Long Lake

Vilas County, Wisconsin

Aquatic Plant Management Plan Update

June 2019

Sponsored by:

Long Lake of Phelps Lake District

WDNR Surface Water Grant Program

ACEI-132-13

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Onterra, LLC
Lake Management Planning

Long Lake

Vilas County, Wisconsin

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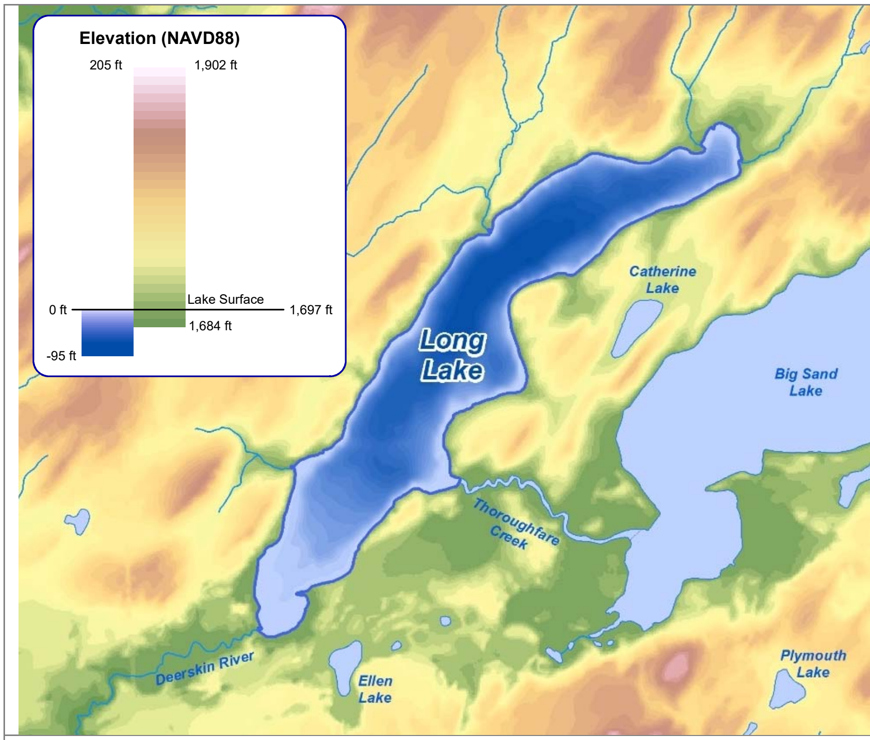


Figure 1.0-1. Long Lake, Vilas County.

According to the 1961 Wisconsin Department of Natural Resources (WDNR) echo sounding Lake Survey Map, Long Lake is 861.5 acres. The WDNR website list the lake as 886 acres. At the time of this report, the most current orthophoto (aerial photograph) was from the *National Agriculture Imagery Program (NAIP*) collected in summer 2017. Based upon heads-up digitizing the water level from that photo, the lake was determined to be 889.3 acres. Long Lake, Vilas County, is a lowland, two-story, drainage lake with a maximum depth of 95 feet. This mesotrophic lake has a relatively large watershed (>14,000 acres) when compared to the size of the lake (15:1). Long Lake contains 50 native plant species, of which southern naiad is currently the most common plant. One submergent non-native plant (Eurasian watermilfoil), and two shoreland emergent non-native plants (sweet flag and reed canary grass) have been identified from Long Lake.

Connected via the approximately 1.25-mile-long Thoroughfare Creek, Big Sand Lake flows into Long Lake (Figure 1.0-1). Big Sand Lake is arguably the first lake in Vilas County to contain Eurasian watermilfoil, with official records of this plant occurring in the lake in 1990. Long Lake is drained by the Deerskin River which flows into Scattering Rice Lake of the Eagle River Chain of Lakes.

In 2000 the presence of Eurasian watermilfoil was verified by the Wisconsin Department of Natural Resources (WDNR) from Long Lake, although it was suspected of inhabiting the system for years before this date. In 2006, the WDNR completed a point-intercept aquatic plant survey, locating Eurasian watermilfoil in approximately 26% of the littoral area of the lake (< 18ft). During that timeframe the Long Lake of Phelps Lake District (LLPLD) was in the process of creating a lake management plan for the system with the aid of Northern Environmental, Inc. This plan was finalized in June 2007.

The LLPLD contracted with Onterra, LLC during the late-summer of 2017 to develop a management strategy for the increasing EWM population within Long Lake. With the assistance of Onterra, the LLPLD secured a Wisconsin Department of Natural Resources (WDNR) Aquatic Invasive Species (AIS) Established Population Control Grant to fund an EWM population management strategy from 2008-2012.

Following the conclusion of the 2008-2012 EWM Control Project, the LLPLD commenced the creation of a lake management plan to reflect the success and limitations learned over the past four years. A six-member Planning Committee served as the focus group for the project. Along with establishing thresholds (triggers) of when specific control strategies warrant implementation, the lake management planning process provided for a holistic understanding of the Long Lake ecosystem involving assessments of the water quality, watershed, shoreline condition, fisheries data integration, and stakeholder perceptions of Long Lake. The *Long Lake Comprehensive Management Plan Update* was finalized and approved by the WDNR in July 2013.

After a failed attempt in August 2012, another WDNR AIS Established Population Control Grant was awarded in February 2013 to fund EWM management and monitoring from 2013-2017. Remaining funds from the grant allowed the project to extend to 2018. This report serves as the final deliverable for this grant-funded project (ACEI-132-13).

2.0 AQUATIC PLANTS

2.1 Primer on Aquatic Plant Data Analysis & Interpretation

Photo 2.1-1. Native aquatic plants.

Native aquatic plants are an important element in every healthy aquatic ecosystem, providing food and habitat to wildlife, improving water quality, and stabilizing bottom sediments (Photo 2.1-1). Because most aquatic plants are rooted in place and are unable to relocate in wake of environmental alterations, they are often the first community to indicate that changes may be occurring within the system. Aquatic plant communities can respond in a variety of ways; there may be increases or declines in the occurrences of some species, or a complete loss. Or, certain growth forms, such as emergent and floating-leaf communities may disappear from certain areas of the waterbody. With periodic monitoring and proper analysis, these changes are relatively easy to detect and provide relevant information for making management decisions.

The point-intercept method as described Wisconsin Department of Natural Resources Bureau of Science Services, PUB-SS-1068 2010 (Hauxwell et al. 2010) have been conducted by the WDNR in 2006 and by Onterra in 2012 and 2017. Based upon guidance from the WDNR, a point spacing (resolution) of 47 meters was used resulting in 1,616 sampling points being evenly distributed across the lake (Map 1). At each point-intercept location within the *littoral zone*, information regarding the depth, substrate type (soft sediment, sand, or rock), and the plant species sampled along with their relative abundance (Figure 2.1-1) on the sampling rake was recorded.

A pole-mounted rake was used to collect the plant samples, depth, and sediment information at point locations of 15 feet or less. A rake head tied to a rope (rope rake) was used at sites greater than 15 feet. Depth information was collected using graduated marks on the pole of the rake (at depths < 15 ft) or using an onboard sonar unit (at depths > 15 feet). Also, when a rope rake was used, information regarding substrate type was not collected due to the inability of the sampler to accurately "feel" the bottom with this sampling device. The point-intercept survey produces a great deal of information about a lake's aquatic vegetation and overall health. These data are analyzed and presented in numerous ways; each is discussed in more detail the following section.

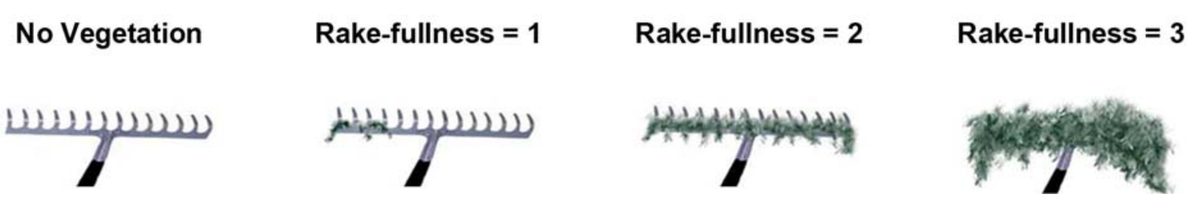


Figure 2.1-1. Aquatic plant rake-fullness ratings. Adapted from Hauxwell et al (2010).

Species List

The species list is simply a list of all of the aquatic plant species, both native and non-native, that were located during the surveys completed in Long Lake in 2017. The list also contains the growth-form of each plant found (e.g. submergent, emergent, etc.), its scientific name, common name, and its coefficient of conservatism. The latter is discussed in more detail below. Changes in this list over time, whether it is differences in total species present, gains and losses of individual species, or changes in growth forms that are present, can be an early indicator of changes in the ecosystem.

Frequency of Occurrence

Littoral Zone is the area of a lake where sunlight is able to penetrate down to the sediment and support aquatic plant growth.

Frequency of occurrence describes how often a certain aquatic plant species is found within a lake. Obviously, all of the plants cannot be counted in a lake, so samples are collected from pre-determined areas. In the case of the whole-lake point-intercept survey completed on Long Lake, plant samples were collected from plots laid out on a grid that covered the lake. Using the data collected from these plots, an estimate of occurrence of each plant species can be determined. The occurrence of aquatic plant species is displayed as the *littoral frequency of occurrence*. Littoral frequency of occurrence is used to describe how often each species occurred in the plots that are within the maximum depth of plant growth (littoral zone), and is displayed as a percentage.

Floristic Quality Assessment

The floristic quality of a lake's aquatic plant community is calculated using its native *species richness* and their *average conservatism*. Species richness is the number of native aquatic plant species that were physically encountered on the rake during the point-intercept survey. Average conservatism is calculated by taking the sum of the coefficients of conservatism (C-values) of the native species located and dividing it by species richness. Every plant in Wisconsin has been assigned a coefficient of conservatism, ranging from 1-10, which describes the likelihood of that species being found in an undisturbed environment. Species which are more specialized and require undisturbed habitat are given higher coefficients, while species which are more tolerant of environmental disturbance have lower coefficients.

For example, algal-leaf pondweed (*Potamogeton confervoides*) is only found in nutrient-poor, acid lakes in northern Wisconsin and is prone to decline if degradation of these lakes occurs. Because of algal-leaf pondweed's special requirements and sensitivity to disturbance, it has a C-value of 10. In contrast, sago pondweed (*Stuckenia pectinata*) with a C-value of 3, is tolerant of disturbance and is often found in greater abundance in degraded lakes that have higher nutrient concentrations and low water clarity. Higher average conservatism values generally indicate a healthier lake as it is able to support a greater number of environmentally-sensitive aquatic plant species. Low average conservatism values indicate a degraded environment, one that is only able to support disturbance-tolerant species.

On their own, the species richness and average conservatism values for a lake are useful in assessing a lake's plant community; however, the best assessment of the lake's plant community health is determined when the two values are used to calculate the lake's floristic quality. The floristic quality is calculated using the species richness and average conservatism value of the aquatic plant species that were solely encountered on the rake during the point-intercept surveys (equation shown below). This assessment allows the aquatic plant community of Long Lake to be compared to other lakes within the region and state.

$$FQI = \text{Average Coefficient of Conservatism} * \sqrt{\text{Number of Native Species}}$$

Long Lake falls within the Northern Lakes and Forests (NLF) *ecoregion* (Figure 2.1-2), and the floristic quality of its aquatic plant community will be compared to other lakes within this ecoregion as well as the entire State of Wisconsin. Ecoregions are areas related by similar climate, physiography, hydrology, vegetation and wildlife potential. Comparing ecosystems within the same ecoregion is sounder than comparing systems within manmade boundaries such as counties, towns, or states. Ecoregional and state-wide medians were calculated from whole-lake point-intercept surveys conducted on 392 lakes throughout Wisconsin by Onterra and WDNR ecologists.

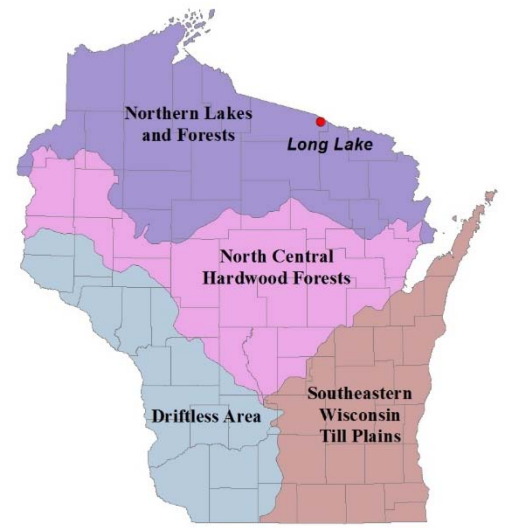


Figure 2.1-2. Location of Long Lake within the ecoregions of Wisconsin. After Nichols 1999.

Species Diversity

Species diversity is often confused with species richness. As defined previously, species richness is simply the number of species found within a given community. While species diversity utilizes species richness, it also takes into account evenness or the variation in abundance of the individual species within the community. For example, a lake with 10 aquatic plant species that had relatively similar abundances within the community would be more diverse than another lake with 10 aquatic plant species were 50% of the community was comprised of just one or two species.

An aquatic system with high species diversity is more stable than a system with a low diversity. This is analogous to a diverse financial portfolio in that a diverse aquatic plant community can withstand environmental fluctuations much like a diverse portfolio can handle economic fluctuations. Some managers believe a lake with a diverse plant community is also better suited to compete against exotic infestations than a lake with a lower diversity. However, in a recent study of 1,100 Minnesota lakes, researches concluded that more diverse communities were not more resistant or resilient to invaders (Muthukrishnan et al. 2018).

The diversity of a lake's aquatic plant community is determined using the Simpson's Diversity Index (1-D):

$$D = \sum (n/N)^2$$

where:
n = the total number of instances of a particular species
N = the total number of instances of all species
D is a value between 0 and 1

If a lake has a diversity index value of 0.90, it means that if two plants were randomly sampled from the lake there is a 90% probability that the two individuals would be of a different species. The Simpson's Diversity Index value from Long Lake is compared to data collected by Onterra and the WDNR Science Services on 212 lakes within the Northern Lakes and Forests (lakes only, does not include flowages) Ecoregion and on 392 lakes throughout Wisconsin.

Community Mapping

A key component of any aquatic plant community assessment is the delineation of the emergent and floating-leaf aquatic plant communities within each lake as these plants are often underrepresented during the point-intercept survey. This survey creates a snapshot of these important communities within each lake as they existed during the survey and is valuable in the development of the management plan and in comparisons with future surveys. Examples of emergent plants include cattails, rushes, sedges, grasses, bur-reeds, and arrowheads, while examples of floating-leaf species include the water lilies. The emergent and floating-leaf aquatic plant communities in Long Lake were mapped using a Trimble Global Positioning System (GPS) with sub-meter accuracy.

2.2 Long Lake Aquatic Plant Survey Results

The aquatic plant point-intercept and aquatic plant community mapping surveys were conducted on Long Lake on July 25-26, 2017 by Onterra. During these surveys, 53 species of aquatic plants were located in Long Lake (Table 2.2-1), three of which are considered to be non-native: Eurasian watermilfoil (*Myriophyllum spicatum*), sweetflag (*Acorus calamus*), and reed canary grass (*Phalaris arundinacea*). Reed canary grass is distributed widely across Wisconsin and was identified in a few shoreland locations on Long Lake. Sweet flag is an emergent wetland species that is thought to have been introduced and spread around Wisconsin by early Native American communities. While non-native, sweet flag typically does not display invasive tendencies and is often found in healthy ecosystems. The population of Eurasian watermilfoil (EWM) is discussed in detail within the subsequent section (Section 2.3).

Table 2.2-1 includes the list of aquatic plant species which were located during surveys completed in 2006, 2012, and 2017. A comparison of the 2017 aquatic plant survey data to these historical datasets is discussed later in this section. Appendix A contains the full matrix of aquatic plant frequencies from point-intercept surveys.

Like terrestrial plants, different aquatic plant species are adapted to grow in certain substrate types; some species are only found growing in soft substrates, others only in sandy areas, and some can be found growing in either. Lakes that have varying substrate types generally support a higher number of plant species because of the different habitat types that are available. In early November 2014, Onterra ecologists completed an acoustic survey on Long Lake. While the survey was primarily aimed at gathering bathymetry data (depth contours), data pertaining to Long Lake's substrate composition were also recorded during this survey. The sonar records substrate hardness, ranging from the hardest substrates (i.e. rock and sand) to the more flocculent, softer organic sediments. These data are then modeled using spatial interpolation techniques. Please note that these data are not ground-truthed and the accuracy of some data, especially in shallow water, is unknown.

Data regarding substrate hardness collected during the 2014 acoustic survey revealed that Long Lake's nearshore had the highest variability with less variance in sediment hardness as water depth increased (Figure 2.2-2, bottom frame). The top frame of Figure 2.2-2 shows that soft sediments are more prevalent in front of the Thoroughfare Creek from Big Sand Lake as well as the southern basin of Long Lake.

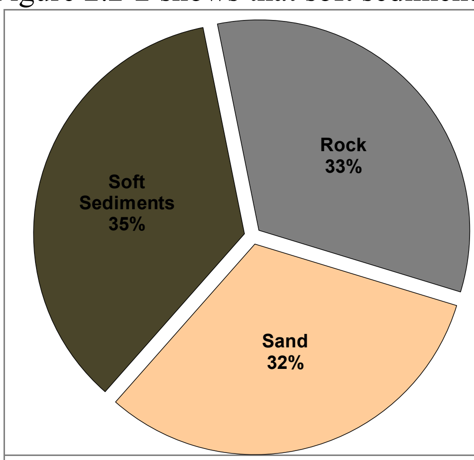
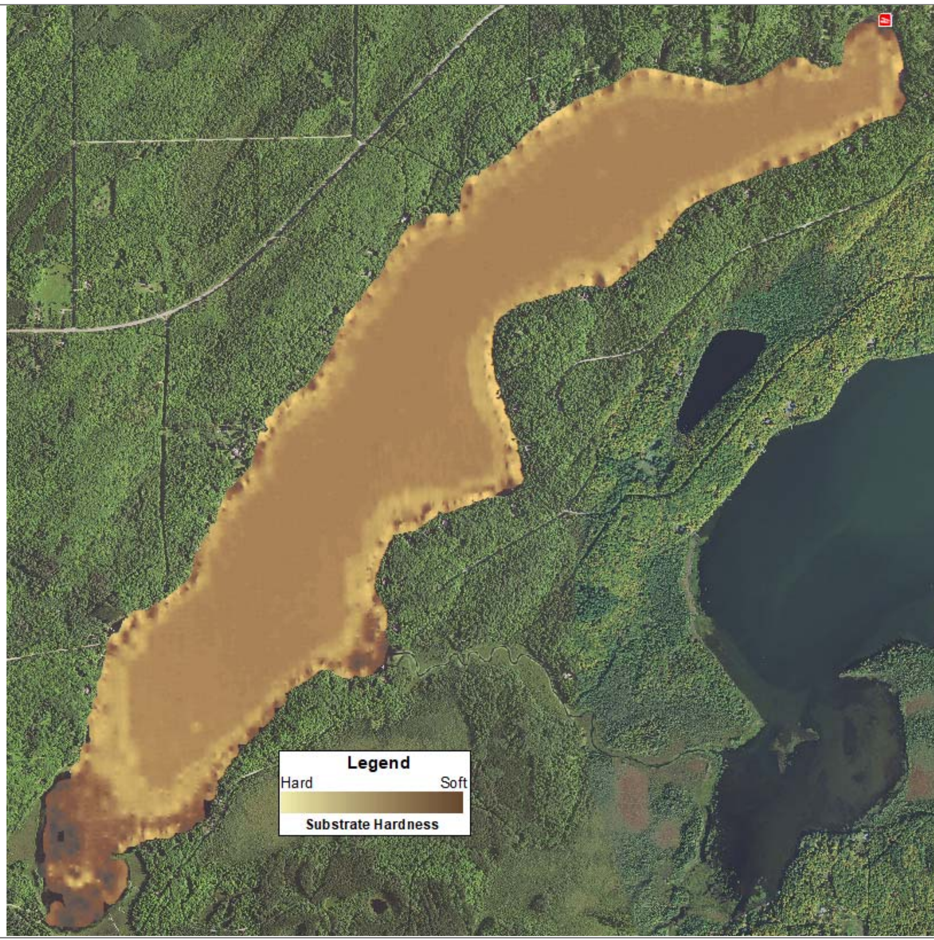


Figure 2.2-1. Long Lake proportion of substrate types. Created using data from 2017 point-intercept survey.

While not as comprehensive as the acoustic-based modeling method, sediment hardness data was also collected as part of that point-intercept method. At each point-intercept sampling location that was sampled with a pole rake (approx. 15 feet or less), sediment was categorized as either soft (i.e. muck), rock, or sand. Within this subset of points, 33% of the point-intercept sampling locations contained rock, 35% contained fine, organic matter (muck), while the remaining 32% contained sand (Figure 2.2-1). The sediment data collected is extremely similar to the data collected in 2006 and 2012.



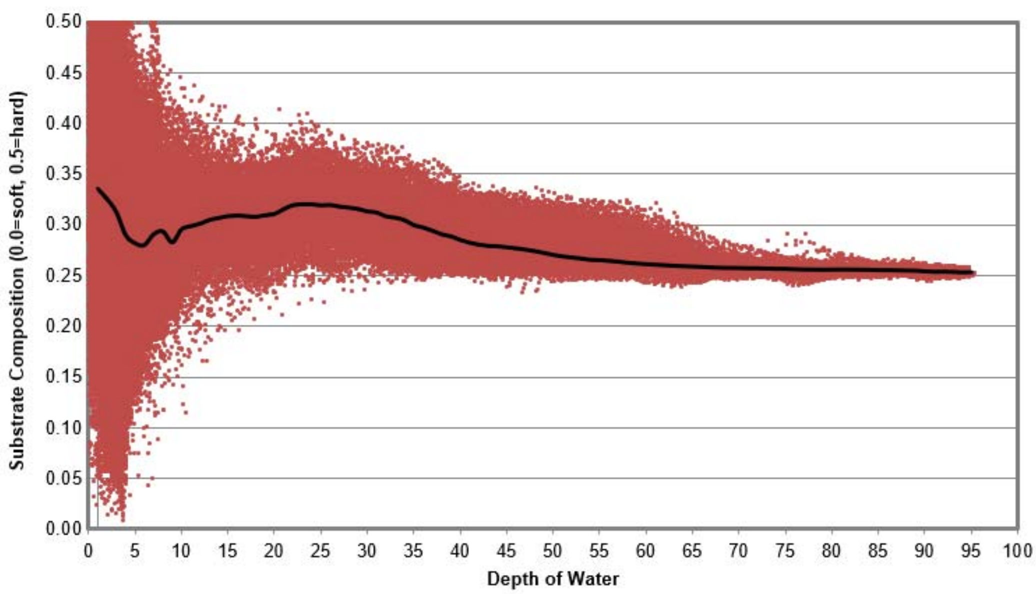


Figure 2.2-2. Long Lake spatial distribution of substrate hardness (top) and substrate hardness across water depth (bottom). Individual data points are displayed in red. Modeled using data from November 2014 acoustic survey.

Table 2.2-1. Aquatic plant species located on Long Lake during 2006, 2012, and 2017 surveys.

Growth Form	Scientific Name	Common Name	Coefficient of Conservatism (C)	2006 (WDNR)	2012 (Onterra)	2017 (Onterra)
Emergent	*Acorus calamus*	Sweetflag	Exotic		I	I
	Calamagrostis canadensis	Bluejoint grass	5			I
	Calla palustris	Water arum	9		I	I
	Carex retrorsa	Retrorse sedge	6		I	
	Carex utriculata	Common yellow lake sedge	7		I	
	Decodon verticillatus	Water-willow	7			I
	Dulichium arundinaceum	Three-way sedge	9			I
	Eleocharis palustris	Creeping spikerush	6	X	X	X
	Equisetum fluviatile	Water horsetail	7		X	I
	Iris sp.	Iris sp.	N/A			I
	Phalaris arundinacea	Reed canary grass	Exotic			I
	Sagittaria latifolia	Common arrowhead	3		I	I
	Schoenoplectus acutus	Hardstem bulrush	5	X	X	X
	Schoenoplectus tabernaemontani	Softstem bulrush	4	X	X	
	Scirpus cyperinus	Wool grass	4		I	I
	Sparganium americanum	Eastern bur-reed	8		X	
	Sparganium androcladum	Shining bur-reed	8			X
	Sparganium eurycarpum	Common bur-reed	5		I	I
	Typha spp.	Cattail spp.	1			I
FL	*Brasenia schreberi*	Watershield	7	X	X	X
	Nuphar variegata	Spatterdock	6	X	X	X
	Nymphaea odorata	White water lily	6	X	X	X
	Persicaria amphibia	Water smartweed	5			I
	Sparganium angustifolium	Narrow-leaf bur-reed	9		I	X
	Sparganium fluctuans	Floating-leaf bur-reed	10			I
FL/E	*Sparganium emersum* var. *acaule*	Short-stemmed bur-reed	8		I	X
Submergent	*Bidens beckii*	Water marigold	8	X	X	X
	Ceratophyllum demersum	Coontail	3	X	X	X
	Chara spp.	Muskgrasses	7	X	X	X
	Elodea canadensis	Common waterweed	3	X	X	X
	Heteranthera dubia	Water stargrass	6	X	X	X
	Isoetes spp.	Quillwort spp.	8	X	X	X
	Myriophyllum sibiricum	Northern watermilfoil	7	X	X	X
	Myriophyllum spicatum	Eurasian watermilfoil	Exotic	X	X	X
	Najas flexilis	Slender naiad	6	X	X	X
	Najas guadalupensis	Southern naiad	7		X	X
	Nitella spp.	Stoneworts	7	X	X	X
	Potamogeton alpinus	Alpine pondweed	9		I	
	Potamogeton amplifolius	Large-leaf pondweed	7	X	X	X
	Potamogeton epihydrus	Ribbon-leaf pondweed	8	X		
	Potamogeton foliosus	Leafy pondweed	6	X	X	X
	Potamogeton friesii	Fries' pondweed	8			X
	Potamogeton gramineus	Variable-leaf pondweed	7	X	X	X
	Potamogeton hybrid 1	Pondweed Hybrid 1	N/A		X	I
	Potamogeton natans	Floating-leaf pondweed	5	X	X	
	Potamogeton praelongus	White-stem pondweed	8	X	X	X
	Potamogeton pusillus	Small pondweed	7	X	X	X
	Potamogeton richardsonii	Clasping-leaf pondweed	5	X	X	X
	Potamogeton robbinsii	Fern-leaf pondweed	8	X	X	X
	Potamogeton spirillus	Spiral-fruited pondweed	8	X	X	X
	Potamogeton strictifolius	Stiff pondweed	8	X	X	X
	Potamogeton zosteriformis	Flat-stem pondweed	6	X	X	X
	Ranunculus aquatilis	White water crowfoot	8			X
	Ranunculus flammula	Creeping spearwort	9	X	X	X
	Sagittaria sp. (rosette)	Arrowhead sp. (rosette)	N/A	X		X
	Utricularia intermedia	Flat-leaf bladderwort	9		I	
	Utricularia minor	Small bladderwort	10			X
	Utricularia vulgaris	Common bladderwort	7	X	X	X
	Vallisneria americana	Wild celery	6	X	X	X
S/E	*Eleocharis acicularis*	Needle spikerush	5	X	X	X
	Juncus pelocarpus	Brown-fruited rush	8		X	X

FL = Floating Leaf; FL/E = Floating Leaf and Emergent; S/E = Submergent and Emergent
X = Located on rake during point-intercept survey; I = Incidental Species

Aquatic plants were found growing to a depth of 17 feet during the 2017 point-intercept survey, which was similar to the 18 feet that aquatic plants grew out to in 2006 and 2012. Of the 382 point-intercept sampling locations that fell at or within the maximum plant growth in 2017, approximately 79% of them contained aquatic vegetation. This compares to 69% in 2006 and 80% in 2012.

Aquatic plant rake fullness data collected in 2017 indicates that 45% of the 382 sampling locations contained vegetation with a total rake fullness rating (TRF) of 1, 19% had a TRF rating of 2, and 15% had a TRF rating of 3 (Figure 2.2-3). The total rake fullness ratings indicate that where plants occurred Long Lake in 2017 they were of moderately low biomass.

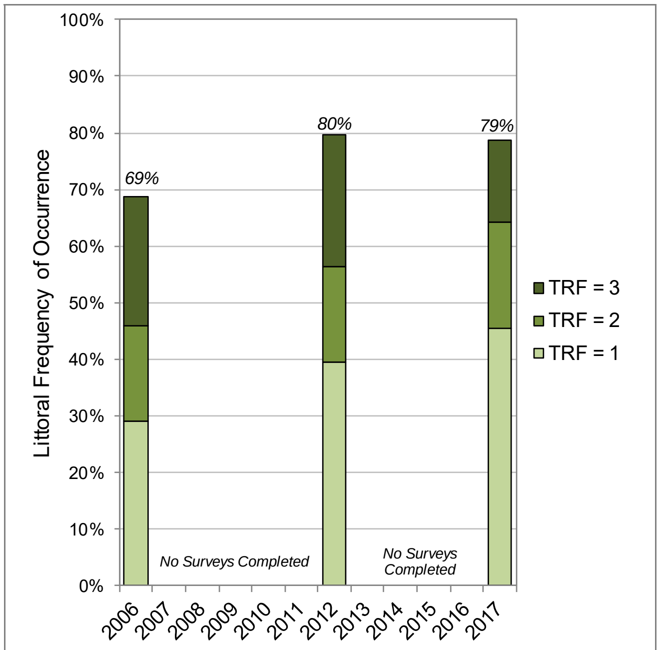


Figure 2.2-3. Aquatic plant frequency of occurrence and total rake fullness (TRF) ratings in Long Lake from the 2006, 2012, and 2017 surveys.

Of the 40 aquatic plant species recorded on the rake during the 2017 point-intercept survey, southern naiad, common waterweed, and muskgrasses were the three-most frequently encountered species (Figure 2.2-4).

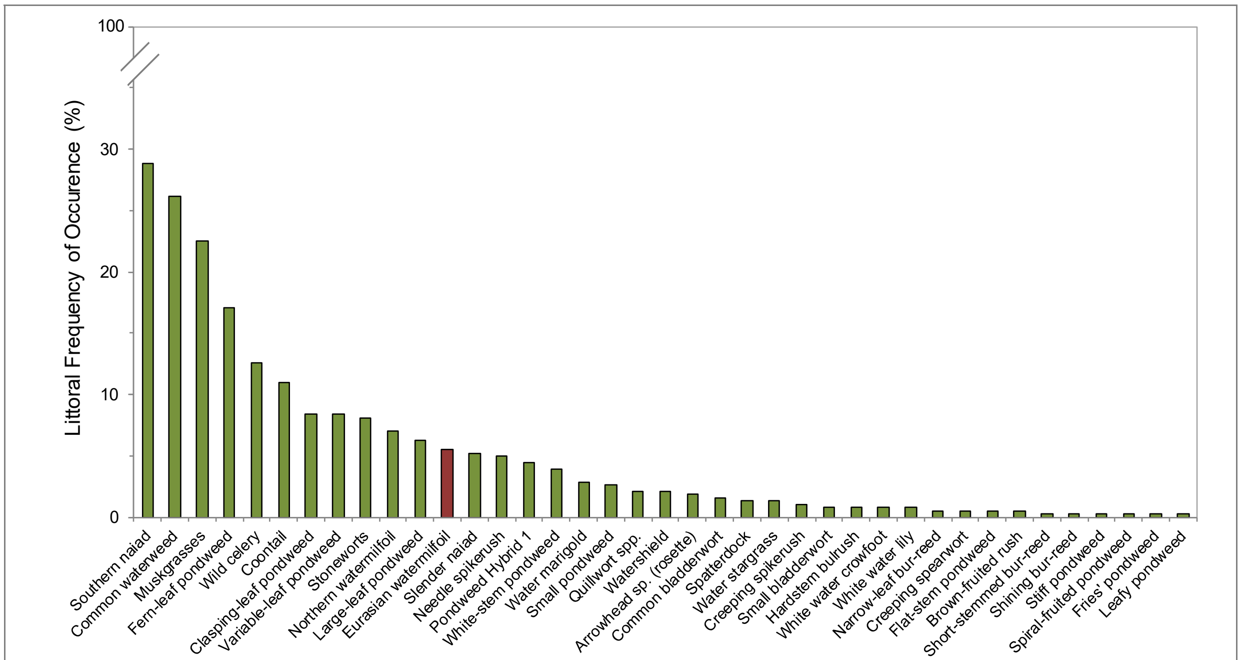


Figure 2.2-4. 2017 Littoral frequency of occurrence of Aquatic Plants in Long Lake.

