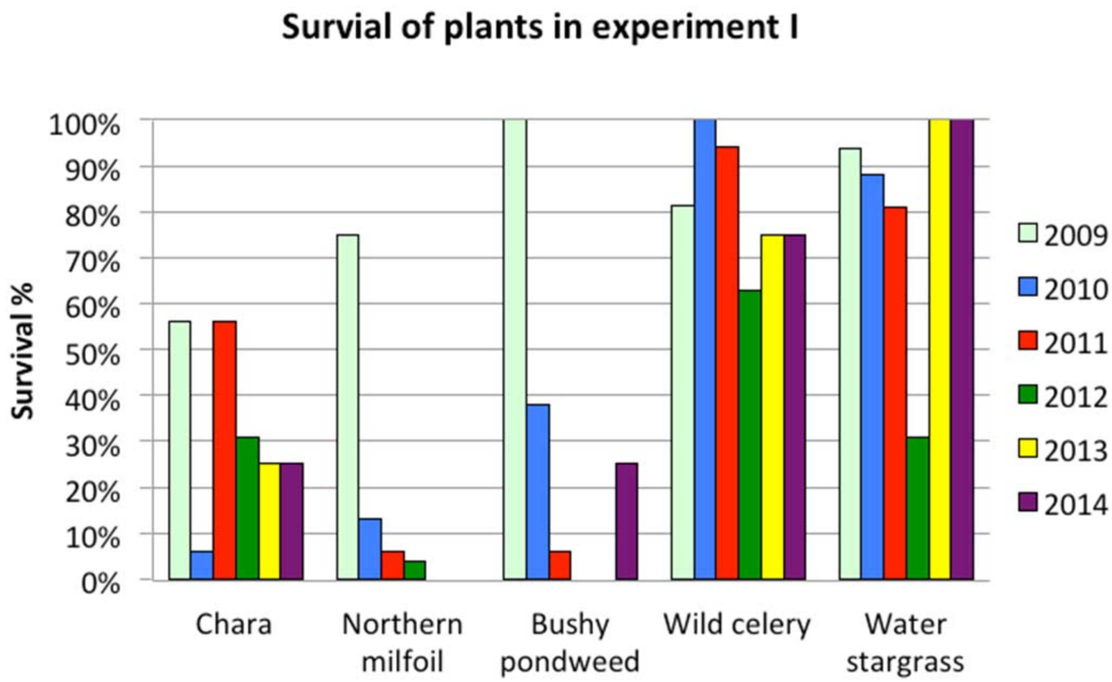

Appendix Figure 4. Dry Aquatic Plant Biomass (g dry/$m^2$) for Lake Susan surveys May, June and August 2009, 2010, 2011, 2012, 2013 and 2014. Biomass was calculated using 4.6 m littoral zone depth. Definitions for abbreviations can be found in Appendix Table 1.



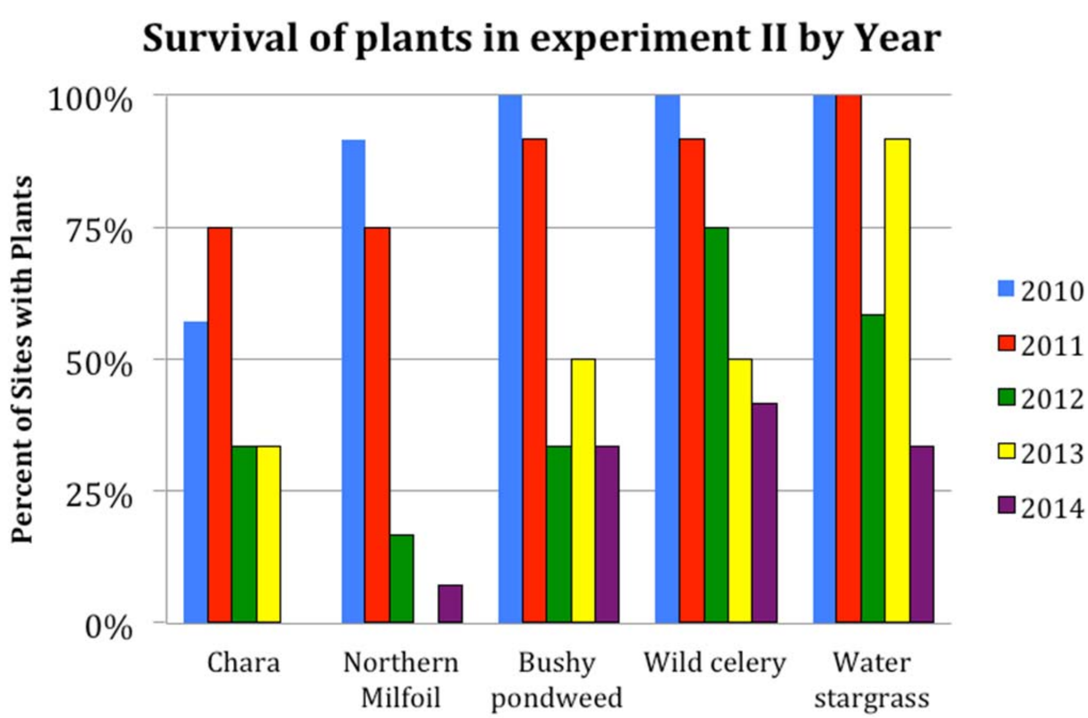

Appendix Figure 5. Frequency of occurrence for Mitchell Lake surveys May, June and August 2013 and 2014. Frequencies were calculated using 4.6 m littoral zone depth. Note there was two separate surveys were conducted in June 2013. Definitions for abbreviations can be found in Appendix Table 1.




Appendix Figure 6. Dry Aquatic Plant Biomass (g dry/m$^2$) for Mitchell Lake surveys May, June and August 2013 and 2014. Biomass was calculated using 4.6 m littoral zone depth. Note there was two separate surveys conducted in June 2013. Definitions for abbreviations can be found in Appendix Table 1.




Appendix Figure 7. The survival of transplants by year for experiment I in Lake Susan.




Appendix Figure 8. The survival of transplants by year for experiment II in Lake Susan.