Southern naiad was not recorded during the 2006 surveys and was first recorded during the 2012 point-intercept survey with a LFOO of approximately 1.2% (Figure 2.2-5). It was the most frequently encountered aquatic plant in 2017 with a LFOO of 28.8% (Figure 2.2-4). Southern naiad is similar to slender naiad, a native plant also found in Long Lake (Photo 2.2-1). While southern naiad is native to North America, its invasive behavior in nearby Big Sand Lake and other area lakes may indicate that this species was not historically present in these waterbodies and was recently introduced. Or, if southern naiad was historically present in these waterbodies, recent environmental conditions are favoring the rapid expansion of this plant within these lakes. Southern naiad has been shown to be particularly susceptible to some herbicides (e.g. fluridone) whereas relatively resilient to weak-auxin herbicides (e.g. 2,4-D, triclopyr). Some have speculated that the historic 2,4-D treatments completed on Long Lake may have favored the competition of southern naiad over other native species that are prone to decline following these treatments.

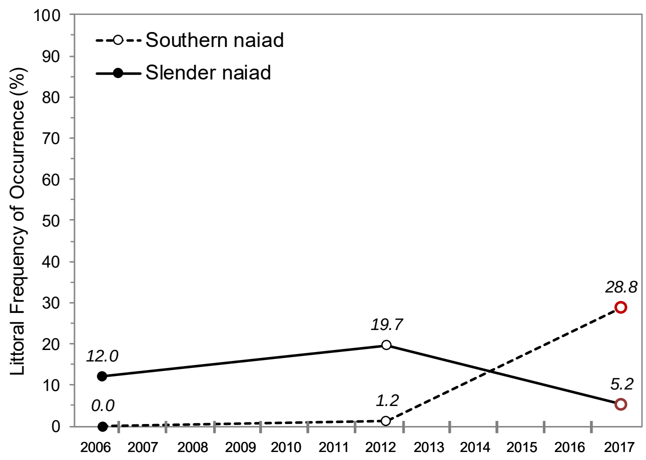


Figure 2.2-5. Littoral frequency of occurrence of slender naiad and southern naiad. Open circle represents statistically valid change from previous survey. Circle outlined with red indicates 2017 was statistically different from 2006 (Chi-Square α = 0.05).

Photo 2.2-1. Slender naiad (Najas flexilis; left) and southern naiad (N. guadalupensis; right). Photo credit Onterra.

Common waterweed was the second-most frequently encountered aquatic plant during the 2017 point-intercept survey in Long Lake with a LFOO of 26.2% (Figure 2.2-4). The occurrence of common waterweed has remained statistically indifferent between the 2006 (29.8%), 2012 (24.4%) and 2017 surveys (Figure 2.2-6). Common waterweed provides habitat and food sources to both aquatic and terrestrial wildlife. Lacking true roots and able to obtain the majority of its nutrients directly from the water, common waterweed often forms large mats which break free from the bottom and can continue to grow suspended in the water column or floating on the lake's surface. While not problematic in Long Lake, in

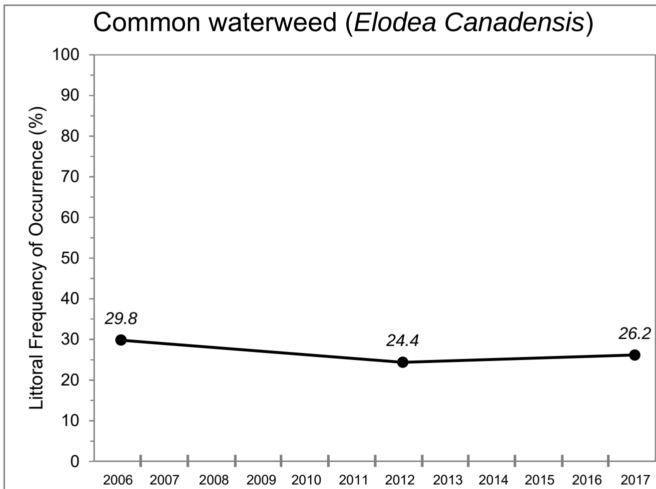


Figure 2.2-6. Littoral frequency of occurrence of common waterweed from 2006-2017. Open circle represents statistically valid change from previous survey. Circle outlined with red indicates 2017 was statistically different from 2006 (Chi-Square α = 0.05).

lakes with higher nutrient content, common waterweed can grow to excessive levels where it can interfere with recreational activity. The 2017 point-intercept survey found common waterweed to be present at the majority of the sampling locations in the southern end of the lake and in the small bay near the inlet from Big Sand Lake.

Muskgrasses and stoneworts are genera of macroalgae. Muskgrasses had a LFOO of 10.0% in 2006 which is very similar to the 10.1% LFOO observed in 2012 (Figure 2.2-7). By 2017, the occurrence of muskgrasses increased to 22.5%. Stoneworts increased between 2006 and 2012 and remained relatively constant between 2012 and 2017. These macroalgae require lakes with good water clarity, and their large beds stabilize bottom sediments. Studies have also shown that muskgrasses sequester phosphorus in the calcium carbonate incrustations which from on these plants, aiding in improving water quality by making the phosphorus unavailable to phytoplankton (Coops 2002).

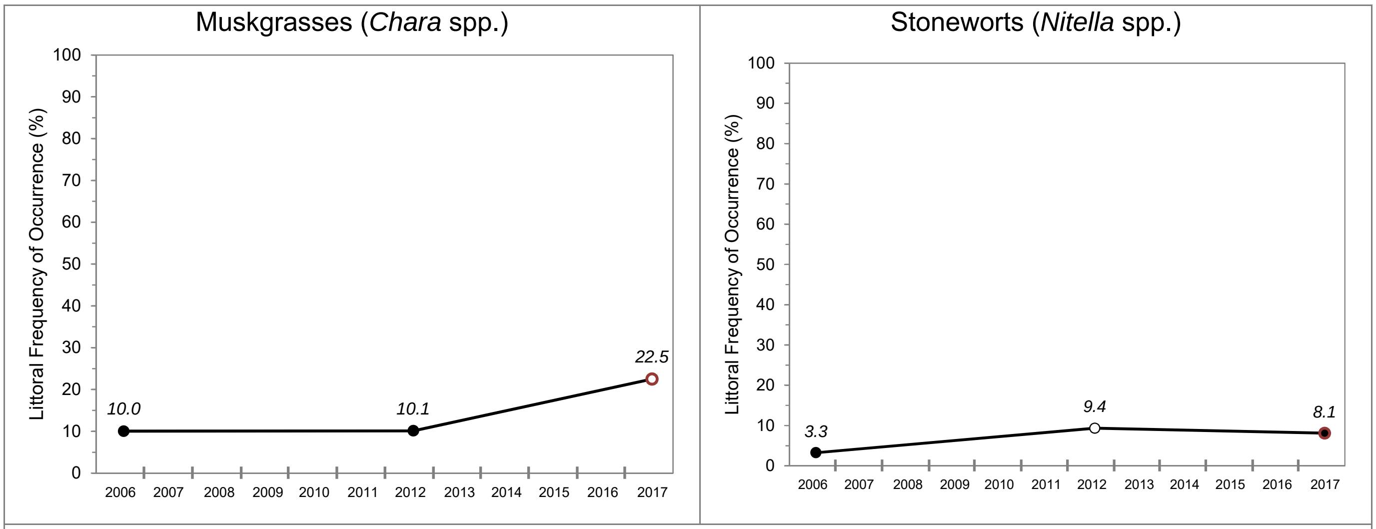


Figure 2.2-7. Littoral frequency of occurrence of native dicot aquatic plant species from 2006-2017. Open circle represents statistically valid change from previous survey. Circle outlined with red indicates 2017 was statistically different from 2006 (Chi-Square α = 0.05).

Eurasian watermilfoil is a dicot (broad-leaved plant) and the herbicides (2,4-D) which have been used historically on Long Lake in an effort to control EWM were historically believed to only have impacts to dicot species. Research conducted by the US Army Corps of Engineers, the WDNR, and private consultants have shown that these herbicides can be impactful to the broad-leaved plant community and certain non-dicot native plants are sensitive as well. Figure 2.2-8 shows how the dicot species in Long Lake have changed over time. Coontail populations have steadily increased over this time period. The population of northern watermilfoil decreased in 2012, potentially in response to the active herbicide control strategy occurring during that time frame. The northern watermilfoil rebounded in 2017 to slightly above 2006 levels (although not statistically different).

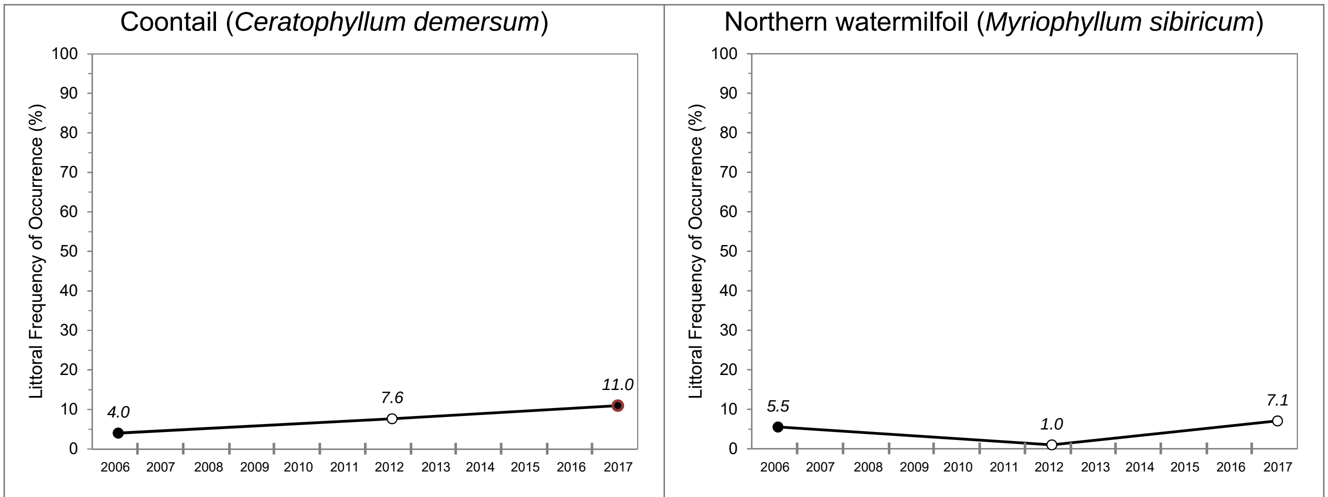


Figure 2.2-8. Littoral frequency of occurrence of native dicot aquatic plant species from 2006-2017. Open circle represents statistically valid change from previous survey. Circle outlined with red indicates 2017 was statistically different from 2006 (Chi-Square α = 0.05).

Ongoing research is indicating the thin-leaved pondweeds are also susceptible to impact from early-season 2,4-D treatments. This grouping of plants includes small pondweed and stiff pondweed, both with historically low population levels in Long Lake (Figure 2.2-9). Changes in these species has been noted and may be related to the herbicide control program that has occurred during this timeframe.

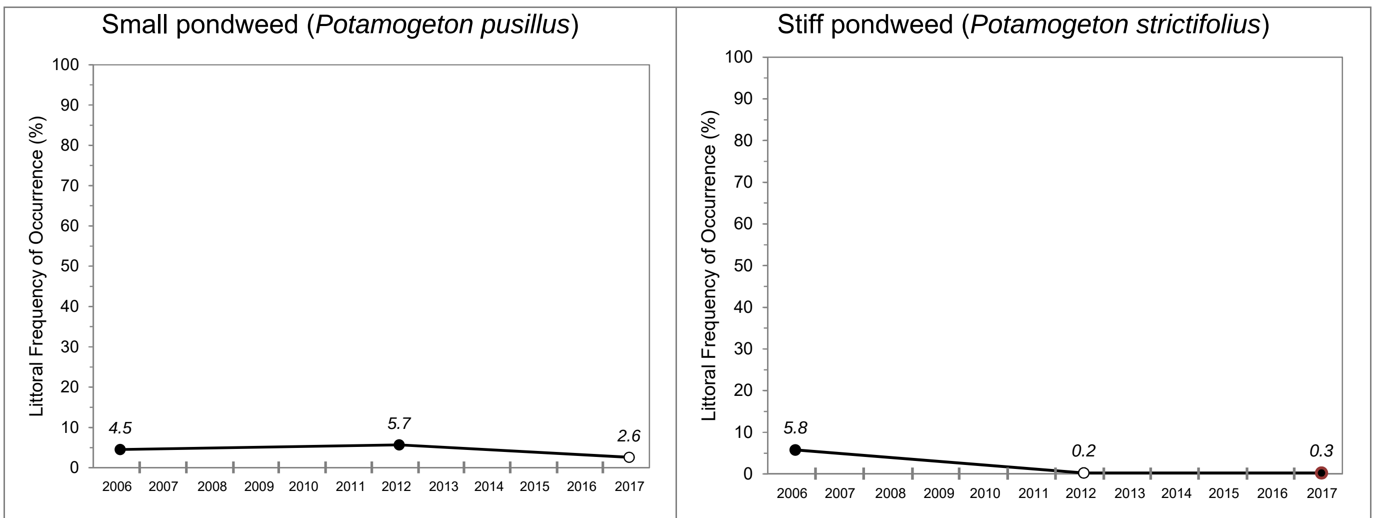


Figure 2.2-9. Littoral frequency of occurrence of thin-leaved pondweeds from 2006-2017. Open circle represents statistically valid change from previous survey. Circle outlined with red indicates 2017 was statistically different from 2006 (Chi-Square α = 0.05).

Figure 2.2-10 displays the LFOO of native narrow-leaf (monocot) plant species from the three point-intercept surveys conducted on Long Lake that are typically resilient to herbicide treatment strategies. Some species such as water celery and needle spikerush have displayed relatively similar populations indicating a stable population in the lake over time. Large-leaf pondweed and fern-leaf pondweed populations declined between 2012 and 2017. Clasping-leaf pondweed and variable pondweed both increased in population over this time period.

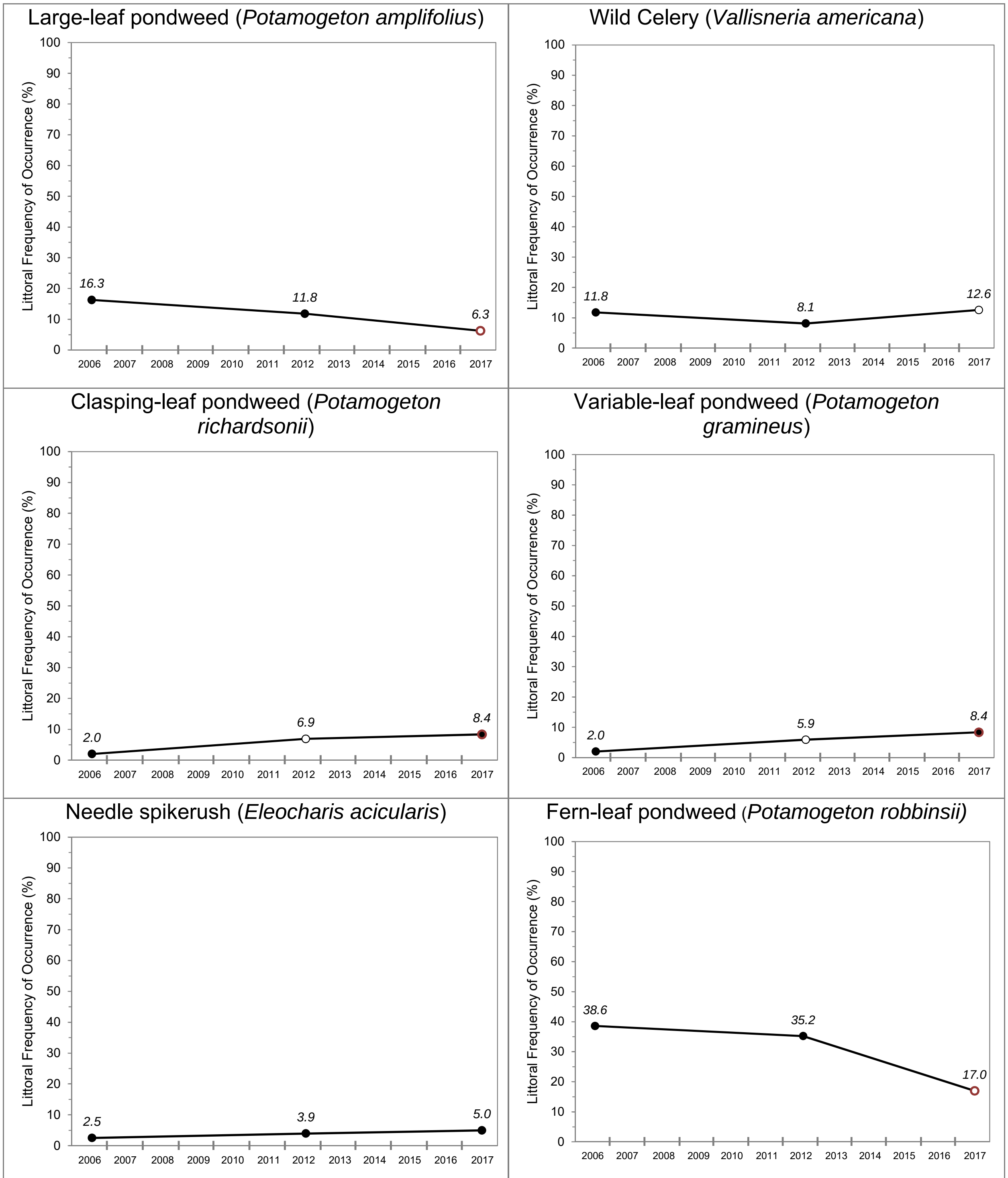


Figure 2.2-10. Littoral frequency of occurrence of native monocot aquatic plant species from 2006-2017. Open circle represents statistically valid change from previous survey. Circle outlined with red indicates 2017 was statistically different from 2006 (Chi-Square α = 0.05).

The calculations used for the Floristic Quality Index (FQI) for a lake's aquatic plant community are based on the aquatic plant species that were encountered on the rake during the point-intercept survey and does not include incidental species. The native aquatic plant species located on the rake during the point-intercept surveys from 2006 to 2017 and their conservatism values were used to calculate the FQI for each year. Native plant species richness has ranged from 32 in 2006 to 39 in 2017 with an average of 35 species (Figure 2.2-11). Native plant species richness in 2017 in Long Lake falls above the upper quartile values for other lakes within the NLFL ecoregion (28) and for lakes throughout Wisconsin (25). The native species richness values found in 2006 and 2012 were also above the upper quartile values for the ecoregion and state but were both less than the species richness found in 2017.

Average species conservatism ranged from 6.4 in 2006 to 6.9 in 2017 with an average of 6.6, falling above the median value for lakes in the NLFL region (6.7) and for lakes within the state (6.3) (Figure 2.2-11). Using Long Lake's annual species richness and average conservatism to calculate the annual FQI yielded values ranging from 36.3 in 2006 to 43 in 2017 with an average of 39.3 (Figure 2.2-5). The FQI values for Long Lake's aquatic plant community fall above the upper quartile value for lakes within the NLFL ecoregion (35.1) and for lakes throughout Wisconsin (32.6).

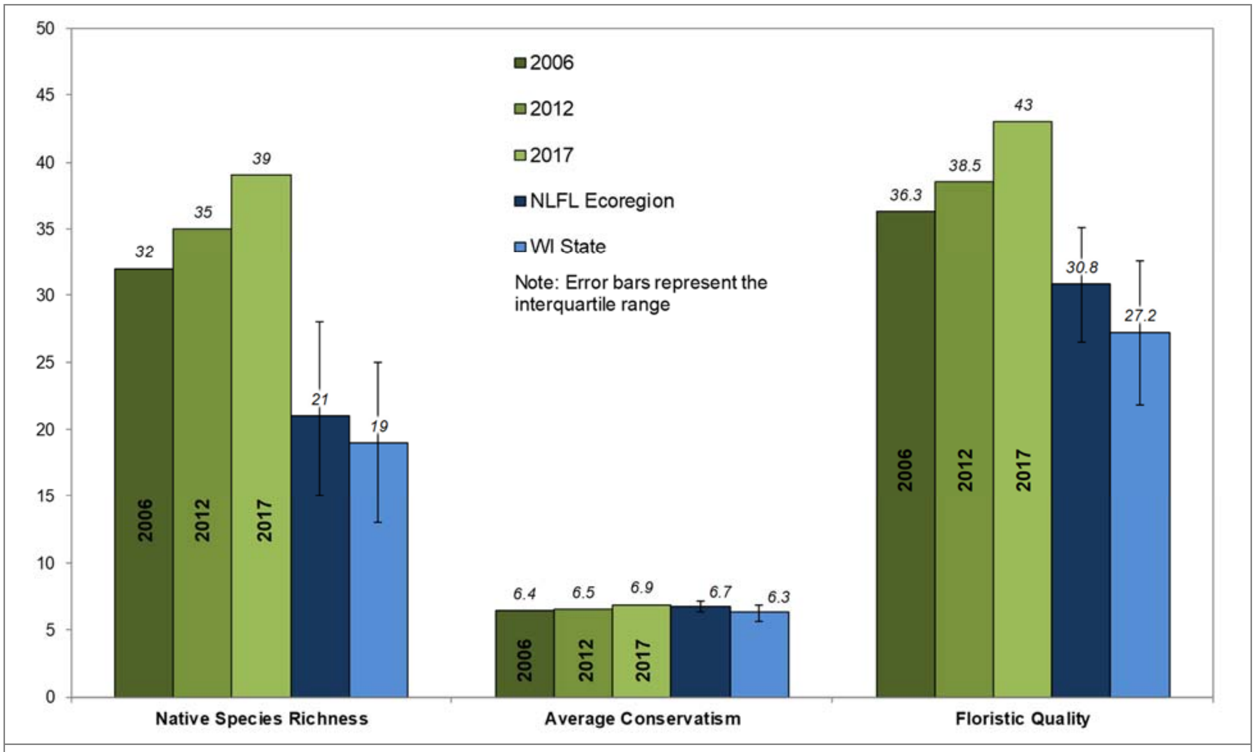


Figure 2.2-11. Long Lake Floristic Quality Assessment. Created using data from 2006, 2012, and 2017 surveys. Analysis following Nichols (1999) where NLFL = Northern Lakes and Forest Lakes Ecoregion.

When compared to other lakes in the NLFL ecoregion and the state, Long Lake has a higher number of native aquatic plant species and a higher number of conservative species, or species that are sensitive to environmental degradation. Overall, the FQI analysis indicates that the native plant community of Long Lake is of higher quality when compared to regional lakes and to lakes throughout the state.

While a method for characterizing diversity values of fair, poor, etc. does not exist, lakes within the same ecoregion may be compared to provide an idea of how Long Lake's diversity values rank. Using data collected by Onterra and WDNR Science Services, quartiles were calculated for 212 lakes within the NLFL Ecoregion (Figure 2.2-12). Using the data collected from the 2006-2017 whole-lake point-intercept surveys, Long Lake's aquatic plant species diversity ranged from 0.90 in 2006 to 0.93 in 2017. The 2006, 2012, and 2017 species diversity values fall at or above the upper quartile value (0.90) for lakes within the NLFL ecoregion, indicating high species diversity.

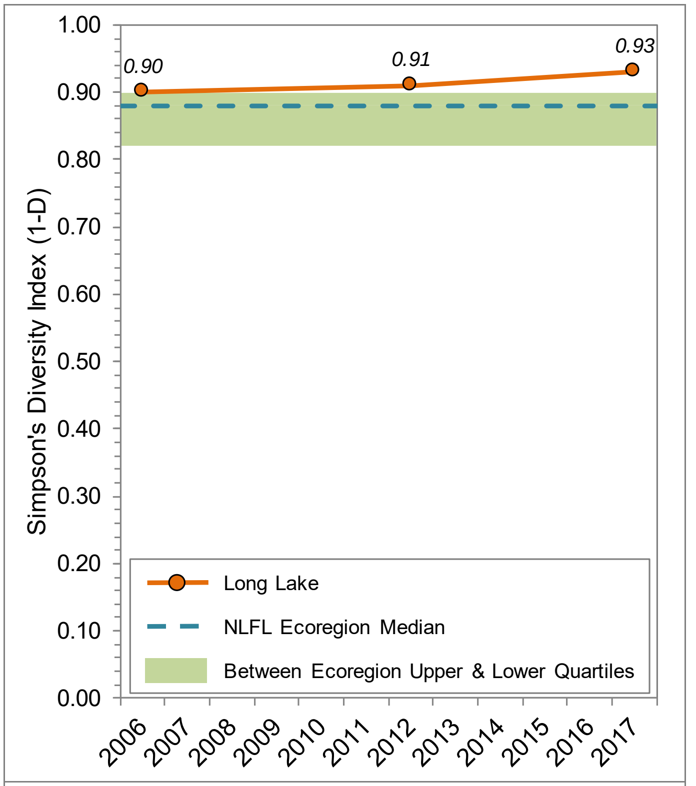


Figure 2.2-12. Long Lake 2006-2017 Simpson's Diversity Index. Created using data from 2006-2017 whole lake-lake point intercept surveys.

As explained earlier in the Primer on Data Analysis and Data Interpretation Section, the littoral frequency of occurrence analysis allows for an understanding of how often each of the plants is located during the point-intercept survey. Because each sampling location may contain numerous plant species, relative frequency of occurrence is one tool to evaluate how often each plant species is found in relation to all other species found (composition of population). For instance, while southern naiad was found at 29% of the sampling locations in Long Lake in 2017, its relative frequency of occurrence was approximately 14% (Figure 2.2-13). Explained another way, if 100 plants were randomly sampled from Long Lake, 14 of them would be southern naiad. In 2012, only approximately 8 out of 100 (7.7% relative frequency of occurrence) would have been southern naiad and 0 out of 100 would have been southern naiad in 2006. This analysis can demonstrate how the aquatic plant community has shifted over this time period.

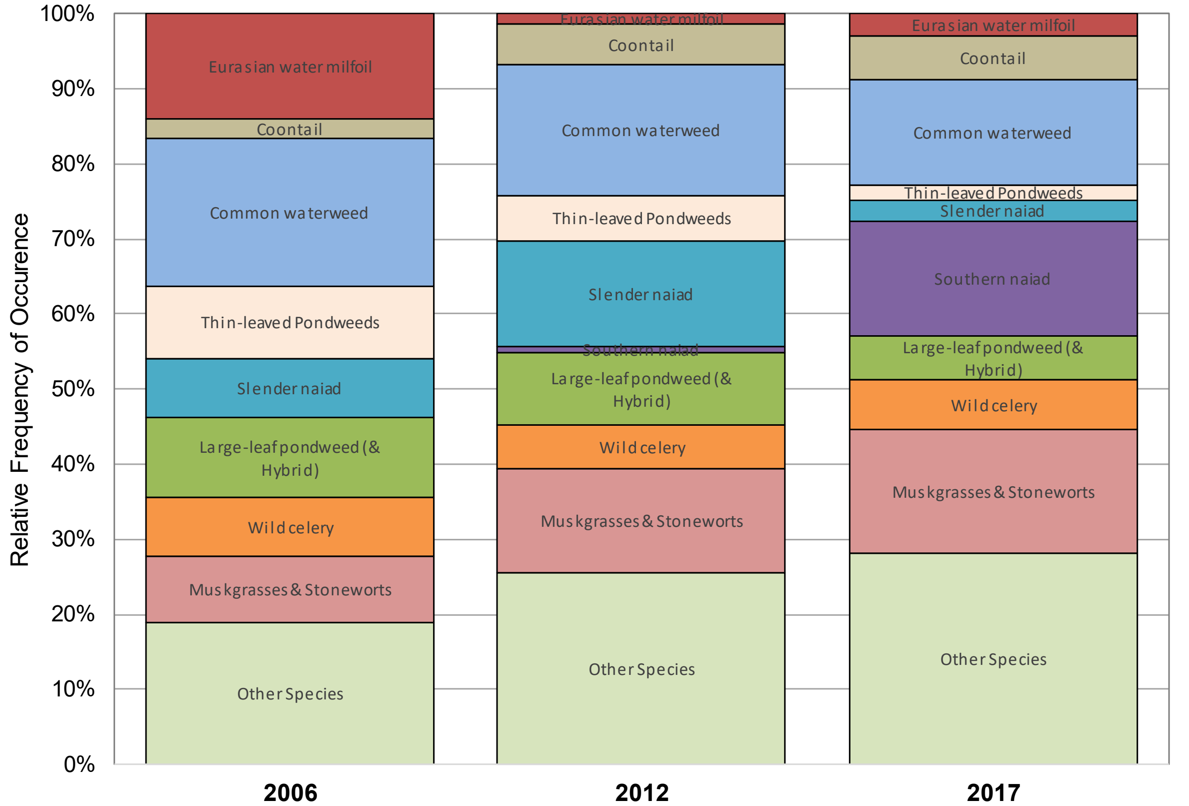


Figure 2.2-13. Relative frequency of occurrence analysis of Long Lake.

The quality of Long Lake's plant community is also indicated by the high incidence of emergent and floating-leaf plant communities that occur in near-shore areas around the lake. The 2017 community map indicates that approximately 27.5 acres (3.1%) of the 889 acre-lake contain these types of plant communities (Table 2.2-2 and Map 2 and Map 3). Twenty-two native floating-leaf and emergent species were located on Long Lake in 2017, providing valuable structural habitat for invertebrates, fish, and other wildlife. These communities also stabilize lake substrate and shoreland areas by dampening wave action from wind and watercraft.

Table 2.2-2. Long Lake acres of plant community types. Created from 2012 and 2017 community mapping survey.

	Acres	
Plant Community	**2012**	**2017**
Emergent	12.2	13.2
Floating-leaf	7.8	13.6
Mixed Emergent & Floating-leaf	2.5	0.6
Total	**22.5**	**27.5**

The community map represents a 'snapshot' of the important emergent and floating-leaf plant communities, a replication of this survey in the future will provide a valuable understanding of the dynamics of these communities within Long Lake. This is important because these communities are often negatively affected by recreational use and shoreland development. Radomski and Goeman (2001) found a 66% reduction in vegetation coverage on developed shorelands when compared to the undeveloped shorelands in Minnesota lakes. Furthermore, they also found a significant reduction in abundance and size of northern pike (*Esox lucius*), bluegill (*Lepomis macrochirus*), and pumpkinseed (*Lepomis gibbosus*) associated with these developed shorelands.

Overlaying the 2012 and 2017 community mapping surveys, there are no large-scale differences in the floating-leaf and emergent plant communities on Long Lake. Small increases in the coverage of emergent plant communities may be occurring in some nearshore areas. Also, a slight expansion of the floating-leaf communities can be observed along the western shoreline of the southern basing (Figure 2.2-14).

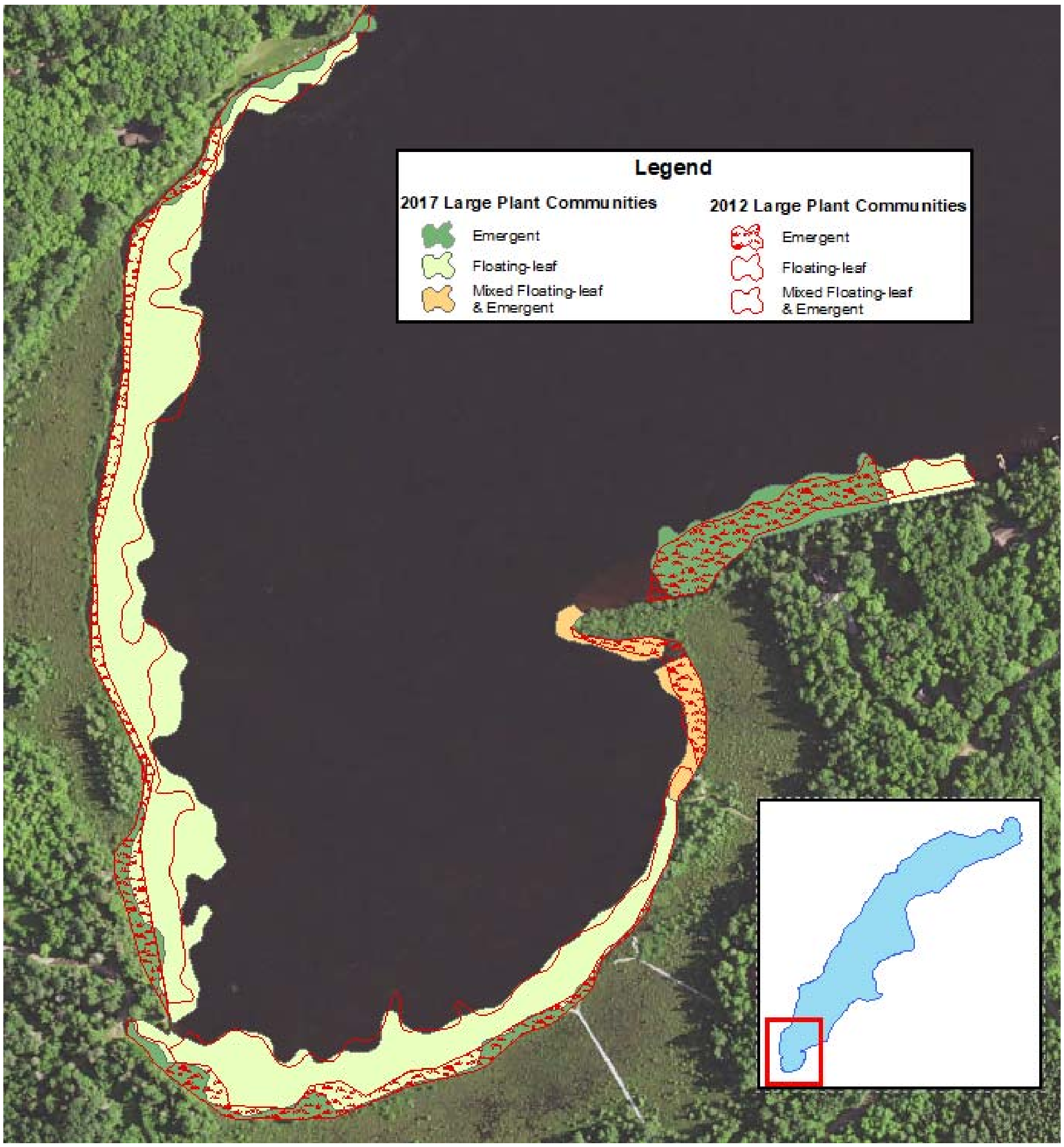


Figure 2.2-14. Comparison of 2012 and 2017 emergent and floating-leaf plant communities in southern basin of Long Lake.

2.3 Eurasian Watermilfoil

Eurasian watermilfoil is an invasive species, native to Europe, Asia and North Africa, that has spread to most Wisconsin counties (Figure 2.3-1). Eurasian watermilfoil is unique in that its primary mode of propagation is not by seed. It actually spreads by shoot fragmentation, which has supported its transport between lakes via boats and other equipment. In addition to its propagation method, EWM has two other competitive advantages over native aquatic plants, 1) it starts growing very early in the spring when water temperatures are too cold for most native plants to grow, and 2) once its stems reach the water surface, it sometimes does not stop growing like most native plants, instead it continues to grow along the surface creating a canopy that blocks light from reaching native plants. Eurasian watermilfoil can create dense stands and dominate submergent communities, reducing important natural habitat for fish and other wildlife, and impeding recreational activities such as swimming, fishing, and boating. However, in some lakes, EWM appears to integrate itself within the community without becoming a nuisance or having a measurable impact to the ecological function of the lake.

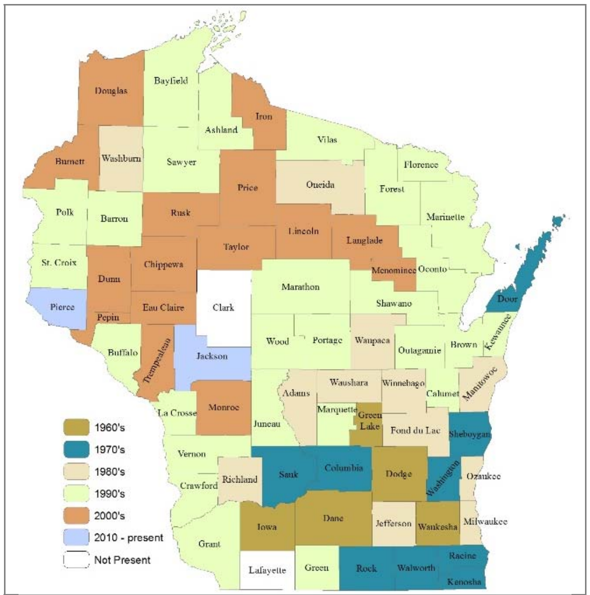


Figure 2.3-1. Spread of Eurasian watermilfoil within WI counties. WDNR Data 2015 mapped by Onterra.

WDNR Long-Term EWM Trends Monitoring Research Project

Starting in 2005, WDNR Science Services began conducting annual point-intercept aquatic plant surveys on a set of lakes to understand how EWM populations vary over time. This was in response to commonly held beliefs of the time that once EWM becomes established in a lake, its population would continue to increase over time. As outlined in *The Science Behind the "So-Called" Super Weed* (Nault 2016), EWM population dynamics on lakes are not that simplistic.

Like other aquatic plants, EWM populations are dynamic and annual changes in EWM frequency of occurrence have been documented in many lakes, including those that are not being actively managed for EWM control (no herbicide treatment or hand-harvesting program). The data are most clear for unmanaged lakes in the Northern Lakes and Forests Ecoregion (Figure 2.3-1). The upper frame of Figure 2.3-1 shows the EWM littoral frequency of occurrence for these unmanaged systems by year, and the lower frame shows the same data based on the number years the survey was conducted following the year of initial detection of EWM listed on the WDNR website. During this study, six of the originally selected "unmanaged lakes" were moved into the "managed" category as the EWM populations were targeted for control by the local lake organization as populations increased.

The results of the study clearly indicate that EWM populations in unmanaged lakes can fluctuate greatly between years. Following initial infestation, EWM expansion was rapid on some lakes, but overall was variable and unpredictable (Nault 2016). On some lakes, the EWM populations reached a relatively stable equilibrium whereas other lakes had more moderate year-to-year variation. Regional climatic factors also seem to be a driver in EWM populations, as many EWM populations declined in 2015 even though the lakes were at vastly different points in time following initial detection within the lake.

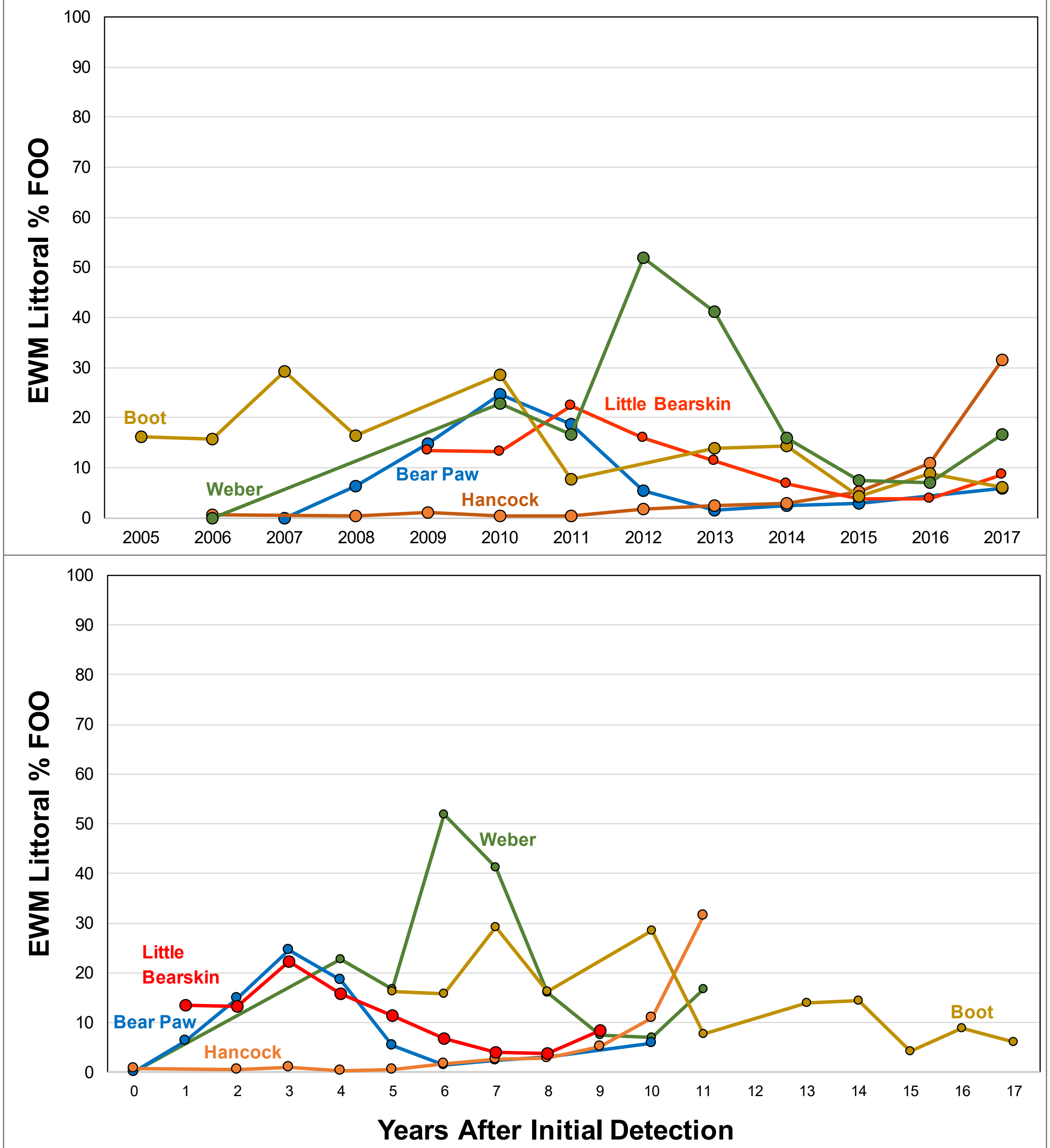


Figure 2.3-2. Littoral frequency of occurrence of EWM in the Northern Lakes and Forests Ecoregion without management. Data provided by and used with permission from the WDNR Bureau of Science Services.

Long Lake Historic EWM Management

It is important to note that two types of surveys are discussed in the subsequent materials: 1) point-intercept surveys and 2) AIS mapping surveys. As discussed above, the point-intercept survey provides a standardized way to gain quantitative information about a lake's aquatic plant population. The survey methodology allows comparisons to be made over time, as shown on Figure 2.3-3. It also allows comparison to be made between lakes, as shown in Figures 2.3-2.

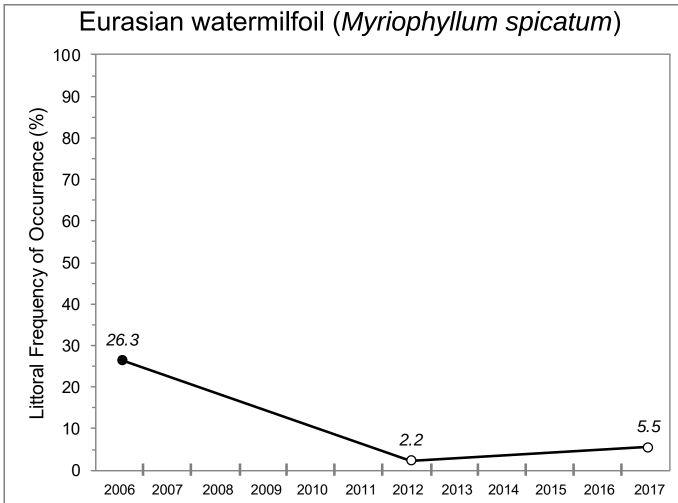


Figure 2.3-3. Littoral frequency of occurrence of EWM from 2006-2017. Open circle represents statistically valid change from previous survey.

Photo 2.3-1. EWM fragment with adventitious roots. Photo credit Onterra.

While the point-intercept survey is a valuable tool to understand the overall plant population of a lake, it does not offer a full account (census) of where a particular species exists in the lake. During the AIS mapping surveys, the entire littoral area of the lake was surveyed through visual observations from the boat (Photo 2.3-2). Field crews supplemented the visual survey by deploying a submersible camera along with periodically doing rake tows. The EWM population is mapped using sub-meter GPS technology by using either 1) point-based or 2) area-based methodologies. Large colonies >40 feet in diameter are mapped using polygons (areas) and were qualitatively attributed a density rating based upon a five-tiered scale from *highly scattered* to *surface matting*. Point-based techniques were applied to EWM locations that were considered as *small plant colonies* (<40 feet in diameter), *clumps of plants*, or *single or few plants*.

Photo 2.3-2. EWM mapping survey on Cloverleaf Lakes, Shawano County. Photo credit Onterra.

For reference, both the point-intercept survey and EWM mapping surveys occurred in 2017 and are shown on Map 5. EWM was located at 5.5% of the littoral point-intercept sampling locations, which are displayed on the left map. Within the

2017 point-intercept survey, only three sampling locations contained EWM in the northeastern two-thirds of the lake. However, the meander-based 2017 EWM mapping survey documented numerous *single or few* occurrences in this part of the lake. Overall, each survey has its strengths and weaknesses, which is why both are utilized in different ways as part of this project.

As discussed within the Introduction Section (1.0), EWM was first officially documented from Long Lake in 2000 although suspected to be present earlier. In 2006, the WDNR completed a point-intercept aquatic plant survey, locating EWM in approximately 26% of the littoral area of the lake. Onterra was contracted by the Long Lake of Phelps Lake District (LLPLD) during the late-summer of 2007, during which mapping surveys found the majority of the littoral zone contained colonized EWM of which approximately 25 acres was *surface matted*.

Following the finalization of a lake management plan by Northern Environmental, Inc. in 2007, the LLPLD successfully applied for WDNR grant funds in August of 2009 to initiate EWM control measures outlined within their management plan which used commonly considered best management practices (BMPs) of the time – spatially targeted spot herbicide treatments. The funds were to cover the first of a five-year program (2008-2012) aimed at significantly reducing the EWM within the lake through annual early-season herbicide spot treatments.

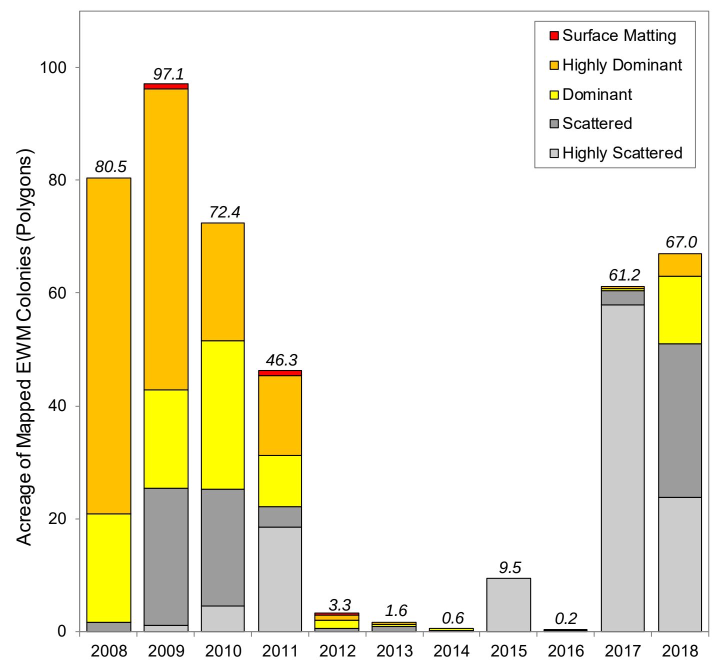


Figure 2.3-4. Acreage of mapped EWM colonies on Long Lake from 2009 to 2017. Data from Onterra Late-summer EWM peak-biomass surveys.

Starting in 2008, late-season EWM mapping surveys commenced using a consistent density rating system (Figure 2.3-4). Please note that this figure only represents only the acreage of mapped EWM polygons, not EWM mapped within point-based methodologies (*Single or Few Plants, Clumps of Plants,* or *Small Plant Colonies*). Said another way, EWM marked with point-based mapping methods do not contribute to colonized acreage as shown on Figure 2.3-4.

In 2008, just over 80 acres of colonized EWM was located in the lake with numerous additional locations of EWM marked with point-based data being located within the littoral zone. In 2008, almost the entirety of the colonized acreage was comprised of EWM with *dominant* or *highly dominant* density ratings.

Figure 2.3-5 shows the quantity of active ingredient of herbicide applied (primary vertical axis) and the application acreage (secondary vertical axis) of the LLPLD's EWM control program. It is important to note that application areas typically extend around a mapped EWM colony by a predefined buffer distance (e.g. 40-feet). The application areas may also encompass noncontiguous EWM colonies or EWM marked with point-based methods which result in an application area much greater than the EWM colonies they target. Specifications regarding the design of each years' application areas is contained within the respective annual *AIS Monitoring Report* and can be found here: https://dnr.wi.gov/lakes/grants/project.aspx?project=79127825.

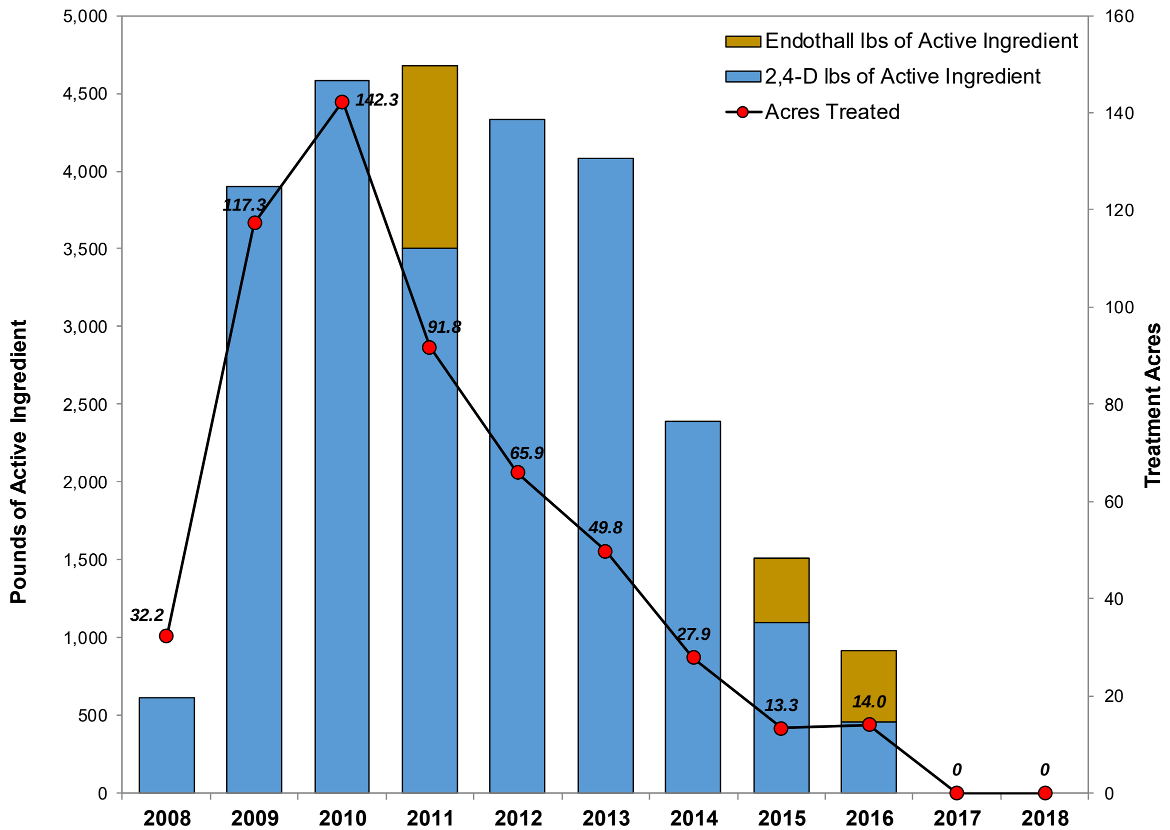


Figure 2.3-5. Herbicide Use History on Long Lake from 2008-2018.

The term *Best Management Practice (BMP)* is often used in environmental management fields to represent the management option that is currently supported by that latest science and policy. When used in an action plan, the term can be thought of as a placeholder with anticipation of having an evolving definition over time. In 2008, the BMP for managing EWM was through granular 2,4-D (ester) spot treatments. At the time of this writing, that strategy is no longer a BMP. Emerging science demonstrated that liquid treatments provided more consistent results at a fraction of the cost of granular products, larger application areas appeared to retain herbicide concentrations and exposure times better, and attention needed to be paid to the addition of individual spot treatments that may cumulatively function as a whole-lake treatment. Additional toxicological studies have also been published since 2008 which are import considerations within the risk assessments.

Onterra believes the largest advances in BMPs in regards to EWM management was gained as a part of a cooperative research project between the WDNR, US Army Corps of Engineers Research and Development Center (USACE), and private consultants. This program took place roughly from 2009 to 2016. The LLPLD was one of the first lake organizations in northern Wisconsin to become involved with this research project. Starting in 2010, consultation with the USACE occurred annually to develop an EWM control and monitoring strategy for Long Lake that utilized the emerging BMPs. Volunteers from the LLPLD participated in the collection of water samples surrounding the herbicide applications to understand the concentrations and exposure times of the treatments. Additional layers of data collection, such as rigorous sub-sample point-intercept surveys were conducted to evaluate the level of control and native plant collateral impacts from each treatment.

Long Lake contained the largest colonized EWM acreage in 2008 and 2009, which correspond to the start of the first grant-funded EWM control project (Figure 2.3-4). EWM management with herbicides prior to 2010 were largely ineffective resulting in less than seasonal EWM suppression. Higher concentrations of 2,4-D were subsequently applied to larger areas. The greatest EWM population reductions occurred in 2010-2013, where the size of the treatments and the proximity of the application areas to one another likely resulted in holding herbicide concentrations and exposure times (CETs) for an extended period of time compared to traditional spot-treatment scenarios. Herbicide concentration data collected in association with these treatments indicated that while the treatments did not technically function as whole-lake treatments, the treatments resulted in CETs slightly longer than traditional spot treatments. These factors likely lead to a more efficacious treatment. At the end of the first grant-funded control project in 2012, the colonized EWM population of Long Lake was 3.3 acres (Figure 2.3-4), with additional point-based EWM occurrences being noted around the lake.

The *Long Lakes Comprehensive Management Plan Update* (July 2013) contains, amongst others, a goal to: Control Existing and Prevent Further AIS Infestations within Long Lake. In order to build off the successes that have come slowly during the previous 5 years (2008-2012), the LLPLD created a plan that took a more aggressive approach to EWM management. During 2013 to present, the LLPLD's herbicide treatment threshold (trigger) included targeting all colonized areas of EWM (mapped with polygons) as well as immediately adjacent areas of EWM mapped with point-based techniques, with areas mapped as *small plant colonies* being targeted if possible. This strategy was approved at the July 2012 annual meeting (40 in favor, 0 against, 0 abstain), indicating support by district members of the management direction. The management plan was approved by the WDNR. The LLPLD was successfully awarded a Wisconsin Department of Natural Resources (WDNR) Aquatic Invasive Species (AIS) Established Population Control Grant in February 2013 to implement the EWM management program outlined within the *Long Lake Comprehensive Lake Management Plan Update* (July 2013) from 2013-2017.

During a Lake Management Planning project in 2012-2013, the LLPLD reviewed their EWM management strategies and revised their goals for future management. The LLPLD found that some of the herbicide treatments during this time period were not as effective as previous control strategies. As can be observed on Figure 2.3-5, the treatment acreage (line chart) declined to 50 acres in 2013, but the herbicide dosing strategy was increased and resulted in an overall similar amount being applied (bar chart) to years where over 100 acres were targeted.

The colonized EWM population of Long Lake was the lowest in recent record during 2013-2015. Following their WDNR-approved *Plan*, the LLPLD targeted largely point-based EWM occurrences in 2013-2014 with herbicide treatments. Data coming out of the cooperative WDNR and USACE research project indicated that in small spot treatments, the herbicide dissipates too rapidly to cause EWM mortality if systemic herbicides like 2,4-D are used. Even in some cases where larger treatment areas can be constructed, their narrow shape or exposed location within a lake may result in insufficient herbicide concentrations and exposure times for long-term control.

During 2015 and 2016, small areas of EWM that met the trigger definition were targeted with a combination of 2,4-D amine (4.0 ppm ae) and endothall dipotassium salt (1.5 ppm ai). This herbicide combination was suggested to be more effective under short exposure situations. These treatments appeared to have better short-term control than 2,4-D alone, yielding EWM reductions in targeted areas for 1-2 growing seasons before population rebound occurred.

Figure 2.3-6 illustrates the overlap of herbicide treatments being conducted from 2008 to 2016. Approximately 161 acres of the lake received direct application of herbicide, with the vast majority of this acreage was targeted multiple times. Just under 10 acres of the lake has been the target of treatment for seven years out of the nine years herbicide treatments were conducted.

This type of strategy can be analogous to the "whack-a-mole" arcade game; where areas are targeted, rebound, and then are targeted again. As outlined above, maintaining this strategy resulted in a reduced EWM population. However, the repeated need for exposing the lake to herbicides as is required when engaged in an annual spot treatment program has gone out of favor with some lake managers due to concerns over the non-target impacts that can accompany this type of strategy.

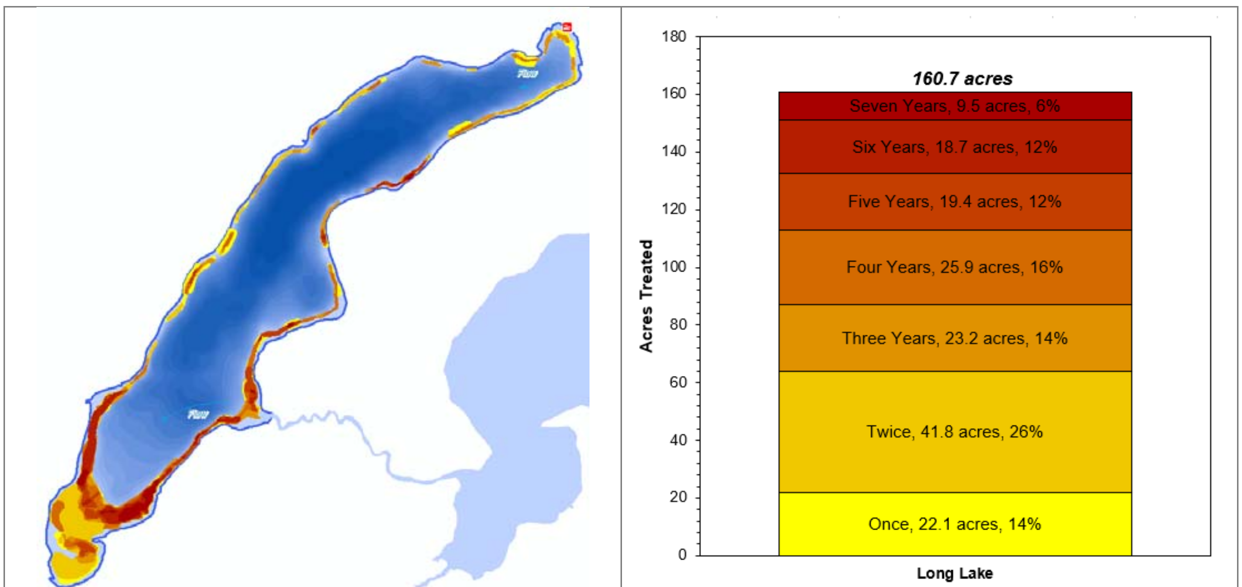


Figure 2.3-6. Footprint analysis of 2008-2016 herbicide treatments on Long Lake.

Starting in 2013, the LLPLD adopted an integrated approach to EWM management. The LLPLD sought to conduct EWM professional-based hand-harvesting methods in areas where spot herbicide treatments were not anticipated to be effective due to their small size or exposed nature. Professional hand-harvesting firms can be contracted for these efforts and can either use basic snorkeling or scuba divers, whereas others might employ the use of a Diver Assisted Suction Harvest (DASH) which involves divers removing plants and feeding them into a suctioned hose for delivery to the deck of the harvesting vessel. The DASH methodology is considered a form of mechanical harvesting and thus requires a WDNR approved permit. DASH is thought to be more efficient in removing target plants than divers alone and is believed to limit fragmentation during the harvesting process.

The 2013 trial program was conducted using traditional hand-harvesting techniques by a contractor (Aquatic Plant Management, LLC). These efforts targeted general areas of the lake with a low population level at that time. The 2013 efforts were difficult to access as professional firms of the time had not yet developed a robust reporting system of their efforts, which in more recent years are in place. The subsequent hand-harvesting program (2014-2016) was conducted using a DASH component (Figure 2.3-7). The DASH sites were smaller and more concise to coincide with WDNR permit requirements. The 2014-2016 hand-harvesting activities with DASH were conducted by Many Waters, LLC.

The EWM population of Long Lake increased in 2017 to levels that were beyond the capacity for hand-harvesting to be used for lake-wide population management. A few areas of high traffic were singled out for hand-harvesting with a goal to lessen the recreation and aesthetic impairment being caused in these areas by EWM. However, late-season filamentous algae conditions in these areas prevented the 2017 hand-harvesting efforts from occuring.

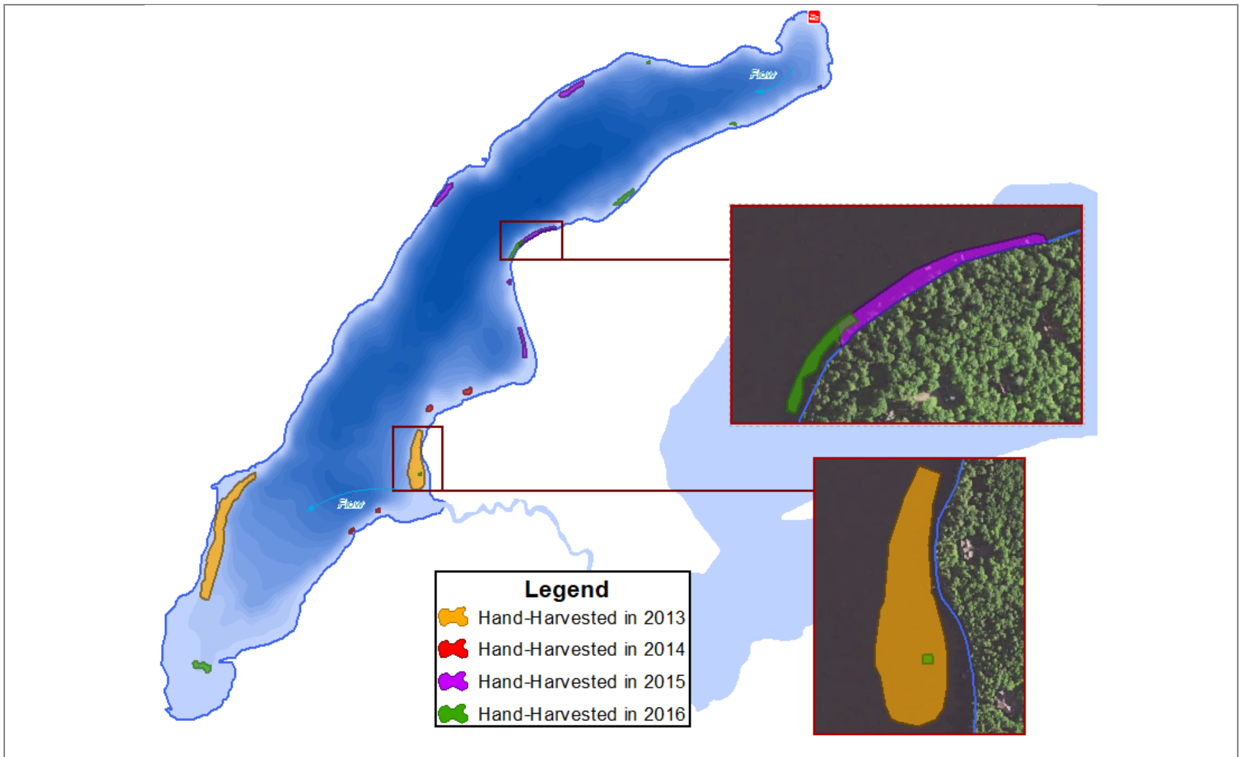


Figure 2.3-7. Location of hand-harvesting activities conducted from 2013-2016 on Long Lake.

Continued EWM population expansion occurred again in 2018 (Map 4). Similar to 2017, areas of perceived recreation impairment were targeted for hand-harvesting during 2018. The goal of these efforts was to target small areas in front of riparian frontage as opposed to the prioritized population management strategy used in 2013-2016. Many Waters, LLC conducted nine days of hand-harvesting with DASH during mid-August 2018. Their efforts removed over a thousand pounds of EWM.

On an August 12, 2016 visit to the lake, Many Waters, LLC encountered some suspected hybrid water-milfoil plants while conducting hand-harvesting in a northeastern area of Long Lake. Hybrid water milfoil (HWM) is a cross between the native northern water milfoil (*Myriophyllum sibiricum)* and the invasive Eurasian water milfoil (EWM, *Myriophyllum spicatum*). The suspected HWM was collected and sent to a lab in lower Michigan (GenPass, LLC) for genetic analysis where they confirmed the sample to be HWM.

In general, HWM typically has thicker stems, is a prolific flowerer, and grows much faster than pure-strain EWM (LaRue et al. 2012). These conditions may likely contribute to this plant being less susceptible to being controlled by standard use rates of certain herbicide control strategies (Glomski and Netherland 2010, Nault et al. 2018). Hybrid watermilfoils tend to interfere with recreation, navigation, and aesthetics more than pure-strain EWM populations.

While understood in terrestrial herbicide applications for years, tolerance evolution is an emerging topic amongst herbicide applicators, lake management planners, and researchers. Herbicide tolerance is when a plant population develops reduced susceptibility to an herbicide over time. This occurs in a population when some of the targeted plants have an innate tolerance to the herbicide and some do not. Following an herbicide treatment, the more tolerant strains will rebound whereas the more sensitive strains will be controlled. Thus, the plants that re-populate the lake will be those that are more tolerant to that herbicide resulting in a more tolerant population. Concern exists that the more-easily controlled EWM component of a lake's invasive milfoil population may be controlled by herbicide treatment, but the slightly less-susceptible HWM component will survive, rebound in a short period of time, and then become a larger proportion of the invasive milfoil population. Rotating herbicide use-patterns, especially away from continued use of auxin mimic herbicides like 2,4-D, can help avoid population-level herbicide tolerance evolution from occurring.

Long Lake Future Management Discussions

Following the distribution of a draft version of the *Long Lake 2017 EWM Control & Monitoring Report* on November 21, 2017, the LLPLD worked to form a Planning Committee to develop updated aquatic plant management goals based upon information learned over the previous 5-year control project. A few fragmented teleconferences occurred and the decision was made to finalize this report (March28-2018) and develop an interim nuisance EWM control strategy for 2018 using an increased level of hand-harvesting with DASH technology.

Following the 2018 field season on November 2, 2018, a draft version of the report sections (Section 2.0 and 3.0) of this document was submitted to the LLPLD including the three broad potential EWM population goals listed below (Figure 2.3-8). The LLPLD reviewed these potential EWM management goals, including the associated potential action plans for applicability on Long Lake. The following paragraphs provide brief overview of these extensive conversations. During these discussions, conversation regarding risk assessment of the various management actions were prominent. Onterra provided extracted relevant chapters from the WDNR's *APM Strategic Analysis Document* (Draft Dec2018) to serve as an objective baseline for the LLPLD to weigh the benefits of the management strategy with the collateral impacts each management action may have on the Long Lake Ecosystem. These chapters are included as Appendix B.

1. **No Coordinated Active Management**
   (Let Nature Take its Course)
   - Focus on education of manual removal by property owners
2. **Reduce EWM Population on a lake-wide level**
   (Lake-Wide Population Management)
   - Would likely rely on herbicide treatment strategies (risk assessment)
   - Will not "eradicate" EWM
   - Set triggers (thresholds) of implementation and tolerance
3. **Minimize navigation and recreation impediment**
   (Nuisance Control)
   - May be accomplished through professional hand-harvesting of areas or lanes
   - Hand-harvesting may not be able to accomplish this goal and herbicides or a mechanical harvester may be required

Figure 2.3-8. Potential EWM Management Goals.

Let Nature Take its Course: In some instances, the EWM population of a lake may plateau or reduce without conducting active management (Figure 2.3-2). Some lake groups decide to periodically monitor the EWM population, typically through an annual or semi-annual point-intercept survey, but do not coordinate active management (e.g. hand-harvesting or herbicide treatments). This requires that the riparians tolerate the conditions caused by the EWM, acknowledging that some years may be problematic to recreation, navigation, and aesthetics. In certain situation, could potential result in altered environment that would have responding shifts to the ecosystem. Individual riparians may choose to hand-remove the EWM within their recreational footprint, but the lake group would not assist financially or assist with securing permits. In some instances, the lake group may select this management goal, but also set an EWM population threshold or "trigger" where they would revisit their management strategy if the population reached that level.

Lake-Wide Population Management: Some believe that there is an intrinsic responsibility to correct for changes in the environment that are caused by humans. For lakes with EWM populations, that may be to manage the EWM population at a reduced level with the perceived goal to allow the lake to function as it had prior to EWM establishment. It must also be acknowledged that some lake managers and natural resource regulators question whether that is an achievable goal.

In early EWM populations, the entire population may be targeted through hand-harvesting or spot treatments. This is the strategy the LLPLD employed from 2008-2016, where a density-based trigger dictated when and where herbicide treatments and hand-harvesting would occur.

On more advanced or established populations, this may be accomplished through large-scale control efforts such as water-level drawdowns or whole-lake herbicide treatment strategies. If conducted properly, large-scale management can reduce EWM populations for several years, but will not eradicate it from the lake. Subsequent smaller scale management (e.g. hand-harvesting or spot treatments) is typically employed to slow the rebound of the population until another large-scale effort is likely required again. Typically, complete rebound of an EWM population following a large-scale control action is 4-6 years, with quicker rebound on some lakes and longer control observed on others. Large-scale control efforts, especially using herbicide treatments, can be impactful of some native plant species as well as carry a risk of environmental toxicity. Some argue that the impacts of the control actions may have greater negative impacts to the ecology of the system than if the EWM population was not managed.

For reference, the Big Sand Lake Property Owners Association (BSLPOA) has adopted a Lake-Wide EWM Population Management approach. The EWM conditions are tolerated and no coordinated management is conducted until the EWM population reaches a particular threshold (trigger). Some groups use EWM mapping survey data to define thresholds, where others prefer to use data from the point-intercept surveys. The BSLPOA coupled the point-intercept data with the Late-Summer EWM Mapping Survey data and surmised that when historic EWM populations exceeded 10%, *highly dominant* and *surface matted* conditions started becoming apparent. Therefore, a threshold of 15% littoral frequency of EWM measured by the point-intercept survey was adopted by the BSLPOA. If/when the EWM population reaches this threshold, the BSLPOA has chosen to review the most currently accepted management strategies, likely consisting of a whole-lake herbicide treatment, to determine applicability. Selection of management thresholds vary based on a number of factors, most notably how the EWM manifests in each system.

Nuisance Control: The concept of ecosystem services is that the natural world provides a multitude of services to humans, such as the production of food and water (provisioning), control of climate and disease (regulating), nutrient cycles and pollination (supporting), and spiritual and recreational benefits (cultural). Some lake groups acknowledge that the most pressing issues with the EWM population on their lake is the reduced recreation, navigation, and aesthetics compared to before EWM became established in their lake. Particularly on lakes with large EWM populations that may be impractical or unpopular to target on a lake-wide basis, the lake group would coordinate (secure permits and financially support the effort) a strategy to improve the navigability within the lake. In order to reach this goal, a strategic network of common use lanes and riparian spokes through EWM colonies are maintained by either professional hand-harvesting or mechanical harvesting (i.e. weed cutting machine). On lakes with surface matted or near surface matted EWM in high navigation corridors, mechanical harvesting may be able to temporarily remove the top few feet of EWM of select areas whereas herbicide spot treatments may provide an entire season of nuisance relief.

A Nuisance Control Strategy was employed on Long Lake in 2018 through the use of hand-harvesting with DASH. If the LLPLD continues to enact a Nuisance Control Strategy, they will need to determine if hand-harvesting is sufficient to meet the needs of its constituents or if another method, potentially contracting a mechanical harvesting firm, should be explored. A risk assessment of using a mechanical harvester also needs to be conducted, particularly in regards to non-target bi-catch (e.g. fish, insects), exacerbated spread of EWM, and increased plant fragments washing up on shoreland properties.

3.0 LAKE WATER QUALITY TRENDS

Many types of analyses are available for assessing the condition of a particular lake's water quality. Within the *Long Lake Comprehensive Lake Management Plan Update* (July 2013), analysis of Long Lake's water quality was focused upon attributes that are directly related to the productivity of the lake. In other words, the water quality that impacts and controls the fishery, plant production, and even the aesthetics of the lake are related. Three water quality parameters were focused upon:

> **Phosphorus** is the nutrient that controls the growth of plants in the vast majority of Wisconsin lakes. It is important to remember that in lakes, the term "plants" includes both algae and macrophytes. Monitoring and evaluating concentrations of phosphorus within the lake helps to create a better understanding of the current and potential growth rates of the plants within the lake.
>
> **Chlorophyll-*a*** is the green pigment in plants used during photosynthesis. Chlorophyll-*a* concentrations are directly related to the abundance of free-floating algae in the lake. Chlorophyll-*a* values increase during algal blooms.
>
> **Secchi disc transparency** is a measurement of water clarity. Of all limnological parameters, it is the most used and the easiest for non-professionals to understand. Furthermore, measuring Secchi disc transparency over long periods of time is one of the best methods of monitoring the health of a lake. The measurement is conducted by lowering a weighted, 20-cm diameter disk with alternating black and white quadrates (a Secchi disc) into the water and recording the depth just before it disappears from sight.

The parameters described above are interrelated. Phosphorus controls algal abundance, which is measured by chlorophyll-*a* levels. Water clarity, as measured by Secchi disc transparency, is directly affected by the particulates that are suspended in the water. In the majority of natural Wisconsin lakes, the primary particulate matter is algae; therefore, algal abundance directly affects water clarity.

Water quality data is currently been collected by the Wisconsin Valley Improvement Corporation (WVIC) for a 3-year period, once every 10 years. The next sampling period will be conducted in 2020-2023. In addition to the WVIC's efforts, volunteer water quality monitoring has been completed annually by Long Lake riparians through the Citizen Lake Monitoring Network (CLMN). The CLMN is a WDNR program in which volunteers are trained to collect water quality information on their lake. Data have been collected through the advanced CLMN program in the past on Long Lake, consisting of Secchi disc readings and water chemistry collections three times during the spring turnover and three summer months.

The *Long Lake Comprehensive Lake Management Plan Update* (July 2013) provides context in regards to what the levels of these parameters means for the health of Long Lake. The focus of water quality investigation in this report is to determine if changes in the water quality of Long Lake have occurred. Linear regression analysis is a relatively basic way for lake ecologists a way to discover if statistically valid trends (increases or decreases) in water quality parameters are occurring. Linear regression analysis generates an equation or line of best fit (regression line) that minimizes the distance between the data points. A statistical measure of how close the measured data are to the regression line is called the r-squared statistic (r^2) and ranges from 0 to 1 (0% to 100%). An r^2 value of 0 indicates that the model does not explain any of the variability in the data

(0% of the data), while an r^2 value of 1 indicates that the model explains all of the variability in the data (100% of the data).

In addition to r^2, linear regression analysis also generates a *p*-value, which indicates if time is a significant predictor of change in a water quality parameter (i.e. is a trend occurring). A low *p*-value (≤ 0.05) indicates that a statistically valid change in a water quality parameter has occurred over time, while a larger *p*-value (> 0.05) indicates that a statistically valid change has not occurred.

Linear regression analysis was performed for near-surface average summer total phosphorus (Figure 3.0-1, left frame) and chlorophyll-*a* concentrations (Figure 3.0-1, right frame) from 1993 to 2017. Average summer total phosphorus concentrations exhibit a weak but statistically valid increasing trend and yielded a moderate r^2 value (0.274) and a low *p*-value (0.031). However, average summer chlorophyll-*a* concentrations did not exhibit a statistically valid trend and yielded a low r^2 value (0.194) and a high *p*-value (0.076). The increasing phosphorus is not causing a statistically valid increase in chlorophyll-*a* concentrations.

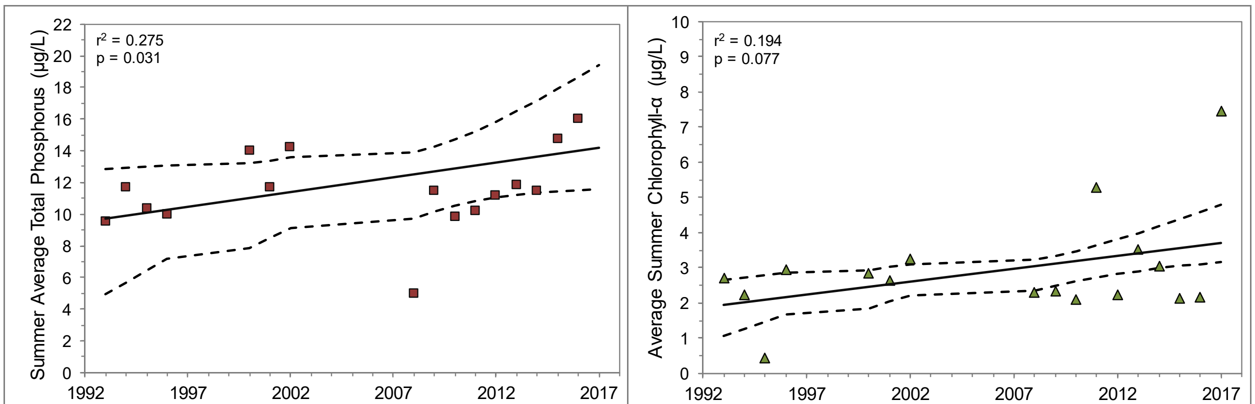


Figure 3.0-1. Long Lake linear regressions for average summer total phosphorus(left) and chlorophyll-α(right) from 1979-2017. Solid line indicates regression line, dashed lines indicated upper and lower confidence limits (95%).

The increasing phosphorus trend is also not causing a trend in decreased water clarity. Linear regression analysis was completed on the average summer Secchi disc depth data from 1988-2017, a period of which these data are available from almost every year (Figure 3.0-2). This linear regression did not exhibit a statistically valid trend ($p = 0.895$). This indicates that water clarity has been variable but not trending towards increased or decreased values. Please note that an investigation of water clarity from 1993 to 2017 was also conducted, as it corresponds to the

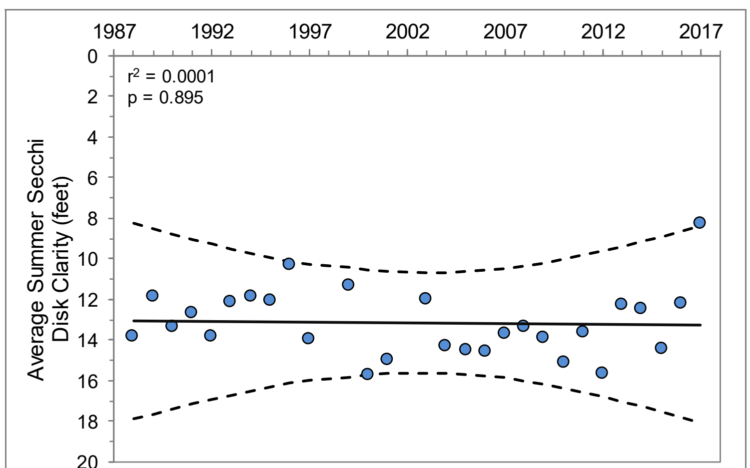


Figure 3.0-2. Long Lake linear regression for Secchi disc depth from 1988-2017. Solid line indicates regression line, dashed lines indicated upper and lower confidence limits (95%).

period in which total phosphorus has had a weak statistically increasing trend (Figure 3.0-1, left). This investigation also did not yield a statistically valid trend ($p = 0.733$).
A measure of water clarity once all of the suspended material (i.e. algae and sediments) have been removed, it is termed *true color*, and indicates the level of dissolved material within the water. True color can be measured in standard units (SU) or in Platinum-cobalt units (Pt-co units, or PCU). Lillie and Mason (1983) categorized lakes with 0-40 PCU as having "low" color, 40-100 PCU as "medium" color, and >100 PCU as "high" color. True color in Long Lake was 21 PCU in July 1979, indicating that the lake's water clarity is not influenced by dissolved components in the water (i.e. stained water).

It is believed that the increase in total phosphorus concentrations since 1993 is likely due to increased precipitation (Figure 3.0-3, $p = 0.017$). Precipitation data measured in Phelps, Wisconsin approximately two miles west of Long Lake indicate that annual precipitation has increased by approximately 5.2 inches from 1993-2017. Increased precipitation would likely result in increased nutrient loading to the lake from Long Lake's watershed.

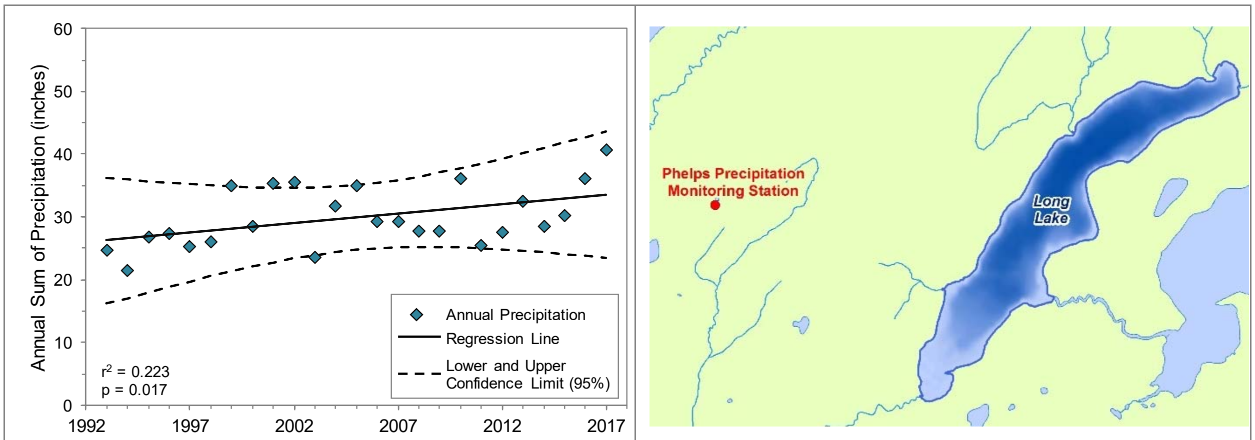


Figure 3.0-3. Annual precipitation from 1993-2017 (left) measured in Phelps. Monitoring station is approximately 2 miles from Long Lake. Data retrieved from the Midwestern Regional Climate Center (MRCC). Station GHCN ID: USC00476518.

4.0 AQUATIC PLANT IMPLEMENTATION PLAN SECTION

The *Long Lake Comprehensive Management Plan* was finalized and approved by the WDNR in July 2013. The Implementation Plan Section of the *Long Lake Comprehensive Management Plan* (July2013) includes the following management goals along with specific management actions developed to help reach those goals. The *Long Lake Comprehensive Management Plan* (July2013) can be found on the WDNR website located here:

https://dnrx.wisconsin.gov/swims/downloadDocument.do?id=84922433

1. Increase LLPLD's Capacity to Communicate with Lake Stakeholders and Facilitate Partnerships with Other Management Entities
    - Use education to promote lake protection and enjoyment through stakeholder education
    - Continue LLPLD's involvement with other entities that have responsibilities in managing (management units) Long Lake
2. Maintain Current Water Quality Conditions
    - Monitor water quality through WDNR Citizens Lake Monitoring Network
3. Control Existing and Prevent Further Aquatic Invasive Species Infestations within Long Lake
    - Continue implementation of an herbicide application strategy to control Eurasian water milfoil infestation on Long Lake
    - Continue Clean Boats Clean Waters watercraft inspections at Long Lake public access location
    - Enhance volunteer Eurasian water milfoil surveillance monitoring and hand removal program
4. Improve Fishery Resource and Fishing
    - Continue to work with fisheries managers to enhance the overall fishery on Long Lake

Figure 4.0-1. Long Lake management goals (numbered) and actions developed to assist in reaching the goal. From *Long Lake Comprehensive Management Plan* (July2013)

The LLPLD was awarded a WDNR AIS Established Population Control Grant in February 2013 (ACEI-132-13) to fund EWM management and monitoring from 2013-2017. Remaining funds from the grant allowed the project to extend to 2018. As a part of that project, the LLPLD would revisit their aquatic plant management-related Implementation Plan to update its content based on the lessons learned during the project. This section provides an update to those management goals and actions.

Information received by Onterra from teleconferences with members of the LLPLD board (Nov13-2018 and Dec20-2018) allowed the creation of a draft Implementation Plan Section Outline for the 2-member LLPLD Planning Committee (Feb11-2019) to review. Feedback from a subsequent teleconference with the LLPLD Planning Committee (Feb18-2019) was integrated into a revised version of the outline which was subsequently provided to the WDNR (Carol Warden). A teleconference between WDNR, LLPLD Planning Committee, and Onterra occurred on Feb19-2019 to gain feedback from the WDNR on the preliminary strategy before presenting it to a larger audience of LLPLD board members. These discussions along with follow-up communications with the WDNR resulted in the framework of the Implementation Plan Section included here.

The LLPLD Board of Directors received a complete draft copy of this document on March 13, 2019 and held a subsequent internal conference call. Discussions during this call were primarily focused on the EWM active management actions, including risk assessments, scale-appropriate implementation, and regulatory limitations. The LLPLD Board of Directors approved moving forward with the document as written for subsequent review by external partners.

In late-April 2019, an official first draft of the LLPLD's *Long Lake Aquatic Plant Management Plan Update* was provided to WDNR, Wisconsin Valley Improvement Company, Great Lakes Indian Fish and Wildlife Commission, Vilas County, and Town of Phelps Lakes Committee for external review. Written review of the draft plan was received on May 21, 2019 from WDNR team leader Carol Warden (UW Trout Lake AIS Specialist). The WDNR comments and how they are addressed in the final plan are contained in Appendix C. This appendix also contains a response to the WDNR comments from the LLPLD Planning Committee authored by Dan Anderson.

The Implementation Plan is a living document in that it will be under constant review and adjustment depending on the condition of the lake, the availability of funds, level of volunteer involvement, and the needs of the stakeholders.

Management Goal 1: Maintain EWM Populations Below Nuisance Levels

Management Action:	Continue Clean Boats Clean Waters watercraft inspections at critical public access locations
Timeframe:	Continuation of current effort
Facilitator:	Board of Directors or Invasive Species Committee
Description:	Currently the LLPLD monitors the public boat landings using training provided by the Clean Boats Clean Waters program. Long Lake is a popular destination by recreationists and anglers, making the lake vulnerable to new infestations of exotic species. The intent of the boat inspections would not only be to prevent additional invasive species from entering the lake through its public access point, but also to prevent the spread of invasive species that originated in Long Lake to other waterbodies. The goal would be to cover landing during the busiest times in order to maximize contact with lake users, spreading the word about the negative impacts of AIS on lakes and educating people about how they are the primary vector of its spread. Inspections at the Long Lake landing have exceeded 200 hours annually since 2012 through a paid effort and the LLPLD intends to continue the inspection efforts at this level. This program has been historically sustained through a streamline CBCW grant program with partnership from Vilas County. As that program has been dissolved, the LLPLD will take a lead role in ensuring that watercraft inspections are in place through a combination of paid and volunteer staffing.
Action Steps:	
	See description above as this is an established program.

Management Action:	Conduct three-tiered EWM population management on Long Lake
Timeframe:	Continuation of current effort
Facilitator:	Board of Directors or Invasive Species Committee
Description:	The goal of this action will be to minimize the periodic nuisance conditions that EWM causes on Long Lake. The following management options are not listed in order of preference, but are in order of decreasing scale. The WDNR has indicated their preference for hand-harvesting. 1. **Herbicide Spot Treatment** When a Late Season AIS Survey documents colonized EWM populations that are *dominant or greater in density*, an herbicide spot treatment would be considered for the following early-spring. Herbicide spot treatment techniques would be implemented if the colonies have a size/shape/location where management is anticipated to be effective. In general, this would be areas confined to bays (not exposed), broad in shape (not narrow bands), and over roughly 5 acres in size. On Long Lake, this will be difficult as most areas contain a narrow littoral footprint of EWM. Future spot herbicide treatments may need to consider herbicides (diquat, florpyrauxifen-benzyl, etc) or herbicide combinations (2,4-D/endothall, diquat/endothall, etc) thought to be more effective under short exposure situations than with traditional weak-acid auxin herbicides (e.g. 2,4-D, triclopyr). Advancements in research into new herbicides and use patterns will need to be integrated into future management strategies, including effectiveness, native plant selectivity, and environmental risk profile. ▪ Early consultation with WDNR would occur. ▪ The proceeding annual AIS monitoring report would outline the control and monitoring strategy. • Monitoring EWM efficacy by comparing annual late-summer EWM mapping surveys. • If grant funds are being used or new-to-the-region herbicide strategies are being considered, the WDNR may request a quantitative evaluation monitoring plan be constructed that is consistent with Appendix D of the WDNR Guidance Document, Aquatic Plant Management in Wisconsin (WDNR 2010). This generally consist of collecting quantitative point-intercept sub-sampling on sites approximately 10-acres or greater during the summer before the treatment (pre) and summer following the treatment

(post). Herbicide concentration monitoring may also occur surrounding the treatment in these instances.

- An herbicide applicator firm would be selected in late-winter and a conditional permit application would be applied to the WDNR.
- A focused pretreatment survey would take place approximately a week or so prior to treatment (approx. 2-3 weeks after ice-out). This site visit would evaluate the growth stage of the EWM (and native plants) as well as to confirm the proposed treatment area extents and water depths. This information would be used to finalize the permit, potentially with adjustments and dictate approximate ideal treatment timing.
- The herbicide treatment would occur when mid-depth water temperatures are roughly below 60°F and active growth tissue is confirmed on the target plants. Treatments would occur when wind conditions are low.

2. **<u>Mechanical Harvesting</u>** When the Late Season AIS Survey documents colonized EWM populations that are *highly dominant* or *surface matting*, but the areas are not conducive and/or popular to target with an herbicide spot treatment, contracting with a mechanical harvesting firm (i.e. weed cutter) would be considered for the following summer. The mechanical harvester would remove the dense EWM biomass that is near the surface. It is likely that a predetermined minimum acreage of mechanical harvesting would be required in a given year to be commensurate with the costs of mobilization. Many mechanical harvesting contract firms have a minimum project size (e.g. 4 day's worth of harvesting) that needs to be considered. At this time, the EWM population in Long Lake is below thresholds that would justify the LLPLD purchasing their own equipment, but the concept could be revisited at a later date.

    - Early consultation with WDNR would occur.
    - The annual AIS monitoring report would outline the control strategy.
    - A mechanical harvester firm would be selected in late-winter and a conditional permit application would be applied to the WDNR.
    - A focused pre-harvesting survey (likely in mid-June) may be requested by WDNR to finalize the permit, potentially with adjustments, and dictate approximate ideal implementation timing.
    - Mechanical harvesting operations would have the following guidelines:
        - The harvester would not be permitted in waters less than 3-feet to minimize sediment disturbance.

- Cut to half the water depth or 4', whichever is shallower
- An attempt would be made to return all gamefish and panfish to the water immediately.
- Harvesting should occur in late-June to maximize reduction for July-August. A second cutting may be required.
- The WDNR has indicated that they would not allow for the harvest of areas below the thresholds outlined above in order to give a harvester sufficient work to satisfy a minimum contract size.

3. **<u>Hand-Harvesting (includes DASH)</u>** While mechanical harvesting and herbicide spot treatment may be considered by the LLPLD, it is likely that those strategies would only be employed in select situations. The LLPLD would largely need to rely on annual hand-harvesting to alleviate nuisance conditions in select parts of the lake. If large and contiguous EWM colonies exist, removing EWM in navigation lanes through hand-harvesting, likely with Diver-Assisted Suction Harvest (DASH), would be appropriate. Typically 10-ft wide lanes are created extending from a riparian's pier towards deeper water.

   - The LLPLD may choose to defer the costs of conducting the hand-harvesting to the benefitting riparians even though the LLPLD would be the entity applying for and funding the permit.
     - In high-use areas that benefit more than adjacent riparians, the LLPLD would give considerations to incurring the hand-harvesting costs.
     - A hand-harvesting firm would be selected and a conditional permit application would be applied to the WDNR.

A point-intercept survey occurred during 2006 documenting EWM at 26.3% of sampling locations. That level of EWM existed for a few years in the mid- to late- 2000s and the LLPLD is well aware of the conditions of the lake at that time. The strategy outlined above does not specifically address the EWM population of Long Lake, rather the nuisance conditions that may exist in some areas. If future EWM populations exceed 20% as measured by the point-intercept survey, the nuisance management strategy outlined here would be revisited by the LLPLD. The WDNR will be notified when the trigger is reached and consulted when an alternative management perspective and action is being considered.

The LLPLD would evaluate BMPs available at that time for potentially reducing the EWM population on a lake-wide basis such as a whole-

	lake herbicide treatment. Implementing whole-lake treatments on lakes that have similar morphology to Long Lake have proven difficult, with an increased risk of incorrect dosing. Therefore, this type of management many not be appropriate for Long Lake.
Action Steps:	
	See description steps above.

<u>Management Action:</u>	Conduct Periodic Riparian Stakeholder Surveys
Timeframe:	Every 5-6 years
Facilitator:	Board of Directors, or possibly formation of an Education Committee
Description:	Approximately once every 5-6 years, an updated stakeholder survey would be distributed to Long Lake riparians. Periodically conducting an anonymous stakeholder survey would gather comments and opinions from lake stakeholders to gain important information regarding their understanding of the lake and thoughts on how it should be managed. In regards to EWM management, this would help the LLPLD understand the current level of support for various management strategies (e.g. herbicide treatment) and identify areas of needed education for its constituency. This information would be critical to the development of a realistic plan by supplying an indication of the needs of the stakeholders and their perspective on the management of the lake. A formal anonymous stakeholder survey has not been conducted by the LLPLD to date. It is recommended that he LLPLD work with a qualified consultant to develop the questionnaire, likely using a base survey template from the consultant as a starting point with additional questions and omissions as appropriate. To ensure that the survey questions are not biased or misleading, gaining approval of the survey from a WDNR Research Social Scientist is suggested. If WDNR grant funds are being sought to offset the cost of the survey, WDNR approval is required. Care needs to be taken to maintain anonymity of survey respondents while also ensuring that multiple submissions by a single respondent do not occur. Working with a third-party firm is often suggested.
Action Steps:	
	See description above

Management Goal 2: Monitor Aquatic Vegetation on Long Lake

Management Action:	Coordinate Periodic Point-Intercept Surveys
Timeframe:	Every 3-5 years depending on management strategies being employed
Facilitator:	Board of Directors or Invasive Species Committee
Description:	The point-intercept method as described Wisconsin Department of Natural Resources Bureau of Science Services, PUB-SS-1068 2010 (Hauxwell et al. 2010) have been conducted on Long Lake by the WDNR in 2006 and by Onterra in 2012 and 2017. Based upon guidance from the WDNR, a point spacing (resolution) of 47 meters was used resulting in 1,616 sampling points being evenly distributed across the lake. At each point-intercept location within the *littoral zone* (typically around 400 sampling locations) information regarding the depth, substrate type (soft sediment, sand, or rock), and the plant species sampled along with their relative abundance on the sampling rake is recorded. For all lakes, the WDNR generally recommends that a whole-lake point-intercept survey be conducted once every 5 years. This will allow an understanding of the submergent aquatic plant community dynamics within the Long Lake. This will also allow an understanding of changes in the EWM population for determination if active management should be considered, particularly if EWM populations exceed 20% of the littoral zone as measured by the point-intercept survey. For lakes conducting active management, a whole-lake point-intercept surveys should be conducted at a minimum once every 3 years. In some instances of particularly aggressive active management, the WDNR may require annual point-intercept surveys.
Action Steps:	
	See description above as this is an established program.

Management Action:	Coordinate annual professional monitoring of EWM
Timeframe:	Continuation of current effort
Facilitator:	Board of Directors or Invasive Species Committee
Description:	As the name implies, the EWM peak-biomass survey is completed when the plant is at its peak growth, allowing for a true assessment of the amount of this exotic within the lake. For Long Lake, this survey will likely take place in mid-August to late-September. This survey would include a complete meander survey of the lake's littoral zone by professional ecologists and mapping using GPS technology (sub-meter accuracy is preferred). This survey would serve three main roles: 1) document the EWM population at the peak of its growth stage in a given year, 2) access the management efforts that took place over the growing season, and 3) be used to formulate a management strategy for the following year.
Action Steps:	
	See description above as this is an established program.

Management Action:	Coordinate Periodic Community Mapping (floating-leaf and emergent) Surveys
Timeframe:	Every 10 years unless prompted
Facilitator:	Board of Directors or Invasive Species Committee
Description:	In order to understand the dynamics of the emergent and floating-leaf aquatic plant communities in Long Lake, a community mapping survey would be conducted approximately every 10 years unless a specific rationale prompts a shorter interval. This survey would delineate the margins of floating-leaf (e.g. water lilies) and emergent (e.g. cattails, bulrushes) plant species suing GPS technology (preferably sub-meter accuracy) within Long Lake as well as document the primary species present within each community. Changes in the footprint of these communities can be strong and early indicators of environmental perturbation as well as provide information regarding various habitat types within the system. The community mapping survey has been conducted on Long Lake in 2012 and 2017 as part of the past two management planning projects.
Action Steps:	
	See description above as this is an established program.

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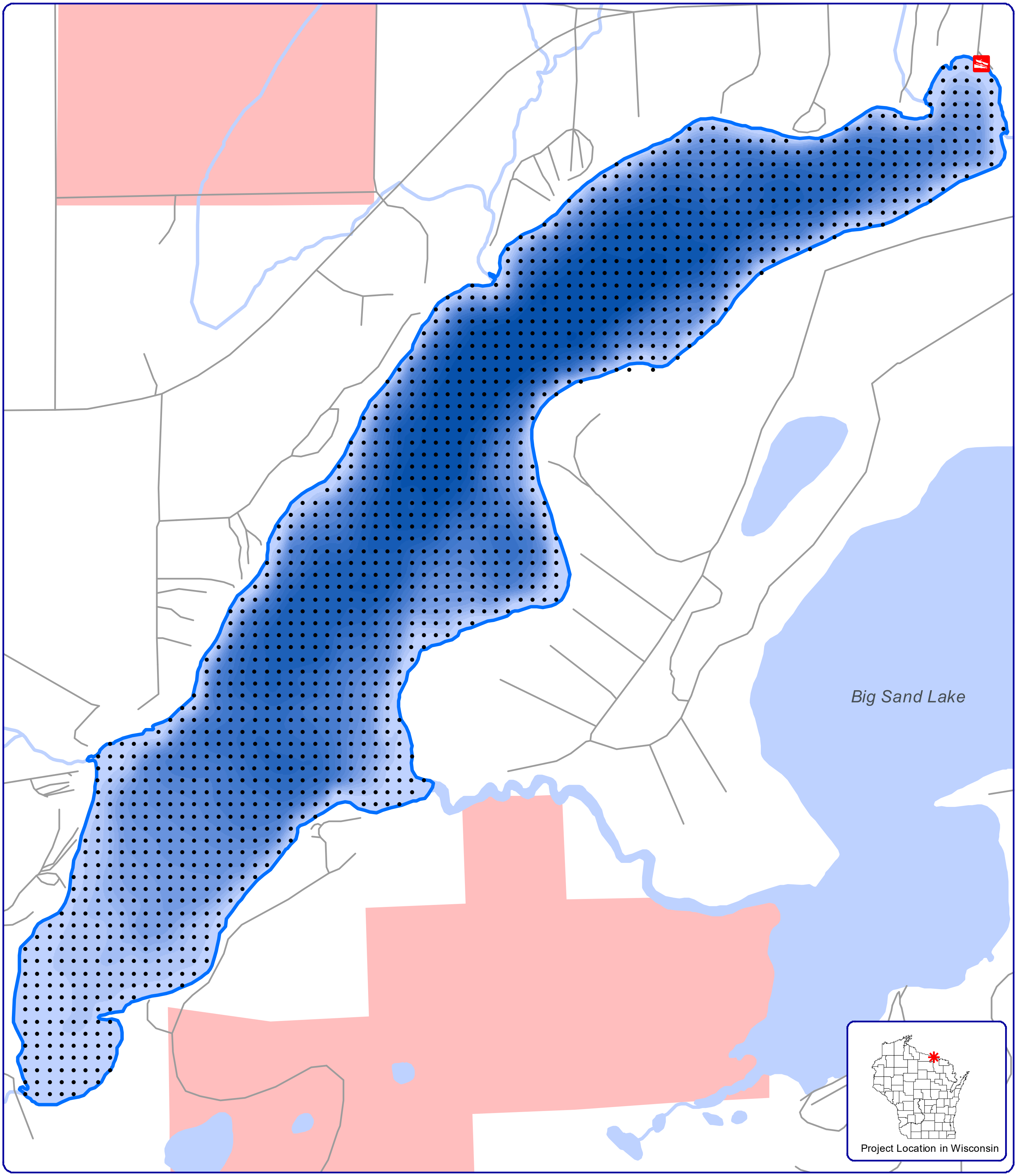


Legend

Long Lake

Point-intercept Location
47-meter spacing

Land Ownership

Federal

State (Parklands)

County *(none shown)*

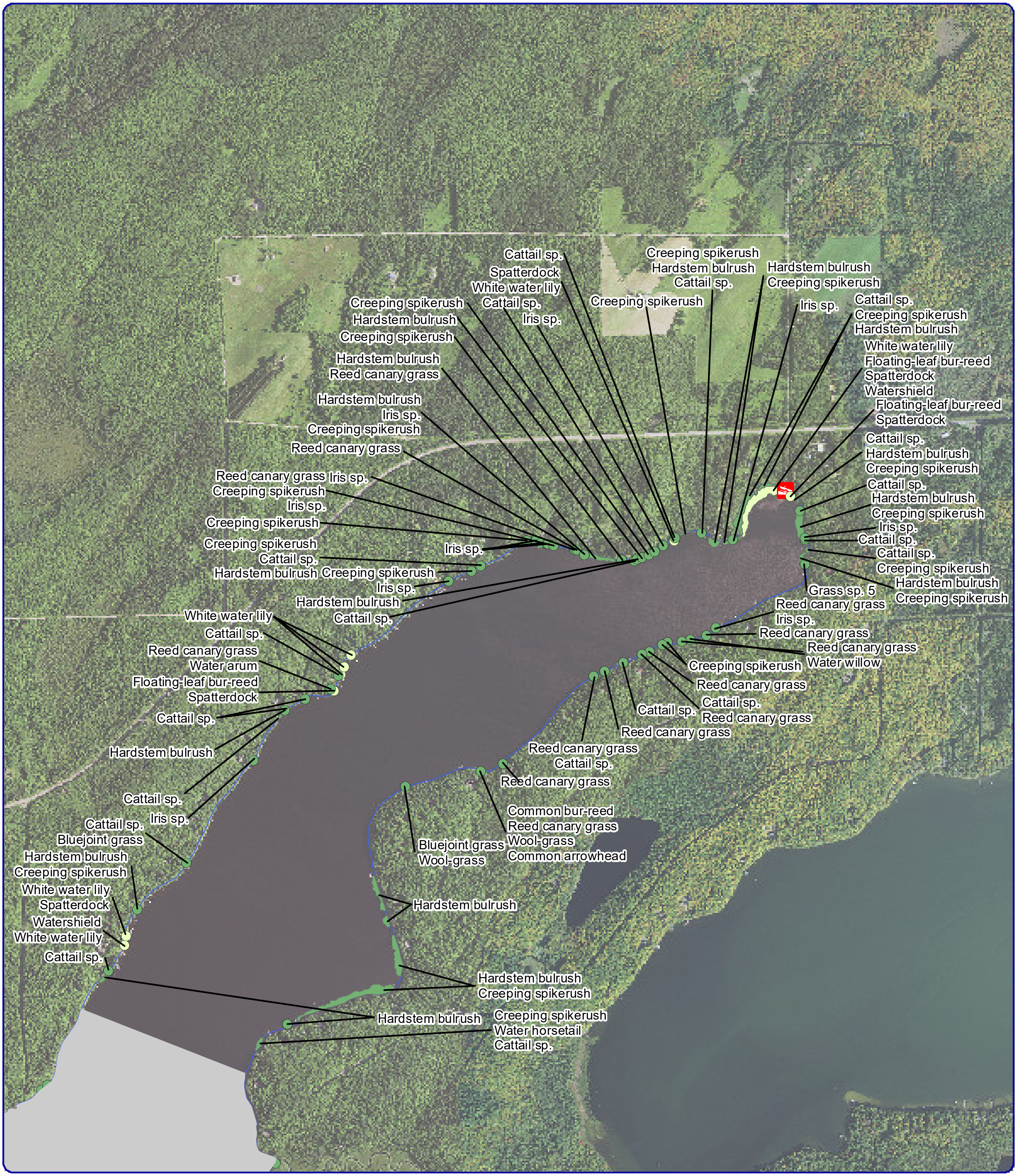
Onterra, LLC
Lake Management Planning
815 Prosper Rd
De Pere, WI 54115
920.338.8860
www.onterra-eco.com

Sources:
Roads, Hydro, & Ownership: WDNR
Bathymetry: Onterra, 2014
Map Date: October 22, 2018

Map 1

Long Lake
Vilas County, Wisconsin

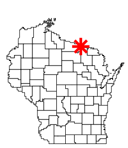
Project Location & Lake Boundaries



N
W E
S

1,750

Feet



Project Location in Wisconsin

Onterra LLC
Lake Management Planning
815 Prosper Road
De Pere, WI 54115
920.338.8860
www.onterra-eco.com

Sources:
Hydro: WDNR
Aquatic Plants: Onterra, 2017
Orthophotography: NAIP, 2015
Map date: October 25, 2017
***Filename:** LongV_Comm_2017_North.mxd*

Legend

Small Plant Communities	Large Plant Communities
Emergent	Emergent
Floating-leaf	Floating-leaf
Mixed Floating-leaf & Emergent	Mixed Floating-leaf & Emergent

Map 2

Long Lake - North

Vilas County, Wisconsin

Aquatic Plant Communities

1,300
Feet

Onterra LLC
Lake Management Planning
815 Prosper Road
De Pere, WI 54115
920.338.8860
www.onterra-eco.com

Sources:
Hydro: WDNR
Aquatic Plants: Onterra, 2017
Orthophotography: NAIP, 2017
Map date: October 25, 2017
***Filename:** LongV_Comm_2017_South.mxd*

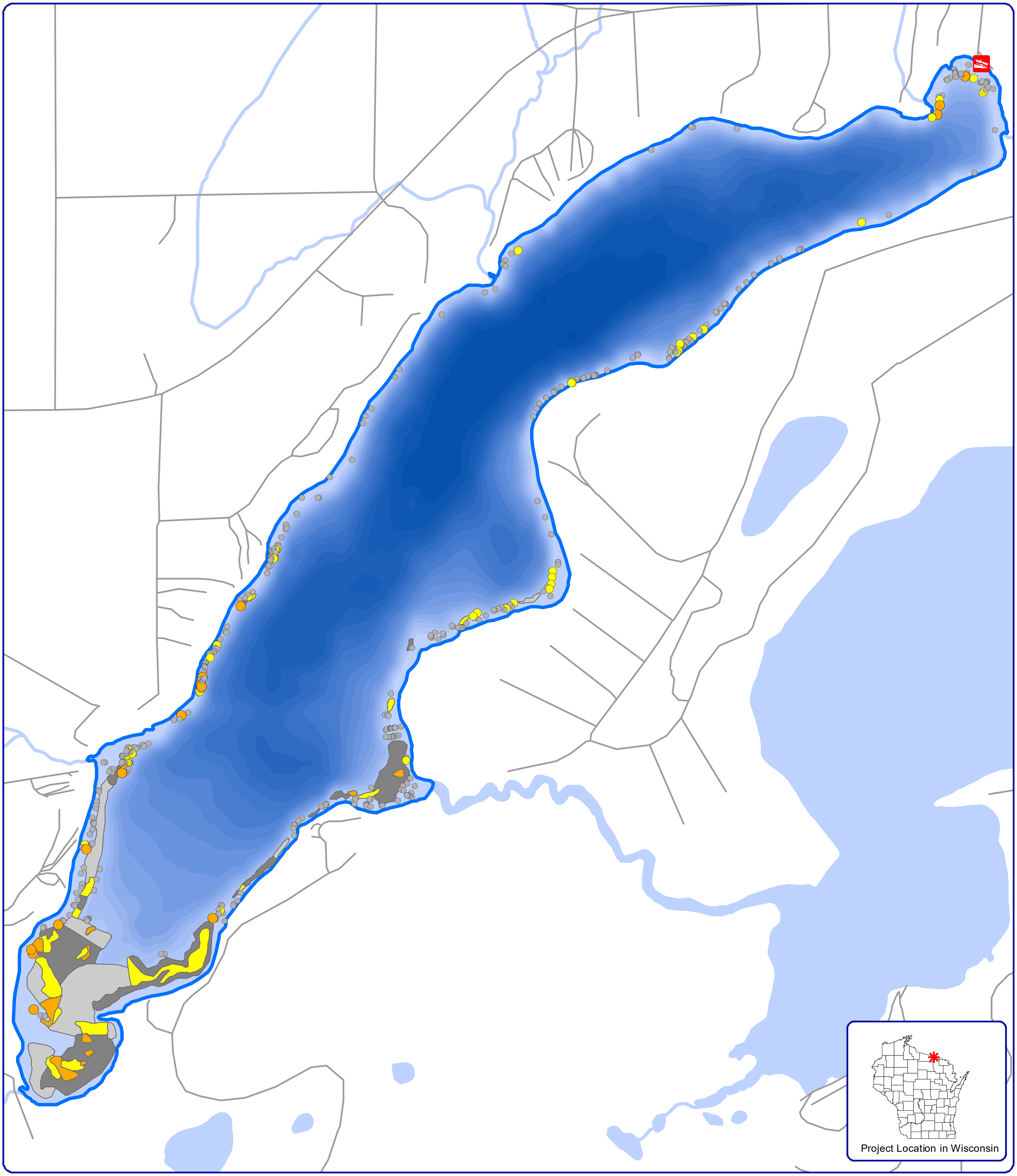
Project Location in Wisconsin

Legend

Small Plant Communities
- Emergent
- Floating-leaf
- Mixed Floating-leaf & Emergent

Large Plant Communities
- Emergent
- Floating-leaf
- Mixed Floating-leaf & Emergent

Map 3
Long Lake - South
Vilas County, Wisconsin
Aquatic Plant Communities



N
W E
S

1,600

Feet

Onterra, LLC
Lake Management Planning
815 Prosper Rd
De Pere, WI 54115
920.338.8860
www.onterra-eco.com

Sources:
Roads & Hydro: WDNR
Bathymetry: Onterra, 2014; procesed by C-Map USA
Aquatic Plants: Onterra, 2018
Map Date: October 22, JLW
Filename: *LongVilas_EWM_Sept18.mxd*

Legend

- Highly Scattered
- Scattered
- Dominant
- Highly Dominant
- Surface Matting *(none found)*
- Single or Few Plants
- Clumps of Plants
- Small Plant Colony

Map 4

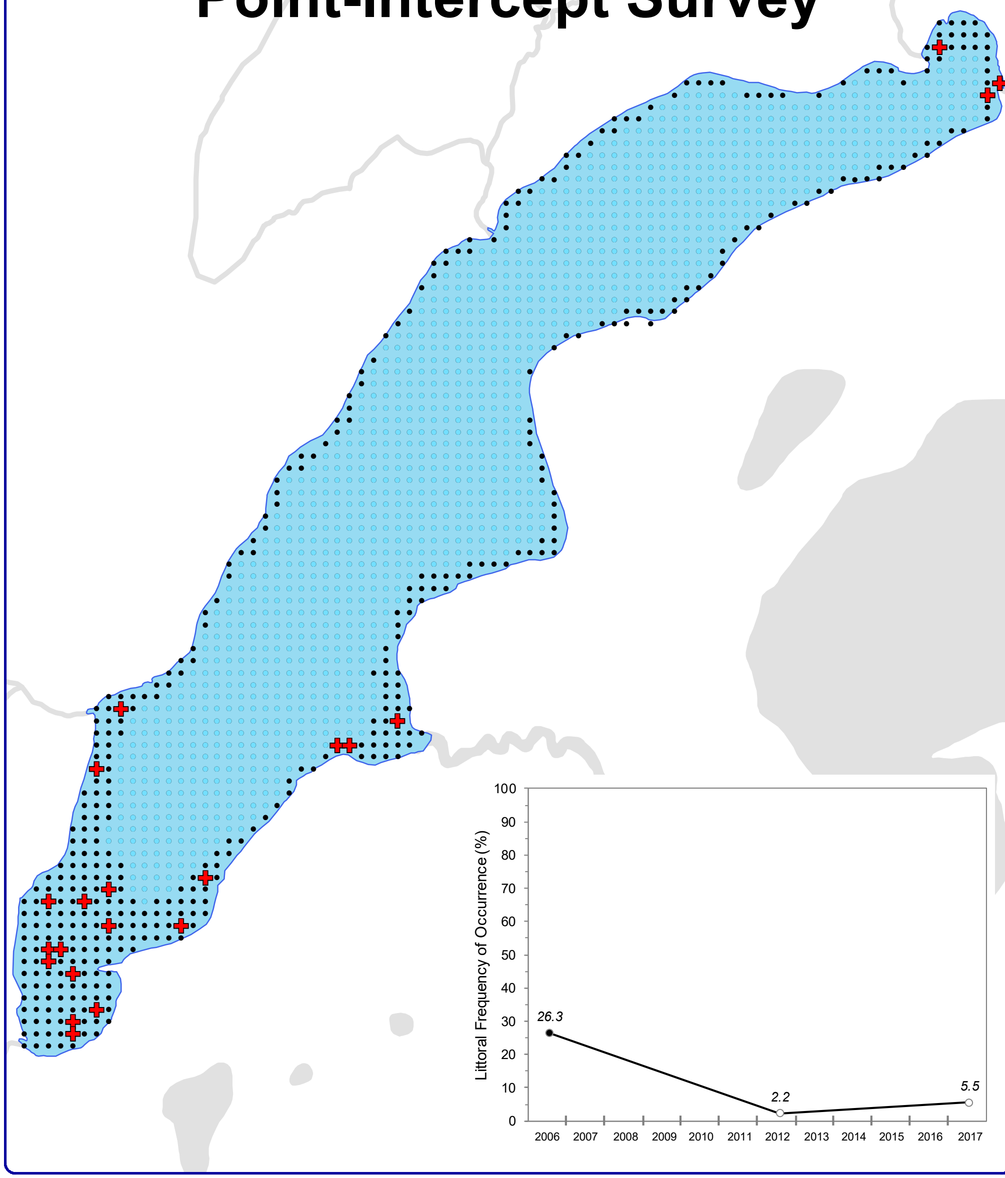
Long Lake

Vilas County, Wisconsin

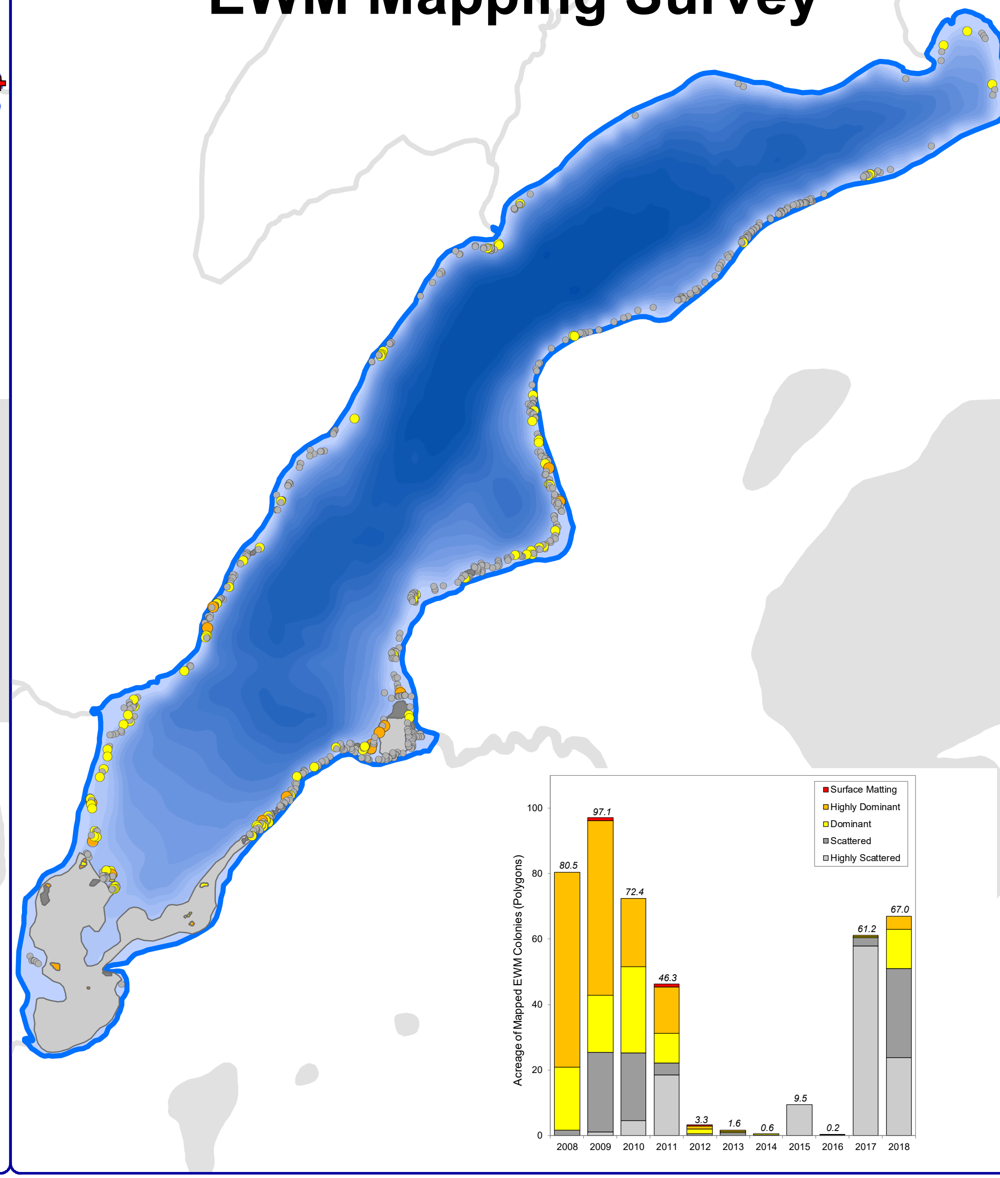
September 2018

EWM Survey Results

Summer 2017 Point-Intercept Survey



Summer 2017 EWM Mapping Survey



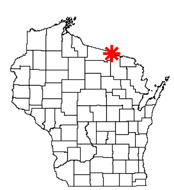
N
W
E
S

700

Feet

Onterra LLC
Lake Management Planning
815 Prosper Road
De Pere, WI 54115
920.338.8860
www.onterra-eco.com

Sources:
Roads and Hydro: WDNR
Bathymetry: Onterra, 2014
Aquatic Plant Survey: Onterra, 2017
Map Date: September 30, 2014
File Name: LongV_EWM_PB_13-14.mxd



Project Location in Wisconsin

Legend

Left Frame

- > Max Depth of Plant Growth
- No EWM, Within Littoral Zone
- EWM Present

Right Frame

- Highly Scattered
- Scattered
- Dominant
- Highly Dominant
- Surface Matting
- Single or Few Plants
- Clump of Plants
- Small Plant Colony

Map 5

Long Lake
Vilas County, Wisconsin

Summer 2017 Comparison EWM Mapping vs PI Survey

A

APPENDIX A

Long Lake 2006, 2012, and 2017 Point-Intercept Survey Results

	Scientific Name	Common Name	LFOO (%) 2006	LFOO (%) 2012	LFOO (%) 2017	2006-2012 % Change	2006-2012 Direction	2012-2017 % Change	2012-2017 Direction	2006-2017 % Change	2006-2017 Direction
Dicots	*Myriophyllum spicatum*	Eurasian watermilfoil	26.3	2.2	5.5	-91.6	▼	148.0	▲	-79.1	▼
	Ceratophyllum demersum	Coontail	4.0	7.6	11.0	90.4	▲	44.0	▲	174.2	▲
	Myriophyllum sibiricum	Northern watermilfoil	5.5	1.0	7.1	-82.1	▼	617.4	▲	28.2	▲
	Bidens beckii	Water marigold	3.0	2.2	2.9	-26.3	▼	29.9	▲	-4.3	▼
	Brasenia schreberi	Watershield	0.3	2.2	2.1	784.5	▲	-5.5	▼	735.6	▲
	Nuphar variegata	Spatterdock	2.0	1.5	1.3	-26.3	▼	-11.4	▼	-34.7	▼
	Utricularia vulgaris	Common bladderwort	0.3	1.2	1.6	391.4	▲	27.5	▲	526.7	▲
	Nymphaea odorata	White water lily	0.3	2.0	0.8	686.2	▲	-60.1	▼	213.4	▲
	Ranunculus flammula	Creeping spearwort	0.5	0.5	0.5	-1.7	▼	6.3	▲	4.5	▲
	Utricularia minor	Small bladderwort	0.0	0.0	0.8		-		▲		▲
	Ranunculus aquatilis	White water crowfoot	0.0	0.0	0.8		-		▲		▲
Non-dicots	*Potamogeton crispus*	Curly-leaf pondweed	0.0	0.0	0.0		-		-		-
	Potamogeton robbinsii	Fern-leaf pondweed	38.6	35.2	17.0	-8.7	▼	-51.7	▼	-55.9	▼
	Elodea canadensis	Common waterweed	29.8	24.4	26.2	-18.2	▼	7.4	▲	-12.2	▼
	Chara spp.	Muskgrasses	10.0	10.1	22.5	0.7	▲	122.9	▲	124.6	▲
	Najas guadalupensis	Southern naiad	0.0	1.2	28.8		▲	2238.2	▲		▲
	Vallisneria americana	Wild celery	11.8	8.1	12.6	-31.0	▼	54.6	▲	6.7	▲
	Najas flexilis	Slender naiad	12.0	19.7	5.2	63.8	▲	-73.4	▼	-56.5	▼
	Potamogeton amplifolius	Large-leaf pondweed	16.3	11.8	6.3	-27.4	▼	-46.9	▼	-61.4	▼
	Nitella spp.	Stoneworts	3.3	9.4	8.1	187.3	▲	-13.3	▼	149.1	▲
	Potamogeton richardsonii	Clasping-leaf pondweed	2.0	6.9	8.4	244.0	▲	21.5	▲	317.8	▲
	Potamogeton gramineus	Variable-leaf pondweed	2.0	5.9	8.4	194.8	▲	41.7	▲	317.8	▲
	Eleocharis acicularis	Needle spikerush	2.5	3.9	5.0	57.2	▲	26.2	▲	98.5	▲
	Potamogeton pusillus	Small pondweed	4.5	5.7	2.6	25.6	▲	-53.8	▼	-42.0	▼
	Potamogeton praelongus	White-stem pondweed	0.8	2.2	3.9	194.8	▲	77.1	▲	422.3	▲
	Potamogeton hybrid 1	Pondweed Hybrid 1	0.0	0.0	4.5		-		▲		▲
	Potamogeton strictifolius	Stiff pondweed	5.8	0.2	0.3	-95.7	▼	6.3	▲	-95.5	▼
	Potamogeton zosteriformis	Flat-stem pondweed	3.5	1.2	0.5	-64.9	▼	-57.5	▼	-85.1	▼
	Potamogeton spirillus	Spiral-fruited pondweed	3.8	1.2	0.3	-67.2	▼	-78.7	▼	-93.0	▼
	Isoetes spp.	Quillwort spp.	0.8	0.7	2.1	-1.7	▼	183.4	▲	178.5	▲
	Sagittaria sp. (rosette)	Arrowhead sp. (rosette)	1.8	0.0	1.8	-100.0	▼		▲	4.5	▲
	Heteranthera dubia	Water stargrass	1.0	0.7	1.3	-26.3	▼	77.1	▲	30.6	▲
	Filamentous algae	Filamentous algae	0.0	1.0	1.0		▲	6.3	▲		▲
	Eleocharis palustris	Creeping spikerush	0.5	0.5	1.0	-1.7	▼	112.6	▲	108.9	▲
	Fissidens spp. & Fontinalis spp.	Aquatic Moss	0.0	0.2	1.3		▲	431.4	▲		▲
	Schoenoplectus acutus	Hardstem bulrush	0.8	0.2	0.8	-67.2	▼	218.8	▲	4.5	▲
	Potamogeton foliosus	Leafy pondweed	0.5	1.5	0.3	194.8	▲	-82.3	▼	-47.8	▼
	Potamogeton natans	Floating-leaf pondweed	0.8	1.2	0.0	63.8	▲	-100.0	▼	-100.0	▼
	Juncus pelocarpus	Brown-fruited rush	0.0	0.5	0.5		▲	6.3	▲		▲
	Schoenoplectus tabernaemontani	Softstem bulrush	0.5	0.7	0.0	47.4	▲	-100.0	▼	-100.0	▼
	Sparganium angustifolium	Narrow-leaf bur-reed	0.0	0.0	0.5		-		▲		▲
	Sparganium emersum var. acaule	Short-stemmed bur-reed	0.0	0.0	0.3		-		▲		▲
	Sparganium androcladum	Shining bur-reed	0.0	0.0	0.3		-		▲		▲
	Potamogeton friesii	Fries' pondweed	0.0	0.0	0.3		-		▲		▲
	Sparganium americanum	American bur-reed	0.0	0.2	0.0		▲	-100.0	▼		-
	Potamogeton epihydrus	Ribbon-leaf pondweed	0.3	0.0	0.0	-100.0	▼		-	-100.0	▼
	Freshwater sponge	Freshwater sponge	0.0	0.2	0.0		▲	-100.0	▼		-
	Equisetum fluviatile	Water horsetail	0.0	0.2	0.0		▲	-100.0	▼		-

▲ or ▼ = Change Statistically Valid (Chi-square; α = 0.05)

▲ or ▼ = Change Statistically Valid (Chi-square; α = 0.05)

B

APPENDIX B

Extracted Relevant Chapters from Aquatic Plant Management in Wisconsin: Draft Strategic Analysis – Draft December 2018

The WDNR is in the process of conducting a Strategy Analysis which will ultimately mold policies and approaches. The strategy the WDNR is following is outlined on the WDNR's APM Strategic Analysis Webpage:

https://dnr.wi.gov/topic/eia/apmsa.html

Below is a table of contents for the extracted materials for use in risk assessment of the discussed management tools within this project. Please refer to the WDNR's full text document cited above for Literature Cited.

Extracted Table of Contents

S.3.3. Herbicide Treatment

Herbicides are the most commonly employed method for controlling aquatic plants in Wisconsin. They are extremely useful tools for accomplishing aquatic plant management (APM) goals, like controlling invasive species, providing waterbody access, and ecosystem restoration. This Chapter includes basic information about herbicides and herbicide formulations, how herbicides are assessed for ecological and human health risks and registered for use, and some important considerations for the use of herbicides in aquatic environments.

A pesticide is a substance used to either directly kill pests or to prevent or reduce pest damage; herbicides are pesticides that are used to kill plants. Only a certain component of a pesticide product is intended to have pesticidal effects and this is called the active ingredient. The active ingredient is listed near the top of the first page on an herbicide product label. Any product claiming to have pesticidal properties must be registered with the U.S. EPA and regulated as a pesticide.

Inert ingredients often make up the majority of a pesticide formulation and are not intended to have pesticidal activity, although they may enhance the pesticidal activity of the active ingredient. These ingredients, such as carriers and solvents, are often added to the active ingredient by manufacturers, or by an herbicide applicator during use, in order to allow mixing of the active ingredient into water, make it more chemically stable, or aid in storage and transport. Manufacturers are not required to identify the specific inert ingredients on the pesticide label. In addition to inert ingredients included in manufactured pesticide formulations, adjuvants are inert ingredient products that may be added to pesticide formulations before they are applied to modify the properties or enhance pesticide performance. Adjuvants are typically not intended to have pesticidal properties and are not regulated as pesticides under the Federal Insecticide, Fungicide and Rodenticide Act. However, research has shown that inert ingredients can increase the efficacy and toxicity of pesticides especially if the appropriate label uses aren't followed (Mesnage et al. 2013; Defarge et al. 2016).

The combination of active ingredients and inert ingredients is what makes up a pesticide formulation. There are often many formulations of each active ingredient and pesticide manufacturers typically give a unique product or trade name to each specific formulation of an active ingredient. For instance, "Sculpin G" is a solid, granular 2,4-D amine product, while "DMA IV" is a liquid amine 2,4-D product, and the inert ingredients in these formulations are different, but both have the same active ingredient. Care should always be taken to read the herbicide product label as this will give information about which pests and ecosystems the product is allowed to be used for. Some formulations (i.e., non-aquatic formulations of glyphosate such as "Roundup") are not allowed for aquatic use and could lead to environmental degradation even if used on shorelines near the water. There are some studies which indicate that the combination of two chemicals (e.g., 2,4-D and endothall) applied together produces synergistic efficacy results that are greater than if each product was applied alone (Skogerboe et al. 2012). Conversely, there are studies which indicate the combination of two chemicals (i.e. diquat and penxosulam) which result in an antagonistic response between the herbicides, and resulted in reduced efficacy than when applying penoxsulam alone (Wersal and Madsen 2010b).

The U.S. EPA is responsible for registering pesticide products before they may be sold. In order to have their product registered, pesticide manufacturers must submit toxicity test data to the EPA that shows that the intended pesticide use(s) will not create unreasonable risks. "Unreasonable" in this context means that the risks of use outweigh the potential benefits. Once registered, the EPA must re-evaluate each pesticide and new information related to its use every 15 years. The current cycle of registration review will end in 2022, with a new cycle and review schedule starting then. In addition, EPA may decide to only register certain uses of any given pesticide product and can also require that only trained personnel can apply a pesticide before the risks outweigh the benefits. Products requiring training before application are called Restricted Use Pesticides.

As part of their risk assessments, EPA reviews information related to pesticide toxicity. Following laboratory testing, ecotoxicity rankings are given for different organismal groups based on the dosage that would cause harmful ecological effects (e.g., death, reduction in growth, reproductive impairment, and others). For example, the ecotoxicity ranking for 2,4-D ranges from "practically non-toxic" to "slightly toxic" for freshwater invertebrates, meaning tests have shown that doses of >100 ppm and 10-100 ppm are needed to cause 50% mortality or immobilization in the test population, respectively. Different dose ranges and indicators of "harm" are used to assess toxicity depending on the organisms being tested. More information can be found on the EPA's website.

Beyond selecting herbicide formulations approved for use in aquatic environments, there are additional factors to consider supporting appropriate and effective herbicide use in those environments. Herbicide treatments are often used in terrestrial restorations, so they are also often requested in the management and restoration of aquatic plant communities. However, unlike applications in a terrestrial environment, the fluid environment of freshwater systems presents a set of unique challenges. Some general best practices for addressing challenges associated with herbicide dilution, migration, persistence, and non-target impacts are described in Chapter 7.4. More detailed documentation of these challenges is described below and in discussions on individual herbicides in Supplemental Chapter S.3.3 (Herbicide Treatment).

As described in Chapter 7.4, when herbicide is applied to waters, it can quickly migrate offsite and dilute to below the target concentrations needed to provide control (Hoeppel and Westerdal 1983; Madsen et al. 2015; Nault et al. 2015). Successful plant control with herbicide is dependent on concentration exposure time (CET) relationships. In order to examine actual observed CET relationships following herbicide applications in Wisconsin lakes, a study of herbicide CET and Eurasian watermilfoil (*Myriophyllum spicatum*) control efficacy was conducted on 98 small-scale (0.1-10 acres) 2,4-D treatment areas across 22 lakes. In the vast majority of cases, initial observed 2,4-D concentrations within treatment areas were far below the applied target concentration, and then dropped below detectable limits within a few hours after treatment (Nault et al. 2015). These results indicate the rapid dissipation of herbicide off of the small treatment areas resulted in water column concentrations which were much lower than those recommended by previous laboratory CET studies for effective Eurasian watermilfoil control. Concentrations in protected treatment areas (e.g., bays, channels) were initially higher than those in areas more exposed to wind and waves, although concentrations quickly dissipated to below detectable limits within hours after treatment regardless of spatial location. Beyond confining small-scale treatments to protected areas, utilizing or integrating faster-acting herbicides with shorter CET requirements may also help to compensate for reductions in plant control due to dissipation (Madsen et al. 2015). The use of chemical curtains or adjuvants (weighting or sticking agents) may also help to maintain adequate CET, however more research is needed in this area.

This rapid dissipation of herbicide off of treatment areas is important for resource managers to consider in planning, as treating numerous targeted areas at a 'localized' scale may actually result in low-concentrations capable of having lakewide impacts as the herbicide dissipates off of the individual treatment sites. In general, if the percentage of treated areas to overall lake surface area is >5% and targeted areas are treated at relatively high 2,4-D concentrations (e.g., 2.0-4.0 ppm), then anticipated lakewide concentrations after dissipation should be calculated to determine the likelihood of lakewide effects (Nault et al. 2018).

Aquatic-use herbicides are commercially available in both liquid and granular forms. Successful target species control has been reported with both granular and liquid formulations. While there has been a commonly held belief that granular products are able to 'hold' the herbicide on site for longer periods of time, actual field comparisons between granular and liquid 2,4-D forms revealed that they dissipated similarly when applied at small-scale sites (Nault et al. 2015). In fact, liquid 2,4-D had higher initial observed water column concentrations than the granular form, but in the majority of cases concentrations of both forms decreased rapidly to below detection limits within several hours after treatment Nault et al. 2015). Likewise, according to United Phosphorus, Inc. (UPI), the sole manufacturer of endothall, the granular formulation of endothall does not hold the product in a specific area significantly longer than the liquid form (Jacob Meganck [UPI], *personal communication*).

In addition, the stratification of water and the formation of a thermal density gradient can confine the majority of applied herbicides in the upper, warmer water layer of deep lakes. In some instances, the entire lake water volume is used to calculate how much active ingredient should be applied to achieve a specific lakewide target concentration. However, if the volume of the entire lake is used to calculate application rates for stratified lakes, but the chemical only readily mixes into the upper water layer, the achieved lakewide concentration is likely to be much higher than the target concentration, potentially resulting in unanticipated adverse ecological impacts.

Because herbicides cannot be applied directly to specific submersed target plants, the dissipation of herbicide over the treatment area can lead to direct contact with non-target plants and animals. No herbicide is completely selective (i.e., effective specifically on only a single target species). Some plant species may be more susceptible to a given herbicide than others, highlighting the importance of choosing the appropriate herbicide, or other non-chemical management approach, to minimize potential non-target effects of treatment. There are many herbicides and plant species for which the CET relationship that would negatively affect the plant is unknown. This is particularly important in the case of rare, special concern, or threatened and endangered species. Additionally, loss of habitat following any herbicide treatment or other management technique may cause indirect reductions in populations of invertebrates or other organisms. Some organisms will only recolonize the managed areas as aquatic plants become re-established.

Below are reviews for the most commonly used herbicides for APM in Wisconsin. Much of the information here was pulled directly from DNR's APM factsheets (http://dnr.wi.gov/lakes/plants/factsheets/), which were compiled in 2012 using U.S. EPA herbicide product labels, U.S. Army Corps of Engineers reports, and communications with natural resource agencies in other northern, lake-rich states. These have been supplemented with more recent information from primary research publications.

Each pesticide has at least one mode of action which is the specific mechanism by which the active ingredient exerts a toxic effect. For example, some herbicides inhibit production of the pigments needed for photosynthesis while others mimic plant growth hormones and cause uncontrolled and unsustainable growth. Herbicides are often classified as either systemic or contact in mode of action, although some herbicides are able to function under various modes of action depending on environmental variables such as water temperature. Systemic pesticides are those that are absorbed by organisms and can be moved or translocated within the organism. Contact pesticides are those that exert toxic effects on the part(s) of an organism that they come in contact with. The amount of exposure time needed to kill an organism is based on the specific mode of action and the concentration of any given pesticide. In the descriptions below herbicides are generally categorized into which environment (above or below water) they are primarily used and a relative assessment of how quickly they impact plants. Herbicides can be applied in many ways. In lakes, they are usually applied to the water's surface (or below the water's surface) through controlled release by equipment including spreaders, sprayers, and underwater hoses. In wetland environments, spraying by helicopter, backpack sprayer, or application by cut-stem dabbing, wicking, injection, or basal bark application are also used.

S.3.3.1. Submersed or Floating, Relatively Fast-Acting Herbicides

Diquat

Registration and Formulations

Diquat (or diquat dibromide) initially received Federal registration for control of submersed and floating aquatic plants in 1962. It was initially registered with the U.S. EPA in 1986, evaluated for reregistration in 1995, and is currently under registration review. A registration review decision was expected in 2015 but has not been released (EPA Diquat Plan 2011). The active ingredient is 6,7-dihydrodipyrido[1,2-α:2',1'-c] pyrazinediium dibromide, and is commercially sold as liquid formulations for aquatic use.

Mode of Action and Degradation

Diquat is a fast-acting herbicide that works through contact with plant foliage by disrupting electron flow in photosystem I of the photosynthetic reaction, ultimately causing the destruction of cell membranes (Hess 2000; WSSA 2007). Plant tissues in contact with diquat become impacted within several hours after application, and within one to three days the plant tissue will become necrotic. Diquat is considered a non-selective herbicide and will rapidly kill a wide variety of plants on contact. Because diquat is a fast-acting herbicide, it is oftentimes used for managing plants growing in areas where water exchange is anticipated to limit herbicide exposure times, such as small-scale treatments.

Due to rapid vegetation decomposition after treatment, only partial treatments of a waterbody should be conducted to minimize dissolved oxygen depletion and associated negative impacts on fish and other aquatic organisms. Untreated areas can be treated with diquat 14 days after the first application.

Diquat is strongly attracted to silt and clay particles in the water and may not be very effective under highly turbid water conditions or where plants are covered with silt (Clayton and Matheson 2010).

The half-life of diquat in water generally ranges from a few hours to two days depending on water quality and other environmental conditions. Diquat has been detected in the water column from less than a day up towards 38 DAT, and remains in the water column longer when treating waterbodies with sandy sediments with lower organic matter and clay content (Coats et al. 1964; Grzenda et al. 1966; Yeo 1967; Sewell et al. 1970; Langeland and Warner 1986; Langeland et al. 1994; Poovey and Getsinger 2002; Parsons et al. 2007; Gorzerino et al. 2009; Robb et al. 2014). One study reported that diquat is chemically stable within a pH range of 3 to 8 (Florêncio et al. 2004). Due to the tendency of diquat to be rapidly adsorbed to suspended clays and particulates, long exposure periods are oftentimes not possible to achieve in the field. Studies conducted by Wersal et al. (2010a) did not observe differences in target species efficacy between daytime versus night-time applications of diquat. While large-scale diquat treatments are typically not implemented, a study by Parsons et al. (2007), observed declines in both dissolved oxygen and water clarity following the herbicide treatment.

Diquat binds indefinitely to organic matter, allowing it to accumulate and persist in the sediments over time (Frank and Comes 1967; Simsiman and Chesters 1976). It has been reported to have a very long-lived half-life (1000 days) in sediment because of extremely tight soil sorption, as well as an extremely low rate of degradation after association with sediment (Wauchope et al. 1992; Peterson et al. 1994). Both photolysis and microbial degradation are thought to play minor roles in degradation (Smith and Grove 1969; Emmett 2002). Diquat is not known to leach into groundwater due to its very high affinity to bind to soils.

One study reported that combinations of diquat and penoxsulam resulted in an antagonistic response between the herbicides when applied to water hyacinth (*Eichhornia crassipes*) and resulted in reduced efficacy than when applying penoxsulam alone. The antagonistic response is likely due to the rapid cell destruction by diquat that limits the translocation and efficacy of the slower acting enzyme inhibiting herbicides (Wersal and Madsen 2010b).

Toxicology

There are no restrictions on swimming or eating fish from waterbodies treated with diquat. Depending on the concentration applied, there is a 1-3 day waiting period after treatment for drinking water. However, in one study, diquat persisted in the water at levels above the EPA drinking water standard for at least 3 DAT, suggesting that the current 3-day drinking water restriction may not be sufficient under all application scenarios (Parsons et al. 2007). Water treated with diquat should not be used for pet or livestock drinking water for one day following treatment. The irrigation restriction for food crops is five days, and for ornamental plants or lawn/turf, it varies from one to three days depending on the concentration used. A study by Mudge et al. (2007)

on the effects of diquat on five popular ornamental plant species (begonia, dianthus, impatiens, petunia, and snapdragon) found minimal risks associated with irrigating these species with water treated with diquat up to the maximum use rate of 0.37 ppm.

Ethylene dibromide (EDB) is a trace contaminant in diquat products which originates from the manufacturing process. EDB is a documented carcinogen, and the EPA has evaluated the health risk of its presence in formulated diquat products. The maximum level of EDB in diquat dibromide is 0.01 ppm (10 ppb). EBD degrades over time, and it does not persist as an impurity.

Diquat does not have any apparent short-term effects on most aquatic organisms that have been tested at label application rates (EPA Diquat RED 1995). Diquat is not known to bioconcentrate in fish tissues. A study using field scenarios and well as computer modelling to examine the potential ecological risks posed by diquat determined that diquat poses a minimal ecological impact to benthic invertebrates and fish (Campbell et al. 2000). Laboratory studies indicate that walleye (*Sander vitreus*) are more sensitive to diquat than some other fish species, such as smallmouth bass (*Micropterus dolomieu),* largemouth bass (*Micropterus salmoides*), and bluegills (*Lepomis macrochirus*), with individuals becoming less sensitive with age (Gilderhus 1967; Paul et al. 1994; Shaw and Hamer 1995). Maximum application rates were lowered in response to these studies, such that applying diquat at recommended label rates is not expected to result in toxic effects on fish (EPA Diquat RED 1995). Sublethal effects such as respiratory stress or reduced swimming capacity have been observed in studies where certain fish species (e.g., yellow perch (*Perca flavescens*), rainbow trout (*Oncorhynchus mykiss*), and fathead minnows (*Pimephales promelas*)) have been exposed to diquat concentrations (Bimber et al. 1976; Dodson and Mayfield 1979; de Peyster and Long 1993). Another study showed no observable effects on eastern spiny softshell turtles (*Apalone spinifera spinifera*; Paul and Simonin 2007). Reduced size and pigmentation or increased mortality have been shown in some amphibians but at above recommended label rates (Anderson and Prahlad 1976; Bimber and Mitchell 1978; Dial and Bauer-Dial 1987). Toxicity data on invertebrates are scarce and diquat is considered not toxic to most of them. While diquat is not highly toxic to most invertebrates, significant mortality has been observed in some species at concentrations below the maximum label use rate for diquat, such as the amphipod *Hyalella azteca* (Wilson and Bond 1969; Williams et al. 1984), water fleas (*Daphnia* spp.). Reductions in habitat following treatment may also contribute to reductions of *Hyalella azteca*. For more information, a thorough risk assessment for diquat was compiled by the Washington State Department of Ecology Water Quality Program (WSDE 2002). Available toxicity data for fish, invertebrates, and aquatic plants is summarized in tabular format by Campbell et al. (2000).

Species Susceptibility

Diquat has been shown to control a variety of invasive submerged and floating aquatic plants, including Eurasian watermilfoil (*Myriophyllum spicatum*), curly-leaf pondweed (*Potamogeton crispus*), parrot feather (*Myriophyllum aquaticum*), Brazilian waterweed (*Egeria densa*), water hyacinth, water lettuce (*Pistia stratiotes*), flowering rush (*Butomus umbellatus*), and giant salvinia (*Salvinia molesta*; Netherland et al. 2000; Nelson et al. 2001; Poovey et al. 2002; Langeland et al. 2002; Skogerboe et al. 2006; Martins et al. 2007, 2008; Wersal et al. 2010a; Wersal and Madsen 2010a; Wersal and Madsen 2012; Poovey et al. 2012; Madsen et al. 2016). Studies conducted on the use of diquat for hydrilla (*Hydrilla verticillata*) and fanwort (*Cabomba caroliniana*) control have resulted in mixed reports of efficacy (Van et al. 1987; Langeland et al. 2002; Glomski et al. 2005; Skogerboe et al. 2006; Bultemeier et al. 2009; Turnage et al. 2015). Non-native phragmites (*Phragmites australis* subsp. *australis*) has been shown to not be significantly reduced by diquat (Cheshier et al. 2012).

Skogerboe et al. 2006 reported on the efficacy of diquat (0.185 and 0.37 ppm) under flow-through conditions (observed half-lives of 2.5 and 4.5 hours, respectively). All diquat treatments reduced Eurasian watermilfoil biomass by 97 to 100% compared to the untreated reference, indicating that this species is highly susceptible to diquat. Netherland et al. (2000) examined the role of various water temperatures (10, 12.5, 15, 20, and 25°C) on the efficacy of diquat applications for controlling curly-leaf pondweed. Diquat was applied at rates of 0.16-0.50 ppm, with exposure times of 9-12 hours. Diquat efficacy on curly-leaf pondweed was inhibited as water temperature decreased, although treatments at all temperatures were observed to significantly reduce biomass and turion formation. While the most efficacious curly-leaf pondweed treatments were conducted at 25°C, waiting until water warms to this temperature limits the potential for reducing turion production. Diquat applied at 0.37 ppm (with a 6 to 12-hour exposure time) or at 0.19 ppm (with a 72-hour exposure time) was effective at reducing biomass of flowering rush (Poovey et al. 2012; Madsen et al. 2016).

Native species that have been shown to be affected by diquat include: American lotus (*Nelumbo lutea*), common bladderwort (*Utricularia vulgaris*), coontail (*Ceratophyllum demersum*), common waterweed (*Elodea canadensis*), needle spikerush (*Eleocharis acicularis*), Illinois pondweed (*Potamogeton illinoensis*), leafy pondweed (*P. foliosus*), clasping-leaf pondweed (*P. richardsonii*), fern pondweed (*P. robbinsii*), sago pondweed (*Stuckenia pectinata*), and slender naiad (*Najas flexilis*) (Hofstra et al. 2001; Glomski et al. 2005; Skogerboe et al. 2006; Mudge 2013; Bugbee et al. 2015; Turnage et al. 2015). Diquat is particularly toxic to duckweeds (*Landoltia punctata* and *Lemna* spp.), although certain populations of dotted duckweed (*Landoltia punctata*) have developed resistance of diquat in waterbodies with a long history (20-30 years) of repeated diquat treatments (Peterson et al. 1997; Koschnick et al. 2006). Variable effects have been observed for water celery (*Vallisneria americana*), long-leaf pondweed (*Potamogeton nodosus*), and variable-leaf watermilfoil (*Myriophyllum heterophyllum*; Skogerboe et al. 2006; Glomski and Netherland 2007; Mudge 2013).

Flumioxazin

Registration and Formulations

Flumioxazin (2-[7-fluoro-3,4-dihydro-3-oxo-4-(2-propynyl)-2H-1,4-benzoxazin-6-yl]-4,5,6,7-tetrahydro-1H-isoindole-1,3(2H)-dione) was registered with the U.S. EPA for agricultural use in 2001 and registered for aquatic use in 2010. The first registration review of flumioxazin is expected to be completed in 2017 (EPA Flumioxazin Plan 2011). Granular and liquid formulations are available for aquatic use.

Mode of Action and Degradation

The mode of action of flumioxazin is through disruption of the cell membrane by inhibiting protoporphyrinogen oxidase which blocks production of heme and chlorophyll. The efficacy of this mode of action is dependent on both light intensity and water pH (Mudge et al. 2012a; Mudge and Haller 2010; Mudge et al. 2010), with herbicide degradation increasing with pH and efficacy decreasing as light intensity declines.

Flumioxazin is broken down by water (hydrolysis), light (photolysis) and microbes. The half-life ranges from approximately 4 days at pH 5 to 18 minutes at pH 9 (EPA Flumioxazin 2003). In the majority of Wisconsin lakes half-life should be less than 1 day.

Flumioxazin degrades into APF (6-amino-7-fluro-4-(2-propynyl)-1,4,-benzoxazin-3(2H)-one) and THPA (3,4,5,6-tetrahydrophthalic acid). Flumioxazin has a low potential to leach into groundwater due to the very quick hydrolysis and photolysis. APF and THPA have a high potential to leach through soil and could be persistent.

Toxicology

Tests on warm and cold-water fishes indicate that flumioxazin is "slightly to moderately toxic" to fish on an acute basis, with possible effects on larval growth below the maximum label rate of 0.4 ppm (400 ppb). Flumioxazin is moderately to highly toxic to aquatic invertebrates, with possible impacts below the maximum label rate. The potential for bioaccumulation is low since degradation in water is so rapid. The metabolites APF and THPA have not been assessed for toxicity or bioaccumulation.

The risk of acute exposure is primarily to chemical applicators. Concentrated flumioxazin doesn't pose an inhalation risk but can cause skin and eye irritation. Recreational water users would not be exposed to concentrated flumioxazin.

Acute exposure studies show that flumioxazin is "practically non-toxic" to birds and small mammals. Chronic exposure studies indicate that flumioxazin is non-carcinogenic. However, flumioxazin may be an endocrine disrupting compound in mammals (EPA Flumioxazin 2003), as some studies on small mammals did show effects on reproduction and larval development, including reduced offspring viability, cardiac and skeletal malformations, and anemia. It does not bioaccumulate in mammals, with the majority excreted in a week.

Species Susceptibility

The maximum target concentration of flumioxazin is 0.4 ppm (400 ppb). At least one study has shown that flumioxazin (at or below the maximum label rate) will control the invasive species fanwort (*Cabomba caroliniana*), hydrilla (*Hydrilla verticillata*), Japanese stiltgrass (*Microstegium vimineum*), Eurasian watermilfoil (*Myriophyllum spicatum*), water lettuce (*Pistia stratiotes*), curly-leaf pondweed (*Potamogeton crispus*), and giant salvinia (*Salvinia molesta*), while water hyacinth (*Eichhornia crassipes*) and water pennyworts (*Hydrocotyle* spp.) do not show significant impacts (Bultemeier et al. 2009; Glomski and Netherland 2013a; Glomski and Netherland 2013b; Mudge 2013; Mudge and Netherland 2014; Mudge and Haller 2012; Mudge and Haller 2010). Flowering rush (*Butomus umbellatus*; submersed form) showed mixed success in herbicide trials

(Poovey et al. 2012; Poovey et al. 2013). Native species that were significantly impacted (in at least one study) include coontail (*Ceratophyllum demersum*), water stargrass (*Heteranthera dubia*), variable-leaf watermilfoil (*Myriophyllum heterophyllum*), America lotus (*Nelumbo lutea*), pond-lilies (*Nuphar* spp.), white waterlily (*Nymphaea odorata*), white water crowfoot (*Ranunculus aquatilis*), and broadleaf cattail (*Typha latifolia*), while common waterweed (*Elodea canadensis*), squarestem spikerush (*Eleocharis quadrangulate*), horsetail (*Equisetum hyemale*), southern naiad (*Najas guadalupensis*), pickerelweed (*Pontederia cordata*), Illinois pondweed (*Potamogeton illinoensis*), long-leaf pondweed (*P. nodosus*), broadleaf arrowhead (*Sagittaria latifolia*), hardstem bulrush (*Schoenoplectus acutus*), common three-square bulrush (*S. pungens*), softstem bulrush (*S. tabernaemontani*), sago pondweed (*Stuckenia pectinata*), and water celery (*Vallisneria americana*) were not impacted relative to controls. Other species are likely to be susceptible, for which the effects of flumioxazin have not yet been evaluated.

S.3.3.2. Submersed, Relatively Slow-Acting Herbicides

2,4-D

Registration and Formulations

2,4-D is an herbicide that is widely used as a household weed-killer, agricultural herbicide, and aquatic herbicide. It has been in use since 1946 and was registered with the U.S. EPA in 1986 and evaluated and reregistered in 2005. It is currently being evaluated for reregistration, and the estimated registration review decision date was in 2017 (EPA 2,4-D Plan 2013). The active ingredient is 2,4-dichloro-phenoxyacetic acid. There are two types of 2,4-D used as aquatic herbicides: dimethyl amine salt (DMA) and butoxyethyl ester (BEE). The ester formulations are toxic to fish and some important invertebrates such as water fleas (*Daphnia* spp.) and midges at application rates. 2,4-D is commercially sold as a liquid amine as well as ester and amine granular products for control of submerged, emergent, and floating-leaf vegetation. Only 2,4-D products labeled for use in aquatic environments may be used to control aquatic plants.

Mode of Action and Degradation

Although the exact mode of action of 2,4-D is not fully understood, the herbicide is traditionally believed to target broad-leaf dicotyledon species with minimal effects generally observed on numerous monocotyledon species, especially in terrestrial applications (WSSA 2007). 2,4-D is a systemic herbicide which affects plant cell growth and division. Upon application, it mimics the natural plant hormone auxin, resulting in bending and twisting of stems and petioles followed by growth inhibition, chlorosis (reduced coloration) at growing points, and necrosis or death of sensitive species (WSSA 2007). Following treatment, 2,4-D is taken up by the plant and translocated through the roots, stems and leaves, and plants begin to die within one to two weeks after application, but can take several weeks to decompose. The total length of target plant roots can be an important in determining the response of an aquatic plant to 2,4-D (Belgers et al. 2007). Treatments should be made when plants are growing. After treatment, the 2,4-D concentration in the water is reduced primarily through microbial activity, off-site movement by water, or adsorption to small particles in silty water.

Previous studies have indicated that 2,4-D degradation in water is highly variable depending on numerous factors such as microbial presence, temperature, nutrients, light, oxygen, organic content of substrate, pH, and whether or not the water has been previously exposed to 2,4-D or other phenoxyacetic acids (Howard et al. 1991). Once in contact with water, both the ester and amine formulations dissociate to the acid form of 2,4-D, with a faster dissociation to the acid form under more alkaline conditions. 2,4-D degradation products include 1,2,4-benzenetriol, 2,4-dichlorophenol, 2,4-dichloroanisole, chlorohydroquinone (CHQ), 4-chlorophenol, and volatile organics.

The half-life of 2,4-D has a wide range depending on water conditions. Half-lives have been reported to range from 12.9 to 40 days, while in anaerobic lab conditions the half-life has been measured at 333 days (EPA RED 2,4-D 2005). In large-scale low-concentration 2,4-D treatments monitored across numerous Wisconsin lakes, estimated half-lives ranged from 4-76 days, and the rate of herbicide degradation was generally observed to be slower in oligotrophic seepage lakes. Of these large-scale 2,4-D treatments, the threshold for irrigation of plants which are not labeled for direct treatment with 2,4-D (<0.1 ppm (100 ppb) by 21 DAT) was exceeded the majority of the treatments (Nault et al. 2018). Previous historical use of 2,4-D may also be an important variable to consider, as microbial communities which are responsible for the breakdown of 2,4-D may potentially exhibit changes in community composition over time with repeated use (de Lipthay et al. 2003; Macur et al. 2007). Additional detailed information on the environmental fate of 2,4-D is compiled by Walters 1999.

There have been some preliminary investigations into the concentration of primarily granular 2,4-D in water-saturated sediments, or pore-water. Initial results suggest the concentration of 2,4-D in the pore-water varies widely from site to site following a chemical treatment, although in some locations the concentration in the pore-water was observed to be 2-3 times greater than the application rate (Jim Kreitlow [DNR], *personal communication*). Further research and additional studies are needed to assess the implications of this finding for target species control and non-target impacts on a variety of organisms.

<u>Toxicology</u>

There are no restrictions on eating fish from treated waterbodies, human drinking water, or pet/livestock drinking water. Based upon 2,4-D ester (BEE) product labels, there is a 24-hour waiting period after treatment for swimming. Before treated water can be used for irrigation, the concentration must be below 0.1 ppm (100 ppb), or at least 21 days must pass. Adverse health effects can be produced by acute and chronic exposure to 2,4-D. Those who mix or apply 2,4-D need to protect their skin and eyes from contact with 2,4-D products to minimize irritation and avoid inhaling the spray. In its consideration of exposure risks, the EPA believes no significant risks will occur to recreational users of water treated with 2,4-D.

There are differences in toxicity of 2,4-D depending on whether the formulation is an amine (DMA) or ester (BEE), with the BEE formulation shown to be more toxic in aquatic environments. BEE formulations are considered toxic to fish and invertebrates such as water fleas and midges at operational application rates. DMA formulations are not considered toxic to fish or invertebrates at operational application rates. Available data indicate 2,4-D does not accumulate at significant levels in the tissues of fish. Although fish exposed to 2,4-D may take up very small amounts of its breakdown products to then be metabolized, the vast majority of these products are rapidly excreted in urine (Ghassemi et al. 1981).

On an acute basis, EPA assessment considers 2,4-D to be "practically non-toxic" to honeybees and tadpoles. Dietary tests (substance administered in the diet for five consecutive days) have shown 2,4-D to be "practically non-toxic" to birds, with some species being more sensitive than others (when 2,4-D was orally and directly administered to birds by capsule or gavage, the substance was "moderately toxic" to some species). For freshwater invertebrates, EPA considers 2,4-D amine to be "practically non-toxic" to "slightly toxic" (EPA RED 2,4-D 2005). Field studies on the potential impact of 2,4-D on benthic macroinvertebrate communities have generally not observed significant changes, although at least one study conducted in Wisconsin observed negative correlations in macroinvertebrate richness and abundance following treatment, and further studies are likely warranted (Stephenson and Mackie 1986; Siemering et al. 2008; Harrahy et al. 2014). Additionally, sublethal effects such as mouthpart deformities and change in sex ratio have been observed in the midge *Chironomus riparius* (Park et al. 2010).

While there is some published literature available looking at short-term acute exposure of various aquatic organisms to 2,4-D, there is limited literature is available on the effects of low-concentration chronic exposure to commercially available 2,4-D formulations (EPA RED 2,4-D 2005). The department recently funded several projects related to increasing our understanding of the potential impacts of chronic exposure to low-concentrations of 2,4-D through AIS research and development grants. One of these studies observed that fathead minnows (*Pimephales promelas*) exposed under laboratory conditions for 28 days to 0.05 ppm (50 ppb) of two different commercial formulations of 2,4-D (DMA® 4 IVM and Weedestroy® AM40) had decreases in larval survival and tubercle presence in males, suggesting that these formulations may exert some degree of chronic toxicity or endocrine-disruption which has not been previously observed when testing pure compound 2,4-D (DeQuattro and Karasov 2016). However, another follow-up study determined that fathead minnow larval survival (30 days post hatch) was decreased following exposure of eggs and larvae to pure 2,4-D, as well as to the two commercial formulations (DMA® 4 IVM and Weedestroy® AM40), and also identified a critical window of exposure for effects on survival to the period between fertilization and 14 days post hatch (Dehnert et al. 2018).

Another related follow-up laboratory study is currently being conducted to examine the effects of 2,4-D exposure on embryos and larvae of several Wisconsin native fish species. Preliminary results indicate that negative impacts of embryo survival were observed for 4 of the 9 native species tested (e.g., walleye, northern pike, white crappie, and largemouth bass), and negative impacts of larval survival were observed for 4 of 7 natives species tested (e.g., walleye, yellow perch, fathead minnows, and white suckers; Dehnert and Karasov, *in progress*).

A controlled field study was conducted on six northern Wisconsin lakes to understand the potential impacts of early season large-scale, low-dose 2,4-D on fish and zooplankton (Rydell et al. 2018). Three lakes were treated with early season low-dose liquid 2,4-D (lakewide epilimnetic target rate: 0.3 ppm (300 ppb)), while the other three lakes served as reference without treatment. Zooplankton densities were similar within lakes during the pre-treatment year and year of treatment, but different trends in several zooplankton species were observed in treatment lakes during the year following treatment. Peak abundance of larval yellow perch (*Perca flavescens*) was lower in the year following treatment, and while this finding was not statistically significant, decreased larval yellow perch abundance was not observed in reference lakes. The observed declines in larval yellow perch abundance and changes in zooplankton trends within treatment lakes in the year after treatment may be a result of changes in aquatic plant communities and not a direct effect of treatment. No significant effect was observed on peak abundance of larval largemouth bass (*Micropterus salmoides*), minnows, black crappie (*Pomoxis nigromaculatus*), bluegill (*Lepomis macrochirus*), or juvenile yellow perch. Larval black crappie showed no detectable response in growth or feeding success. Net pen trials for juvenile bluegill indicated no significant difference in survival between treatment and reference trials, indicating that no direct mortality was associated with the herbicide treatments. Detection of the level of larval fish mortality found in the lab studies would not have been possible in the field study given large variability in larval fish abundance among lakes and over time.

Concerns have been raised about exposure to 2,4-D and elevated cancer risk. Some epidemiological studies have found associations between 2,4-D and increased risk of non-Hodgkin lymphoma in high exposure populations, while other studies have shown that increased cancer risk may be caused by other factors (Hoar et al. 1986; Hardell and Eriksson 1999; Goodman et al. 2015). The EPA determined in 2005 that there is not sufficient evidence to classify 2,4-D as a human carcinogen (EPA RED 2,4-D 2005).

Another chronic health concern with 2,4-D is the potential for endocrine disruption. There is some evidence that 2,4-D may have effects on reproductive development, though other studies suggest the findings may have had other causes (Garry et al. 1996; Coady et al. 2013; Goldner et al. 2013; Neal et al. 2017). The extent and implications of this are not clear and it is an area of ongoing research.

Detailed literature reviews of 2,4-D toxicology have been compiled by Garabrant and Philbert (2002), Jervais et al. (2008), and Burns and Swaen (2012).

<u>Species Susceptibility</u>

With appropriate concentration and exposure, 2,4-D is capable of reducing abundance of the invasive plant species Eurasian watermilfoil (*Myriophyllum spicatum*), parrot feather (*M. aquaticum*), water chestnut (*Trapa natans*), water hyacinth (*Eichhornia crassipes*), and water lettuce (*Pistia stratiotes*; Elliston and Steward 1972; Westerdahl et al. 1983; Green and Westerdahl 1990; Helsel et al. 1996, Poovey and Getsinger 2007; Wersal et al. 2010b; Cason and Roost 2011; Robles et al. 2011; Mudge and Netherland 2014). Perennial pepperweed (*Lepidium latifolium)* and fanwort (*Cabomba caroliniana*) have been shown to be somewhat tolerant of 2,4-D (Bultemeier et al. 2009; Whitcraft and Grewell 2012).

Efficacy and selectivity of 2,4-D is a function of concentration and exposure time (CET) relationships, and rates of 0.5-2.0 ppm coupled with exposure times ranging from 12 to 72 hours have been effective at achieving Eurasian watermilfoil control under laboratory settings (Green and Westerdahl 1990). In addition, long exposure times (>14 days) to low-concentrations of 2,4-

D (0.1-0.25 ppm) have also been documented to achieve milfoil control (Hall et al. 1982; Glomski and Netherland 2010).

According to product labels, desirable native species that may be affected include native milfoils (*Myriophyllum* spp.), coontail (*Ceratophyllum demersum*), common waterweed (*Elodea canadensis*), naiads (*Najas* spp.), waterlilies (*Nymphaea* spp. and *Nuphar* spp.), bladderworts (*Utricularia* spp.), and duckweeds (*Lemna* spp.). While it may affect softstem bulrush (*Schoenoplectus tabernaemontani*), other species such as American bulrush (*Schoenoplectus americanus*) and muskgrasses (*Chara* spp.) have been shown to be somewhat tolerant of 2,4-D (Miller and Trout 1985; Glomski et al. 2009; Nault et al. 2014; Nault et al. 2018).

In large-scale, low-dose (0.073-0.5 ppm) 2,4-D treatments evaluated by Nault et al. (2018), milfoil exhibited statistically significant lakewide decreases in posttreatment frequency across 23 of the 28 (82%) of the treatments monitored. In lakes where year of treatment milfoil control was achieved, the longevity of control ranged from 2–8 years. However, it is important to note that milfoil was not 'eradicated' from any of these lakes and is still present even in those lakes which have sustained very low frequencies over time. While good year of treatment control was achieved in all lakes with pure Eurasian watermilfoil populations, significantly reduced control was observed in the majority of lakes with hybrid watermilfoil (*Myriophyllum spicatum* x *sibiricum*) populations. Eurasian watermilfoil control was correlated with the mean concentration of 2,4-D measured during the first two weeks of treatment, with increasing lakewide concentrations resulting in increased Eurasian watermilfoil control. In contrast, there was no significant relationship observed between Eurasian watermilfoil control and mean concentration of 2,4-D. In lakes where good (>60%) year of treatment control of hybrid watermilfoil was achieved, 2,4-D degradation was slow, and measured lakewide concentrations were sustained at >0.1 ppm (>100 ppb) for longer than 31 days. In addition to reduced year of treatment efficacy, the longevity of control was generally shorter in lakes that contained hybrid watermilfoil versus Eurasian watermilfoil, suggesting that hybrid watermilfoil may have the ability to rebound quicker after large-scale treatments than pure Eurasian watermilfoil populations. However, it is important to keep in mind that hybrid watermilfoil is broad term for multiple different strains, and variation in herbicide response and growth between specific genotypes of hybrid watermilfoil has been documented (Taylor et al. 2017).

In addition, the study by Nault et al. (2018) documented several native monocotyledon and dicotyledon species that exhibited significant declines posttreatment. Specifically, northern watermilfoil (*Myriophyllum sibiricum*), slender naiad (*Najas flexilis*), water marigold (*Bidens beckii*), and several thin-leaved pondweeds (*Potamogeton pusillus*, *P. strictifolius*, *P. friesii* and *P. foliosus*) showed highly significant declines in the majority of the lakes monitored. In addition, variable/Illinois pondweed (*P. gramineus*/*P. illinoensis*), flat-stem pondweed (*P. zosteriformis*), fern pondweed (*P. robbinsii*), and sago pondweed (*Stuckenia pectinata*) also declined in many lakes. Ribbon-leaf pondweed (*P. epihydrus*) and water stargrass (*Heteranthera dubia*) declined in the lakes where they were found. Mixed effects of treatment were observed with water celery (*Vallisneria americana*) and southern naiad (*Najas guadalupensis*), with some lakes showing significant declines posttreatment and other lakes showing increases.

Since milfoil hybridity is a relatively new documented phenomenon (Moody and Les 2002), many of the early lab studies examining CET for milfoil control did not determine if they were examining pure Eurasian watermilfoil or hybrid watermilfoil (*M. spicatum* x *sibiricum*) strains. More recent laboratory and mesocosm studies have shown that certain strains of hybrid watermilfoil exhibit more aggressive growth and are less affected by 2,4-D (Glomski and Netherland 2010; LaRue et al. 2013; Netherland and Willey 2017; Taylor et al. 2017), while other studies have not seen differences in overall growth patterns or treatment efficacy when compared to pure Eurasian watermilfoil (Poovey et al. 2007). Differences between Eurasian and hybrid watermilfoil control following 2,4-D applications have also been documented in the field, with lower efficacy and shorter longevity of hybrid watermilfoil control when compared to pure Eurasian watermilfoil populations (Nault et al. 2018). Field studies conducted in the Menominee River Drainage in northeastern Wisconsin and upper peninsula of Michigan observed hybrid milfoil genotypes more frequently in lakes that had previous 2,4-D treatments, suggesting possible selection of more tolerant hybrid strains over time (LaRue 2012).

Fluridone

Registration and Formulations

Fluridone is an aquatic herbicide that was initially registered with the U.S. EPA in 1986. It is currently being evaluated for reregistration. The estimated registration review decision date was in 2014 (EPA Fluridone Plan 2010). The active ingredient is (1-methyl-3-phenyl-5-[3-(trifluoromethyl) phenyl]-4(1H)-pyridinone). Fluridone is available in both liquid and slow-release granular formulations.

Mode of Action and Degradation

Fluridone's mode of action is to reduce a plant's ability to protect itself from sun damage. The herbicide prevents the plant from making a protective pigment and as a result, sunlight causes the plant's chlorophyll to break down. Treated plants will turn white or pink at the growing tips a week after exposure and will begin to die one to two months after treatment (Madsen et al. 2002). Therefore, fluridone is only effective if plants are actively growing at the time of treatment. Effective use of fluridone requires low, sustained concentrations and a relatively long contact time (e.g., 45-90 days). Due to this requirement, fluridone is usually applied to an entire waterbody or basin. Some success has been demonstrated when additional follow-up 'bump' treatments are used to maintain the low concentrations over a long enough period of time to produce control. Fluridone has also been applied to riverine systems using a drip system to maintain adequate CET.

Following treatment, the amount of fluridone in the water is reduced through dilution and water movement, uptake by plants, adsorption to the sediments, and via breakdown caused by light and microbes. Fluridone is primarily degraded through photolysis (Saunders and Mosier 1983), while depth, water clarity and light penetration can influence degradation rates (Mossler et al. 1989; West et al. 1983). There are two major degradation products from fluridone: n-methyl formamide (NMF) and 3-trifluoromethyl benzoic acid.

The half-life of fluridone can be as short as several hours, or hundreds of days, depending on conditions (West et al. 1979; West et al. 1983; Langeland and Warner 1986; Fox et al. 1991, 1996; Jacob et al. 2016). Preliminary work on a seepage lake in Waushara County, WI detected fluridone in the water nearly 400 days following an initial application that was then augmented to maintain concentrations via a 'bump' treatment at 60 and 100 days later (Onterra 2017a). Light exposure is influential in controlling degradation rate, with a half-life ranging from 15 to 36 hours when exposed to the full spectrum of natural sunlight (Mossler et al. 1989). As light wavelength increases, the half-life increases too, indicating that season and timing may affect fluridone persistence. Fluridone half-life has been shown to be only slightly dependent on fluridone concentration, oxygen concentration, and pH (Saunders and Mosier 1983). One study found that the half-life of fluridone in water was slightly lower when the herbicide was applied to the surface of the water as opposed to a sub-surface application, suggesting that degradation may also be affected by mode of application (West and Parka 1981).

The persistence of herbicide in the sediment has been reported to be much longer than in the overlying water column, with studies showing persistence ranges from 3 months to a year in sediments (Muir et al. 1980; Muir and Grift 1982; West et al. 1983). Persistence in soil is influenced by soil chemistry (Shea and Weber 1983; Mossler et al. 1993). Fluridone concentrations measured in sediments reach a maximum in one to four weeks after treatment and decline in four months to a year depending on environmental conditions. Fluridone adsorbs to clay and soils with high organic matter, especially in pellet form, and can reduce the concentration of fluridone in the water. Adsorption to the sediments is reversible; fluridone gradually dissipates back into the water where it is subject to chemical breakdown.
Some studies have shown variable release time of the herbicide among different granular fluridone products (Mossler et al. 1993; Koschnick et al. 2003; Bultemeier and Haller 2015). In addition, pelletized formulations may be more effective in sandy hydrosoils, while aqueous suspension formulations may be more appropriate for areas with high amounts of clay or organic matter (Mossler et al. 1993)

Toxicology

Fluridone does not appear to have short-term or long-term effects on fish at approved application rates, but fish exposed to water treated with fluridone do absorb fluridone into their tissues. However, fluridone has demonstrated a very low potential for bioconcentration in fish, zooplankton, and aquatic plants (McCowen et al. 1979; West et al. 1979; Muir et al. 1980; Paul et al. 1994). Fluridone concentrations in fish decrease as the herbicide disappears from the water. Studies on the effects of fluridone on aquatic invertebrates (e.g., midge and water flea) have shown increased mortality at label application rates (Hamelink et al. 1986; Yi et al. 2011). Studies on birds indicate that fluridone would not pose an acute or chronic risk to birds. In addition, no treatment related effects were noted in mice, rats, and dogs exposed to dietary doses. No studies have been published on amphibians or reptiles. There are no restrictions on swimming, eating fish from treated waterbodies, human drinking water or pet/livestock drinking water. Depending on the type of waterbody treated and the type of plant being watered, irrigation restrictions may apply for up to 30 days. There is some evidence that the fluridone degradation product NMF causes birth defects, though NMF has only been detected in the lab and not following actual fluridone treatments in the field, including those at maximum label rate (Osborne et al. 1989; West et al. 1990).

Species Susceptibility

Because fluridone treatments are often applied at a lakewide scale and many plant species are susceptible to fluridone, careful consideration should be given to potential non-target impacts and changes in water quality in response to treatment. Sustained native plant species declines and reductions in water clarity have been observed following fluridone treatments in field applications (O'Dell et al. 1995; Valley et al. 2006; Wagner et al. 2007; Parsons et al. 2009). However, reductions in water clarity are not always observed and can be avoided (Crowell et al. 2006). Additionally, the selective activity of fluridone is primarily rate-dependent based on analysis of pigments in nine aquatic plant species (Sprecher et al. 1998b).

Fluridone is most often used for control of invasive species such as Eurasian and hybrid watermilfoil (*Myriophyllum spicatum* x *sibiricum*), Brazilian waterweed (*Egeria densa*), and hydrilla (*Hydrilla verticillata*; Schmitz et al. 1987; MacDonald et al. 1993; Netherland et al. 1993; Netherland and Getsinger 1995a, 1995b; Cockreham and Netherland 2000; Hofstra and Clayton 2001; Madsen et al. 2002; Netherland 2015). However, fluridone tolerance has been observed in some hydrilla and hybrid watermilfoil populations (Michel et al. 2004; Arias et al. 2005; Puri et al. 2006; Slade et al. 2007; Berger et al. 2012, 2015; Thum et al. 2012; Benoit and Les 2013; Netherland and Jones 2015). Fluridone has also been shown to affect flowering rush (*Butomus umbellatus*), fanwort (*Cabomba caroliniana*), buttercups (*Ranunculus* spp.), long-leaf pondweed (*Potamogeton nodosus*), Illinois pondweed (*P. illinoensis*), leafy pondweed (*P. foliosus*), flat-stem pondweed (*P. zosteriformis*), sago pondweed (*Stuckenia pectinata*), oxygen-weed (*Lagarosiphon major*), northern watermilfoil (*Myriophyllum sibiricum*), variable-leaf watermilfoil (*M. heterophyllum*), curly-leaf pondweed (*Potamogeton crispus*), coontail (*Ceratophyllum demersum*), common waterweed (*Elodea canadensis*), southern naiad (*Najas guadalupensis*), slender naiad (*N. flexilis*), white waterlily (*Nymphaea odorata*), water marigold (*Bidens beckii*), duckweed (*Lemna* spp.), and watermeal (*Wolffia columbiana*) (Wells et al. 1986; Kay 1991; Farone and McNabb 1993; Netherland et al. 1997; Koschnick et al. 2003; Crowell et al. 2006; Wagner et al. 2007; Parsons et al. 2009; Cheshier et al. 2011; Madsen et al. 2016). Muskgrasses (*Chara* spp.), water celery (*Vallisneria americana*), cattails (*Typha* spp.), and willows (*Salix* spp.) have been shown to be somewhat tolerant of fluridone (Farone and McNabb 1993; Poovey et al. 2004; Crowell et al. 2006).

Large-scale fluridone treatments that targeted Eurasian and hybrid watermilfoils have been conducted in several Wisconsin lakes. Recently, five of these waterbodies treated with low-dose fluridone (2-4 ppb) have been tracked over time to understand herbicide dissipation and degradation patterns, as well as the efficacy, selectivity, and longevity of these treatments. These field trials resulted in a pre- vs. post-treatment decrease in the number of vegetated littoral zone sampling sites, with a 9-26% decrease observed following treatment (an average decrease in vegetated littoral zone sites of 17.4% across waterbodies). In four of the five waterbodies, substantial decreases in plant biomass (≥10% reductions in average total rake fullness) was documented at sites where plants occurred in both the year of and year after treatment. Good milfoil control was achieved, and long-term monitoring is ongoing to understand the longevity of target species control over time. However, non-target native plant populations were also observed to be negatively impacted in conjunction with these treatments, and long-term monitoring is ongoing to understand their recovery over time. Exposure times in the five waterbodies monitored were found to range from 320 to 539 days before falling below detectable limits. Data from these recent projects is currently being compiled and a compressive analysis and report is anticipated in the near future.

Endothall

Registration and Formulations

Endothall was registered with the U.S. EPA for aquatic use in 1960 and reregistered in 2005 (Menninger 2012). Endothall is the common name of the active ingredient endothal acid (7-oxabicyclo[2,2,1] heptane-2,3-dicarboxylic acid). Granular and liquid formulations are currently registered by EPA and DATCP. Endothall products are used to control a wide range of terrestrial and aquatic plants. Two types of endothall are available: dipotassium salt and dimethylalkylamine salt ("mono-N,N-dimethylalkylamine salt" or "monoamine salt"). The dimethylalkylamine salt form is toxic to fish and other aquatic organisms and is faster-acting than the dipotassium salt form.

Mode of Action and Degradation

Endothall is considered a contact herbicide that inhibits respiration, prevents the production of proteins and lipids, and disrupts the cellular membrane in plants (MacDonald et al. 1993; MacDonald et al. 2001; EPA RED Endothall 2005; Bajsa et al. 2012). Although typical rates of endothall application inhibit plant respiration, higher concentrations have been shown to increase respiration (MacDonald et al. 2001). The mode of action of endothall is unlike any other commercial herbicide. For effective control, endothall should be applied when plants are actively growing, and plants begin to weaken and die within a few days after application.

Uptake of endothall is increased at higher water temperatures and higher amounts of light (Haller and Sutton 1973). Netherland et al. (2000) found that while biomass reduction of curly-leaf pondweed (*Potamogeton crispus*) was greater at higher water temperature, reductions of turion production were much greater when curly-leaf pondweed was treated a lower water temperature (18 °C vs 25 °C).

Degradation of endothall is primarily microbial (Sikka and Saxena 1973) and half-life of the dipotassium salt formulations is between 4 to 10 days (Reinert and Rodgers 1987; Reynolds 1992), although dissipation due to water movement may significantly shorten the effective half-life in some treatment scenarios. Half of the active ingredient from granular endothall formulations has been shown to be released within 1-5 hours under conditions that included water movement (Reinert et al. 1985; Bultemeier and Haller 2015). Endothall is highly water soluble and does not readily adsorb to sediments or lipids (Sprecher et al. 2002; Reinert and Rodgers 1984). Degradation from sunlight or hydrolysis is very low (Sprecher et al. 2002). The degradation rate of endothall has been shown to increase with increasing water temperature (UPI, *unpublished data*). The degradation rate is also highly variable across aquatic systems and is much slower under anaerobic conditions (Simsiman and Chesters 1975). Relative to other herbicides, endothall is unique in that is comprised of carbon, hydrogen, and oxygen with the addition of potassium and nitrogen in the dipotassium and dimethylalkylamine formulations, respectively. This allows for complete breakdown of the herbicide without additional intermediate breakdown products (Sprecher et al. 2002).

Toxicology

All endothall products have a drinking water standard of 0.1 ppm and cannot be applied within 600 feet of a potable water intake. Use restrictions for dimethylalkylamine salt formulations have additional irrigation and aquatic life restrictions.

Dipotassium salt formulations

At recommended rates, the dipotassium salt formulations appear to have few short-term behavioral or reproductive effects on bluegill (*Lepomis macrochirus*) or largemouth bass (*Micropterus salmoides;* Serns 1977; Bettolli and Clark 1992; Maceina et al. 2008). Bioaccumulation of dipotassium salt formulations by fish from water treated with the herbicide is unlikely, with studies showing less than 1% of endothall being taken up by bluegill (Sikka et al. 1975; Serns 1977). In addition, studies have shown the dipotassium salt formulation induces no significant adverse effects on aquatic invertebrates when used at label application rates (Serns 1975; Williams et al. 1984). A freshwater mussel species was found to be more sensitive to dipotassium salt endothall than other invertebrate species tested, but significant acute toxicity was still only found at concentrations well above the maximum label rate. However, as with other plant control approaches, some aquatic plant-dwelling populations of aquatic organisms may be adversely affected by application of endothall formulations due to habitat loss.

During EPA reregistration of endothall in 2005, it was required that product labels state that lower rates of endothall should be used when treating large areas, “such as coves where reduced water movement will not result in rapid dilution of the herbicide from the target treatment area or when treating entire lakes or ponds.”

Dimethylalkylamine salt formulations

In contrast to the respective low to slight toxicity of the dipotassium salt formulations to fish and aquatic invertebrates, laboratory studies have shown the dimethylalkylamine formulations are toxic to fish and macroinvertebrates at concentrations above 0.3 ppm. In particular, the liquid formulation will readily kill fish present in a treatment site. Product labels for the dimethylalkylamine salt formulations recommend no treatment where fish are an important resource.

The dimethylalkylamine formulations are more active on aquatic plants than the dipotassium formulations, but also are 2-3 orders of magnitude more toxic to non-target aquatic organisms (EPA RED Endothall 2005; Keckemet 1969). The 2005 reregistration decision document limits aquatic use of the dimethylalkylamine formulations to algae, Indian swampweed (*Hygrophila polysperma*), water celery (*Vallisneria americana*), hydrilla (*Hydrilla verticillata*), fanwort

(*Cabomba caroliniana*), bur reed (*Sparganium* sp.), common waterweed (*Elodea canadensis*), and Brazilian waterweed (*Egeria densa*). Coontail (*Ceratophyllum demersum*), watermilfoils (*Myriophyllum* spp.), naiads (*Najas* spp.), pondweeds (*Potamogeton* spp.), water stargrass (*Heteranthera dubia*), and horned pondweed (*Zannichellia palustris*) were to be removed from product labels (EPA RED Endothall 2005).

<u>Species Susceptibility</u>

According to the herbicide label, the maximum target concentration of endothall is 5000 ppb (5.0 ppm) acid equivalent (ae). Endothall is used to control a wide range of submersed species, including non-native species such as curly-leaf pondweed and Eurasian watermilfoil (*Myriophyllum spicatum*). The effects of the different formulations of endothall on various species of aquatic plants are discussed below.

Dipotassium salt formulations

At least one mesocosm or lab study has shown that endothall (at or below the maximum label rate) will control the invasive species hydrilla (Netherland et al. 1991; Wells and Clayton 1993; Hofstra and Clayton 2001; Pennington et al. 2001; Skogerboe and Getsinger 2001; Shearer and Nelson 2002; Netherland and Haller 2006; Poovey and Getsinger 2010), oxygen-weed (*Lagarosiphon major*; Wells and Clayton 1993; Hofstra and Clayton 2001), Eurasian watermilfoil (Netherland et al. 1991; Skogerboe and Getsinger 2002; Mudge and Theel 2011), water lettuce (*Pistia stratiotes*; Conant et al. 1998), curly-leaf pondweed (Yeo 1970), and giant salvinia (*Salvinia molesta*; Nelson et al. 2001). Wersal and Madsen (2010a) found that parrot feather (*Myriophyllum aquaticum*) control with endothall was less than 40% even with two days of exposure time at the maximum label rate. Endothall was shown to control the shoots of flowering rush (*Butomus umbellatus*), but control of the roots was variable (Poovey et al. 2012; Poovey et al. 2013). One study found that endothall did not significantly affect photosynthesis in fanwort with 6 days of exposure at 2.12 ppm ae (2120 ppb ae; Bultemeier et al. 2009). Large-scale, low-dose endothall treatments were found to reduce curly-leaf pondweed frequency, biomass, and turion production substantially in Minnesota lakes, particularly in the first 2-3 years of treatments (Johnson et al. 2012).

Native species that were significantly impacted (at or below the maximum endothall label rate in at least one mesocosm or lab study) include coontail (Yeo 1970; Hofstra and Clayton 2001; Hofstra et al. 2001; Skogerboe and Getsinger 2002; Wells and Clayton 1993; Mudge 2013), southern naiad (*Najas guadalupensis*; Yeo 1970; Skogerboe and Getsinger 2001), white waterlily (*Nymphaea odorata*; Skogerboe and Getsinger 2001), leafy pondweed (*Potamogeton foliosus*; Yeo 1970), Illinois pondweed (*Potamogeton illinoensis*; Skogerboe and Getsinger 2001; Shearer and Nelson 2002; Skogerboe and Getsinger 2002; Mudge 2013), long-leaf pondweed (*Potamogeton nodosus*; Yeo 1970; Skogerboe and Getsinger 2001; Shearer and Nelson 2002; Mudge 2013), small pondweed (*P. pusillus*; Yeo 1970), broadleaf arrowhead (*Sagittaria latifolia*; Skogerboe and Getsinger 2001), sago pondweed (*Stuckenia pectinata*; Yeo 1970; Sprecher et al. 1998a; Skogerboe and Getsinger 2002; Slade et al. 2008), water celery (*Vallisneria americana*; Skogerboe and Getsinger 2001; Skogerboe and Getsinger 2002; Shearer and Nelson 2002; Mudge 2013), and horned pondweed (Yeo 1970; Gyselinck and Courter 2015).

Species which were not significantly impacted or which recovered quickly include watershield (*Brasenia schreberi*; Skogerboe and Getsinger 2001), muskgrasses (*Chara* spp.; Yeo 1970; Wells and Clayton 1993; Hofstra and Clayton 2001), common waterweed (Yeo 1970; Wells and Clayton 1993; Skogerboe and Getsinger 2002), water stargrass (Skogerboe and Getsinger 2001), water net (*Hydrodictyon reticulatum*; Wells and Clayton 1993), the freshwater macroalgae *Nitella clavata* (Yeo 1970), yellow pond-lily (*Nuphar advena*; Skogerboe and Getsinger 2002), swamp smartweed (*Polygonum hydropiperoides*; Skogerboe and Getsinger 2002), pickerelweed (*Pontederia cordata*; Skogerboe and Getsinger 2001), softstem bulrush (*Schoenoplectus tabernaemontani*; Skogerboe and Getsinger 2001), and broadleaf cattail (*Typha latifolia*; Skogerboe and Getsinger 2002).

Field trials mirror the species susceptibility above and in addition show that endothall also can impact several high-value pondweed species (*Potamogeton* spp.), including large-leaf pondweed (*P. amplifolius*; Parsons et al. 2004), fern pondweed (*P. robbinsii*; Onterra 2015; Onterra 2018), white-stem pondweed (*P. praelongus*; Onterra 2018), small pondweed (Big Chetac Chain Lake Association 2016; Onterra 2018), clasping-leaf pondweed (*P. richardsonii*; Onterra 2018), and flat-stem pondweed (*P. zosteriformis*; Onterra 2017b).

Dimethylalkylamine salt formulations

The dimethylalkylamine formulations are more active on aquatic plants than the dipotassium formulations (EPA RED Endothall 2005; Keckemet 1969). At least one mesocosm study has shown that dimethylalkylamine formulation of endothall (at or below the maximum label rate) will control the invasive species fanwort (Hunt et al. 2015) and the native species common waterweed (Mudge et al. 2015), while others have shown that the dipotassium formulation does not control these species well.

Imazamox

Registration and Formulations

Imazamox is the common name of the active ingredient ammonium salt of imazamox (2-[4,5-dihydro-4-methyl-4-(1-methylethyl)-5-oxo-1H-imidazol-2-yl]-5-(methoxymethl)-3-pyridinecarboxylic acid. It was registered with U.S. EPA in 2008 and is currently under registration review with an estimated registration decision between 2019 and 2020 (EPA Imazamox Plan 2014). In aquatic environments, a liquid formulation is typically applied to submerged vegetation by broadcast spray or underwater hose application and to emergent or floating leaf vegetation by broadcast spray or foliar application. There is also a granular formulation.

Mode of Action and Degradation

Imazamox is a systemic herbicide that moves throughout the plant tissue and prevents plants from producing a necessary enzyme, acetolactate synthase (ALS), which is not found in animals. Susceptible plants will stop growing soon after treatment, but plant death and decomposition will occur over several weeks (Mudge and Netherland 2014). If used as a post-emergence herbicide, imazamox should be applied to plants that are actively growing. Resistance to ALS-inhibiting

herbicides has appeared in weeds at a higher rate than other herbicide types in terrestrial environments (Tranel and Wright 2002).
Dissipation studies in lakes indicate a half-life ranging from 4 to 49 days with an average of 17 days. Herbicide breakdown does not occur readily in deep, poorly-oxygenated water where there is no light. In this part of a lake, imazamox will tend to bind to sediments rather than breaking down, with a half-life of approximately 2 years. Once in soil, leaching to groundwater is believed to be very limited. The breakdown products of imazamox are nicotinic acid and di- and tricarboxylic acids. It has been suggested that photolytic break down of imazamox is faster than other herbicides, reducing exposure times. However, short-term imazamox exposures have also been associated with extended regrowth times relative to other herbicides (Netherland 2011).

Toxicology

Treated water may be used immediately following application for fishing, swimming, cooking, bathing, and watering livestock. If water is to be used as potable water or for irrigation, the tolerance is 0.05 ppm (50 ppb), and a 24-hour irrigation restriction may apply depending on the waterbody. None of the breakdown products are herbicidal nor suggest concerns for aquatic organisms or human health.

Most concerns about adverse effects on human health involve applicator exposure. Concentrated imazamox can cause eye and skin irritation and is harmful if inhaled. Applicators should minimize exposure by wearing long-sleeved shirts and pants, rubber gloves, and shoes and socks.

Honeybees are affected at application rates so drift during application should be minimized. Laboratory tests using rainbow trout (*Oncorhynchus mykiss*), bluegill (*Lepomis macrochirus*), and water fleas (*Daphnia magna*) indicate that imazamox is not toxic to these species at label application rates.

Imazamox is rated “practically non-toxic” to fish and aquatic invertebrates and does not bioaccumulate in fish. Additional studies on birds indicate toxicity only at dosages that exceed approved application rates.

In chronic tests, imazamox was not shown to cause tumors, birth defects or reproductive toxicity in test animals. Most studies show no evidence of mutagenicity. Imazamox is not metabolized and was excreted by mammals tested. Based on its low acute toxicity to mammals, and its rapid disappearance from the water column due to light and microbial degradation and binding to soil, imazamox is not considered to pose a risk to recreational water users.

Species Susceptibility

In Wisconsin, imazamox is used for treating non-native emergent vegetation such as non-native phragmites (*Phragmites australis* subsp. *australis*) and flowering rush (*Butomus umbellatus*). Imazamox may also be used to treat the invasive curly-leaf pondweed (*Potamogeton crispus*). Desirable native species that may be affected could include other pondweed species (long-leaf pondweed (*P. nodosus*), flat-stem pondweed (*P. zosteriformis*), leafy pondweed (*P. foliosus*), Illinois pondweed (*P. illinoensis*), small pondweed (*P. pusillus*), variable-leaf pondweed (*P.*

gramineus), water-thread pondweed (*P. diversifolius*), perfoliate pondweed (*P. perfoliatus*), large-leaf pondweed (*P. amplifolius*), watershield (*Brasenia schreberi*), and some bladderworts (*Utricularia* spp.). Higher rates of imazamox will control Eurasian watermilfoil (*Myriophyllum spicatum*) but would also have greater non-target impacts on native plants. Imazamox can also be used during a drawdown to prevent plant regrowth and on emergent vegetation.

At low concentrations, imazamox can cause growth regulation rather than mortality in some plant species. This has been shown for non-native phragmites and hydrilla (*Hydrilla verticillata*; Netherland 2011; Cheshier et al. 2012; Theel et al. 2012). In the case of hydrilla, some have suggested that this effect could be used to maintain habitat complexity while providing some target species control (Theel et al. 2012). Imazamox can reduce biomass of non-native phragmites though some studies found regrowth to occur, suggesting a combination of imazapyr and glyphosate to be more effective (Cheshier et al. 2012; Knezevic et al. 2013).

Some level of control of imazamox has also been reported for water hyacinth (*Eichhornia crassipes*), parrot feather (*Myriophyllum aquaticum*), Japanese stiltgrass (*Microstegium vimineum*), water lettuce (*Pistia stratiotes*), and southern cattail (*Typha domingensis*; Emerine et al. 2010; de Campos et al. 2012; Rodgers and Black 2012; Hall et al. 2014; Mudge and Netherland 2014). Imazamox was observed to have greater efficacy in controlling floating plants than emergents in a study of six aquatic plant species, including water hyacinth, water lettuce, parrot feather, and giant salvinia (*Salvinia molesta*; Emerine et al. 2010). Non-target effects have been observed for softstem bulrush (*Schoenoplectus tabernaemontani*), pickerelweed (*Pontederia cordata*), and the native pondweeds long-leaf pondweed, Illinois pondweed, and coontail (*Ceratophyllum demersum*; Koschnick et al. 2007; Mudge 2013). Giant salvinia, white waterlily (*Nymphaea odorata*), bog smartweed (*Polygonum setaceum*), giant bulrush (*Schoenoplectus californicus*), water celery (*Vallisneria americana*; though the root biomass of wide-leaf *Vallisneria* may be reduced), and several algal species have been found by multiple studies to be unaffected by imazamox (Netherland et al. 2009; Emerine et al. 2010; Rodgers and Black 2012; Mudge 2013; Mudge and Netherland 2014). Other species are likely to be susceptible, for which the effects of imazamox have not yet been evaluated.

Florpyrauxifen-benzyl

Registration and Formulations

Florpyrauxifen-benzyl is a relatively new herbicide, which was first registered with the U.S. EPA in September 2017. The active ingredient is 4-amino-3-chloro-6-(4-chloro-2-fluoro-3-methoxyphenyl)-5-fluoro-pyridine-2-benzyl ester, also identified as florpyrauxifen-benzyl. Florpyrauxifen-benzyl is used for submerged, floating, and emergent aquatic plant control (e.g., ProcellaCORTM) in slow-moving and quiescent waters, as well as for broad spectrum weed control in rice (*Oryza sativa*) culture systems and other crops (e.g., RinskorTM).

Mode of Action and Degradation

Florpyrauxifen-benzyl is a member of a new class of synthetic auxins, the arylpicolinates, that differ in binding affinity compared to other currently registered synthetic auxins such as 2,4-D and triclopyr (Bell et al. 2015). Florpyrauxifen-benzyl is a systemic herbicide (Heilman et al. 2017).

Laboratory studies and preliminary field dissipation studies indicate that florpyrauxifen-benzyl in water is subject to rapid photolysis (Heilman et al. 2017). In addition, the herbicide can also convert partially via hydrolysis to an acid form at high pH (>9) and higher water temperatures (>25°C), and microbial activity in the water and sediment can also enhance degradation (Heilman et al. 2017). The acid form is noted to have reduced herbicidal activity (Netherland and Richardson 2016; Richardson et al. 2016). Under growth chamber conditions, water samples at 1 DAT found that 44-59% of the applied herbicide had converted to acid form, while sampling at 7 and 14 DAT indicated that all the herbicide had converted to acid form (Netherland and Richardson 2016). The herbicide is short-lived, with half-lives ranging from 4 to 6 days in aerobic aquatic environments, and 2 days in anaerobic aquatic environments (WSDE 2017). Degradation in surface water is accelerated when exposed to sunlight, with a reported photolytic half-life in laboratory testing of 0.07 days (WSDE 2017).

There is some anecdotal evidence that initial water temperature and/or pH may impact the efficacy of florpyrauxifen-benzyl (Beets and Netherland 2018). Florpyrauxifen-benzyl has a high soil adsorption coefficient (KOC) and low volatility, which allows for rapid plant uptake resulting in short exposure time requirements (Heilman et al. 2017). Florpyrauxifen-benzyl degrades quickly (2-15 days) in soil and sediment (Netherland et al. 2016). Few studies have yet been completed for groundwater, but based on known environmental properties, florpyrauxifen-benzyl is not expected to be associated with potential environmental impacts in groundwater (WSDE 2017).

Toxicology

No adverse human health effects were observed in toxicological studies submitted for EPA herbicide registration, regardless of the route of exposure (Heilman et al. 2017). There are no drinking water or recreational use restrictions, including swimming and fishing. There are no restrictions on irrigating turf, and a short waiting period (dependent on application rate) for other non-agricultural irrigation purposes.
Florpyrauxifen-benzyl showed a good environmental profile for use in water, and is "practically non-toxic" to birds, bees, reptiles, amphibians, and mammals (Heilman et al. 2017). No ecotoxicological effects were observed on freshwater mussel or juvenile chinook salmon (Heilman et al. 2017). Florpyrauxifen-benzyl will temporarily bioaccumulate in freshwater organisms but is rapidly depurated and/or metabolized within 1 to 3 days after exposure to high (>150 ppb) concentrations (WSDE 2017).

An LC50 value indicates the concentration of a chemical required to kill 50% of a test population of organisms. LC50 values are commonly used to describe the toxicity of a substance. Label recommendations for milfoils do not exceed 9.65 ppb and the maximum label rate for an acre-foot of water is 48.25 ppb. Acute toxicity results using rainbow trout (*Oncorhynchus mykiss*), fathead minnow (*Pimephales promelas*), and sheepshead minnows (*Cyprinodon variegatus variegatus*) indicated LC50 values of greater than 49 ppb, 41 ppb, and 40 ppb, respectively when exposed to the technical grade active ingredient (WSDE 2017). An LC50 value of greater than 1,900 ppb was reported for common carp (*Cyprinus carpio*) exposed to the ProcellaCOR end-use formulation (WSDE 2017).

Acute toxicity results for the technical grade active ingredient using water flea (*Daphnia magna*) and midge (*Chironomus* sp.) indicated LC50 values of greater than 62 ppb and 60 ppb, respectively (WSDE 2017). Comparable acute ecotoxicity testing performed on *D. magna* using the ProcellaCOR end-use formulation indicated an LC50 value of greater than 8 ppm (80,000 ppb; WSDE 2017).

The ecotoxicological no observed effect concentration (NOEC) for various organisms as reported by Netherland et al. (2016) are: fish (>515 ppb ai), water flea (*Daphnia* spp.; >21440 ppb ai), freshwater mussels (>1023 ppb ai), saltwater mysid (>362 ppb ai), saltwater oyster (>289 ppb ai), and green algae (>480 ppb ai). Additional details on currently available ecotoxicological information is compiled by WSDE (2017).

Species Susceptibility

Florpyrauxifen-benzyl is a labeled for control of invasive watermilfoils (e.g., Eurasian watermilfoil (*Myriophyllum spicatum*), hybrid watermilfoil (*M. spicatum* x *sibiricum*), parrot feather (*M. aquaticum*)), hydrilla (*Hydrilla verticillata*), and other non-native floating plants such as floating hearts (*Nymphoides* spp.), water hyacinth (*Eichhornia crassipes*), and water chestnut (*Trapa natans*; Netherland and Richardson 2016; Richardson et al. 2016). Natives species listed on the product label as susceptible to florpyrauxifen-benzyl include coontail (*Ceratophyllum demersum*; Heilman et al. 2017), watershield (*Brasenia schreberi*), and American lotus (*Nelumbo lutea*). In laboratory settings, pickerelweed (*Pontederia cordata*) vegetation has also been shown to be affected (Beets and Netherland 2018).

Based on available data, florpyrauxifen-benzyl appears to show few impacts to native aquatic plants such as aquatic grasses, bulrush (*Schoenoplectus* spp.), cattail (*Typha* spp.), pondweeds (*Potamogeton* spp.), naiads (*Najas* spp.), and water celery (*Vallisneria americana*; WSDE 2017). Laboratory and mesocosm studies also found water marigold (*Bidens beckii*), white waterlily (*Nymphaea odorata*), common waterweed (*Elodea canadensis*), water stargrass (*Heteranthera dubia*), long-leaf pondweed (*Potamogeton nodosus*), and Illinois pondweed (*P. illinoensis*) to be relatively less sensitive to florpyrauxifen-benzyl than labeled species (Netherland et al. 2016; Netherland and Richardson 2016). Non-native fanwort (*Cabomba caroliniana*) was also found to be tolerant in laboratory study (Richardson et al. 2016).

Since florpyrauxifen-benzyl is a relatively new approved herbicide, detailed information on field applications is very limited. Trials in small waterbodies have shown control of parrot feather (*Myriophyllum aquaticum*), variable-leaf watermilfoil (*M. heterophyllum*), and yellow floating heart (*Nymphoides peltata*; Heilman et al. 2017).

S.3.3.3. Emergent and Wetland Herbicides

Glyphosate

Registration and Formulations

Glyphosate is a commonly used herbicide that is utilized in both aquatic and terrestrial sites. It was first registered for use in 1974. EPA is currently re-evaluating glyphosate and the registration decision was expected in 2014 (EPA Glyphosate Plan 2009). The use of glyphosate-based herbicides in aquatic environments that are not approved for aquatic use is very unsafe and is a violation of federal and state pesticide laws. Different formulations of glyphosate are available, including isopropylamine salt of glyphosate and potassium glyphosate.

Glyphosate is effective only on plants that grow above the water and needs to be applied to plants that are actively growing. It will not be effective on plants that are submerged or have most of their foliage underwater, nor will it control regrowth from seed.

Mode of Action and Degradation

Glyphosate is a systemic herbicide that moves throughout the plant tissue and works by inhibiting an important enzyme needed for multiple plant processes, including growth. Following treatment, plants will gradually wilt, appear yellow, and will die in approximately 2 to 7 days. It may take up to 30 days for these effects to become apparent for woody species.

Application should be avoided when heavy rain is predicted within 6 hours. To avoid drift, application is not recommended when winds exceed 5 mph. In addition, excessive speed or pressure during application may allow spray to drift and must be avoided. Effectiveness of glyphosate treatments may be reduced if applied when plants are growing poorly, such as due to drought stress, disease, or insect damage. A surfactant approved for aquatic sites must be mixed with glyphosate before application.

In water, the concentration of glyphosate is reduced through dispersal by water movement, binding to the sediments, and break-down by microorganisms. The half-life of glyphosate is between 3 and 133 days, depending on water conditions. Glyphosate disperses rapidly in water so dilution occurs quickly, thus moving water will decrease concentration, but not half-life. The primary breakdown product of glyphosate is aminomethylphosphonic acid (AMPA), which is also degraded by microbes in water and soil.

Toxicology

Most aquatic forms of glyphosate have no restrictions on swimming or eating fish from treated waterbodies. However, potable water intakes within ½ mile of application must be turned off for 48 hours after treatment. Different formulations and products containing glyphosate may vary in post-treatment water use restrictions.

Most glyphosate-related health concerns for humans involve applicator exposure, exposure through drift, and the surfactant exposure. Some adverse effects from direct contact with the herbicide include temporary symptoms of dermatitis, eye ailments, headaches, dizziness, and nausea. Protective clothing (goggles, a face shield, chemical resistant gloves, aprons, and footwear) should be worn by applicators to reduce exposure. Recently it has been demonstrated that terrestrial formulations of glyphosate can have toxic effects to human embryonic cells and linked to endocrine disruption (Benachour et al. 2007; Gasnier et al. 2009).

Laboratory testing indicates that glyphosate is toxic to carp (*Cyprinus* spp.), bluegills (*Lepomis macrochirus*), rainbow trout (*Oncorhynchus mykiss*), and water fleas (*Daphnia* spp.) only at dosages well above the label application rates. Similarly, it is rated “practically non-toxic” to other aquatic species tested. Studies by other researchers examining the effects of glyphosate on important food chain organisms such as midge larvae, mayfly nymphs, and scuds have demonstrated a wide margin of safety between application rates.

EPA data suggest that toxicological effects of the AMPA compound are similar to that of glyphosate itself. Glyphosate also contains a nitrosamine (n-nitroso-glyphosate) as a contaminant at levels of 0.1 ppm or less. Tests to determine the potential health risks of nitrosamines are not required by the EPA unless the level exceeds 1.0 ppm.

Species Susceptibility

Glyphosate is only effective on actively growing plants that grow above the water’s surface. It can be used to control reed canary grass (*Phalaris arundinacea*), cattails (*Typha* spp.; Linz et al. 1992; Messersmith et al. 1992), purple loosestrife (*Lythrum salicaria*), phragmites (*Phragmites australis* subsp. *australis*; Back and Holomuzki 2008; True et al. 2010; Back et al. 2012; Cheshier et al. 2012), water hyacinth (*Eichhornia crassipes*; Lopez 1993; Jadhav et al. 2008), water lettuce (*Pistia stratiotes*; Mudge and Netherland 2014), water chestnut (*Trapa natans*; Rector et al. 2015), Japanese stiltgrass (*Microstegium vimineum*; Hall et al. 2014), giant reed (*Arundo donax*; Spencer 2014), and perennial pepperweed (*Lepidium latifolium*; Boyer and Burdick 2010). Glyphosate will also reduce abundance of white waterlily (*Nymphaea odorata*) and pond-lilies (*Nuphar* spp.; Riemer and Welker 1974). Purple loosestrife biocontrol beetle (*Galerucella calmariensis*) oviposition and survival have been shown not to be affected by integrated management with glyphosate. Studies have found pickerelweed (*Pontederia cordata*) and floating marsh pennywort (*Hydrocotyle ranunculoides*) to be somewhat tolerant to glyphosate (Newman and Dawson 1999; Gettys and Sutton 2004).

Imazapyr

Registration and Formulations

Imazapyr was registered with the U.S. EPA for aquatic use in 2003 and is currently under registration review. It was estimated to have a registration review decision in 2017 (EPA Imazapyr Plan 2014). The active ingredient is isopropylamine salt of imazapyr (2-[4,5-dihydro-4-methyl-4-(1-methylethyl)-5-oxo-1H-imidazol-2-yl]-3-pyridinecarboxylic acid). Imazapyr is used for control of emergent and floating-leaf vegetation. It is not recommended for control of submersed vegetation.

Mode of Action and Degradation

Imazapyr is a systemic herbicide that moves throughout the plant tissue and prevents plants from producing a necessary enzyme, acetolactate synthase (ALS), which is not found in animals. Susceptible plants will stop growing soon after treatment and become reddish at the tips of the plant. Plant death and decomposition will occur gradually over several weeks to months. Imazapyr should be applied to plants that are actively growing. If applied to mature plants, a higher concentration of herbicide and a longer contact time will be required.

Imazapyr is broken down in the water by light and has a half-life ranging from three to five days. Three degradation products are created as imazapyr breaks down: pyridine hydroxy-dicarboxylic acid, pyridine dicarboxylic acid (quinolinic acid), and nicotinic acid. These degradates persist in water for approximately the same amount of time as imazapyr (half-lives of three to eight days). In soils imazapyr is broken down by microbes, rather than light, and persists with a half-life of one to five months (Boyer and Burdick 2010). Imazapyr doesn't bind to sediments, so leaching through soil into groundwater is likely.

Toxicology

There are no restrictions on recreational use of treated water, including swimming and eating fish from treated waterbodies. If application occurs within a ½ mile of a drinking water intake, then the intake must be shut off for 48 hours following treatment. There is a 120-day irrigation restriction for treated water, but irrigation can begin sooner if the concentration falls below 0.001 ppm (1 ppb). Imazapyr degradates are no more toxic than imazapyr itself and are excreted faster than imazapyr when ingested.

Concentrated imazapyr has low acute toxicity on the skin or if ingested but is harmful if inhaled and may cause irreversible damage if it gets in the eyes. Applicators should wear chemical-resistant gloves while handling, and persons not involved in application should avoid the treatment area during treatment. Chronic toxicity tests for imazapyr indicate that it is not carcinogenic, mutagenic, or neurotoxic. It also does not cause reproductive or developmental toxicity and is not a suspected endocrine disrupter.

Imazapyr is "practically non-toxic" to fish, invertebrates, birds and mammals. Studies have also shown imazapyr to be "practically non-toxic" to "slightly toxic" to tadpoles and juvenile frogs (Trumbo and Waligora 2009; Yahnke et al. 2013). Toxicity tests have not been published on reptiles. Imazapyr does not bioaccumulate in animal tissues.

Species Susceptibility

The imazapyr herbicide label is listed to control the invasive plants phragmites (*Phragmites australis* subsp. *australis*), purple loosestrife (*Lythrum salicaria*), reed canary grass (*Phalaris arundinacea*), non-native cattails (*Typha* spp.) and Japanese knotweed (*Fallopia japonica*) in

Wisconsin. Native species that are also controlled include cattails (*Typha* spp.), waterlilies (*Nymphaea* sp.), pickerelweed (*Pontederia cordata*), duckweeds (*Lemna* spp.), and arrowhead (*Sagittaria* spp.).

Studies have shown imazapyr to effectively control giant reed (*Arundo donax*), water hyacinth (*Eichhornia crassipes*), manyflower marsh-pennywort (*Hydrocotyle umbellata*); yellow iris (*Iris pseudacorus*), water lettuce (*Pistia stratiotes*), perennial pepperweed (*Lepidium latifolium*), Japanese stiltgrass (*Microstegium vimineum*), parrot feather (*Myriophyllum aquaticum*), and cattails (Boyer and Burdick 2010; True et al. 2010; Back et al. 2012; Cheshier et al. 2012; Whitcraft and Grewell 2012; Hall et al. 2014; Spencer 2014; Cruz et al. 2015; DiTomaso and Kyser 2016). Giant salvinia (*Salvinia molesta*) was found to be imazapyr-tolerant (Nelson et al. 2001).

S.3.3.4. Herbicides Used for Submersed and Emergent Plants

Triclopyr

Registration and Formulations

Triclopyr was initially registered with the U.S. EPA in 1979, reregistered in 1997, and is currently under review with an estimated registration review decision in 2019 (EPA Triclopyr Plan 2014). There are two forms of triclopyr used commercially as herbicides: the triethylamine salt (TEA) and the butoxyethyl ester (BEE). BEE formulations are considered highly toxic to aquatic organisms, with observed lethal effects on fish (Kreutzweiser et al. 1994) as well as avoidance behavior and growth impairment in amphibians (Wojtaszek et al. 2005). The active ingredient triethylamine salt (3,5,6-trichloro-2-pyridinyloxyacetic acid) is the formulation registered for use in aquatic systems. It is sold both in liquid and granular forms for control of submerged, emergent, and floating-leaf vegetation. There is also a liquid premixed formulation that contains triclopyr and 2,4-D, which when combined together are reported to have synergistic impacts. Only triclopyr products labeled for use in aquatic environments may be used to control aquatic plants.

Mode of Action and Degradation

Triclopyr is a systemic plant growth regulator that is believed to selectively act on broadleaf (dicot) and woody plants. Following treatment, triclopyr is taken up through the roots, stems and leaf tissues, plant growth becomes abnormal and twisted, and plants die within one to two weeks after application (Getsinger et al. 2000). Triclopyr is somewhat persistent and can move through soil, although only mobile enough to permeate top soil layers and likely not mobile enough to potentially contaminate groundwater (Lee et al. 1986; Morris et al. 1987; Stephenson et al. 1990).

Triclopyr is broken down rapidly by light (photolysis) and microbes, while hydrolysis is not a significant route of degradation. Triclopyr photodegrades and is further metabolized to carbon dioxide, water, and various organic acids by aquatic organisms (McCall and Gavit 1986). It has been hypothesized that the major mechanism for the removal of triclopyr from the aquatic environment is microbial degradation, though the role of photolysis likely remains important in near-surface and shallow waters (Petty et al. 2001). Degradation of triclopyr by microbial action is slowed in the absence of light (Petty et al. 2003). Triclopyr is very slowly degraded under anaerobic conditions, with a reported half-life (the time it takes for half of the active ingredient to degrade) of about 3.5 years (Laskowski and Bidlack 1984). Another study of triclopyr under aerobic aquatic conditions yielded a half-life of 4.7 months (Woodburn and Cranor 1987). The initial breakdown products of triclopyr are TCP (3,5,6-trichloro-2-pyridinol) and TMP (3,5,6-trichloro-2-methoxypridine).

Several studies reported triclopyr half-lives between 0.5-7.5 days (Woodburn et al. 1993; Getsinger et al. 2000; Petty et al. 2001; Petty et al. 2003). Two large-scale, low-dose treatments were reported to have longer triclopyr half-lives from 3.7-12.1 days (Netherland and Jones 2015). Triclopyr half-lives have been shown to range from 3.4 days in plants, 2.8-5.8 days in sediment, up to 11 days in fish tissue, and 11.5 days in crayfish (Woodburn et al. 1993; Getsinger et al. 2000; Petty et al. 2003). TMP and TCP may have longer half-lives than triclopyr, with higher levels in bottom-feeding fish and the inedible parts of fish (Getsinger et al. 2000).

Toxicology

Based upon the triclopyr herbicide label, there are no restrictions on swimming, eating fish from treated waterbodies, or pet/livestock drinking water use. Before treated water can be used for irrigation, the concentration must be below 0.001 ppm (1 ppb), or at least 120 days must pass. Treated water should not be used for drinking water until concentrations of triclopyr are less than 0.4 ppm (400 ppb). There is a least one case of direct human ingestion of triclopyr TEA which resulted in metabolic acidosis and coma with cardiovascular impairment (Kyong et al. 2010).

There are substantial differences in toxicity of BEE and TEA, with the BEE shown to be more toxic in aquatic settings. BEE formulations are considered highly toxic to aquatic organisms, with observed lethal effects on fish (Kreutzweiser et al. 1994) as well as avoidance behavior and growth impairment in amphibians (Wojtaszek et al. 2005). Triclopyr TEA is "practically non-toxic" to freshwater fish and invertebrates (Mayes et al. 1984; Gersich et al. 1984). It ranges from "practically non-toxic" to "slightly toxic" to birds (EPA Triclopyr RED 1998). TCP and TMP appear to be slightly more toxic to aquatic organisms than triclopyr; however, the peak concentration of these degradates is low following treatment and depurates from organisms readily, so that they are not believed to pose a concern to aquatic organisms.

Species susceptibility

Triclopyr has been used to control Eurasian watermilfoil (*Myriophyllum spicatum*) and hybrid watermilfoil (*M. spicatum* x *sibiricum*) at both small- and large-scales (Netherland and Getsinger 1992; Getsinger et al. 1997; Poovey et al. 2004; Poovey et al. 2007; Nelson and Shearer 2008; Heilman et al. 2009; Glomski and Netherland 2010; Netherland and Glomski 2014; Netherland and Jones 2015). Getsinger et al. (2000) found that peak triclopyr accumulation was higher in Eurasian watermilfoil than flat-stem pondweed (*Potamogeton zosteriformis*), indicating triclopyr's affinity for Eurasian watermilfoil as a target species.

According to product labels, triclopyr is capable of controlling or affecting many emergent woody plant species, purple loosestrife (*Lythrum salicaria*), phragmites (*Phragmites australis* subsp. *australis*), American lotus (*Nelumbo lutea*), milfoils (*Myriophyllum* spp.), and many others.

Triclopyr application has resulted in reduced frequency of occurrence, reduced biomass, or growth regulation for the following species: common waterweed (*Elodea canadensis*), water stargrass (*Heteranthera dubia*), white waterlily (*Nymphaea odorata*), purple loosestrife, Eurasian watermilfoil, parrot feather (*Myriophyllum aquaticum*), variable-leaf watermilfoil (*M. heterophyllum*), watercress (*Nasturtium officinale*), phragmites, flat-stem pondweed (*Potamogeton zosteriformis*), clasping-leaf pondweed (*P. richardsonii*), stiff pondweed (*P. strictifolius*), variable-leaf pondweed (*P. gramineus*), white water crowfoot (*Ranunculus aquatilis*), sago pondweed (*Stuckenia pectinata*), softstem bulrush (*Schoenoplectus tabernaemontani*), hardstem bulrush (*S. acutus*), water chestnut (*Trapa natans*), duckweeds (*Lemna* spp.), and submerged flowering rush (*Butomus umbellatus*; Cowgill et al. 1989; Gabor et al. 1995; Sprecher and Stewart 1995; Getsinger et al. 2003; Poovey et al. 2004; Hofstra et al. 2006; Poovey and Getsinger 2007; Champion et al. 2008; Derr 2008; Glomski and Nelson 2008; Glomski et al. 2009; True et al. 2010; Cheshier et al. 2012; Netherland and Jones 2015; Madsen et al. 2015; Madsen et al. 2016). Wild rice (*Zizania palustris*) biomass and height has been shown to decrease significantly following triclopyr application at 2.5 mg/L. Declines were not significant at lower concentrations (0.75 mg/L), though seedlings were more sensitive than young or mature plants (Madsen et al. 2008). American bulrush (*Schoenoplectus americanus*), spatterdock (*Nuphar variegata*), fern pondweed (*Potamogeton robbinsii*), large-leaf pondweed (*P. amplifolius*), leafy pondweed (*P. foliosus*), white-stem pondweed (*P. praelongus*), long-leaf pondweed (*P. nodosus*), Illinois pondweed (*P. illinoensis*), and water celery (*Vallisneria americana*) can be somewhat tolerant of triclopyr applications depending on waterbody characteristics and application rates (Sprecher and Stewart 1995; Glomski et al. 2009; Wersal et al. 2010b; Netherland and Glomski 2014).

Netherland and Jones (2015) evaluated the impact of large-scale, low-dose (~0.1-0.3 ppm) granular triclopyr) applications for control of non-native watermilfoil on several bays of Lake Minnetonka, Minnesota. Near complete loss of milfoil in the treated bays was observed the year of treatment, with increased milfoil frequency reported the following season. However, despite the observed increase in frequency, milfoil biomass remained a minor component of bay-wide biomass (<2%). The number of points with native plants, mean native species per point, and native species richness in the bays were not reduced following treatment. However, reductions in frequency were seen amongst individual species, including northern watermilfoil (*Myriophyllum sibiricum*), water stargrass, common waterweed, and flat-stem pondweed.

Penoxsulam

Registration and Formulations

Penoxsulam (2-(2,2-difluoroethoxy)--6-(trifluoromethyl-N-(5,8-dimethoxy[1,2,4] triazolo[1,5-c]pyrimidin-2-yl))benzenesulfonamide), also referred to as DE-638, XDE-638, XR-638 is a post-emergence, acetolactate synthase (ALS) inhibiting herbicide. It was first registered for use by the U.S. EPA in 2009. It is liquid in formulation and used for large-scale control of submerged, emergent, and floating-leaf vegetation. Information presented here can be found in the EPA pesticide fact sheet (EPA Penoxsulam 2004).

Mode of Action and Degradation

Penoxsulam is a slow-acting herbicide that is absorbed by above- and below-ground plant tissue and translocated throughout the plant. Penoxsulam interferes with plant growth by inhibiting the AHAS/ALS enzyme which in turn inhibits the production of important amino acids (Tranel and Wright 2002). Plant injury or death usually occurs between 2 and 4 weeks following application.

Penoxsulam is highly mobile but not persistent in either aquatic or terrestrial settings. However, the degradation process is complex. Two degradation pathways have been identified that result in at least 13 degradation products that persist for far longer than the original chemical. Both microbial- and photo-degradation are likely important means by which the herbicide is removed from the environment (Monika et al. 2017). It is relatively stable in water alone without sunlight, which means it may persist in light-limited areas.

The half-life for penoxsulam is between 12 and 38 days. Penoxsulam must remain in contact with plants for around 60 days. Thus, supplemental applications following initial treatment may be required to maintain adequate concentration exposure time (CET). Due to the long CET requirement, penoxsulam is likely best suited to large-scale or whole-lake applications.

Toxicology

Penoxsulam is unlikely to be toxic to animals but may be "slightly toxic" to birds that consume it. Human health studies have not revealed evidence of acute or chronic toxicity, though some indication of endocrine disruption deserves further study. However, screening-level assessments of risk have not been conducted on the major degradates which may have unknown non-target effects. Penoxsulam itself is unlikely to bioaccumulate in fish.

Species Susceptibility

Penoxsulam is used to control monocot and dicot plant species in aquatic and terrestrial environments. The herbicide is often applied at low concentrations of 0.002-0.02 ppm (2-20 ppb), but as a result long exposure times are usually required for effective target species control (Cheshier et al. 2011; Mudge et al. 2012b). For aquatic plant management applications, penoxsulam is most commonly utilized for control of hydrilla (*Hydrilla verticillata*). It has also been used for control of giant salvinia (*Salvinia molesta*), water hyacinth (*Eichhornia crassipes*), and water lettuce (*Pistia stratiotes*; Richardson and Gardner 2007; Mudge and Netherland 2014). However, the herbicide is only semi-selective; it has been implicated in injury to non-target emergent native species, including arrowheads (*Sagittaria* spp.) and spikerushes (*Eleocharis* spp.) and free-floating species like duckweed (Mudge and Netherland 2014; Cheshier et al. 2011). Penoxsulam can also be used to control milfoils such as Eurasian watermilfoil (*Myriophyllum spicatum*) and variable-leaf watermilfoil (*M. heterophyllum*; Glomski and Netherland 2008). Seedling emergence as well as vegetative vigor is impaired by penoxsulam in both dicots and monocots, so buffer zone and dissipation reduction strategies may be necessary to avoid non-target impacts (EPA Penoxsulam 2004).

When used to treat salvinia, the herbicide was found to have effects lasting through 10 weeks following treatment (Mudge et al. 2012b). The herbicide is effective at low doses, but while low-concentration applications of slow-acting herbicides like penoxsulam often result in temporary growth regulation and stunting, plants are likely to recover following treatment. Thus, complementary management strategies should be employed to discourage early regrowth (Mudge et al. 2012b). In particular, joint biological and herbicidal control with penoxsulam has shown good control of water hyacinth (Moran 2012). Alternately, a low concentration may be maintained over time by repeated low-dose applications. Studies show that maintaining a low concentration for at least 8-12 weeks provided excellent control of salvinia, and that a low dose followed by a high-dose application was even more efficacious (Mudge et al. 2012b).

S.3.4. Physical Removal Techniques

There are several management options which involve physical removal of aquatic plants, either by manual or mechanical means. Some of these include manual and mechanical cutting and hand-pulling or Diver-Assisted Suction Harvesting (DASH).

S.3.4.1. Harvesting: Manual, Mechanical, and DASH

Manual and Mechanical Cutting

Manual and mechanical cutting involve slicing off a portion of the target plants and removing the cut portion from the waterbody. In addition to actively removing parts of the target plants, destruction of vegetative material may help prevent further plant growth by decreasing photosynthetic uptake, and preventing the formation of rhizomes, tubers, and other growth types (Dall Armellina et al. 1996a, 1996b; Fox et al. 2002). These approaches can be quick to allow recreational use of a waterbody but because the plant is still established and will continue to grow from where it was cut, it often serves to provide short-term relief (Bickel and Closs 2009; Crowell et al. 1994).

The amount of time for macrophytes to return to pre-cutting levels can vary between waterbodies and with the dominant plant species present (Kaenel et al. 1998). Some studies have suggested that annual or biannual cutting of Eurasian watermilfoil (*Myriophyllum spicatum*) may be needed, while others have shown biomass can remain low the year after cutting (Kimbel and Carpenter 1981; Painter 1988; Barton et al. 2013). Hydrilla (*Hydrilla verticillata*) has been shown to recover beyond pre-harvest levels within weeks in some cases (Serafy et al. 1994). In deeper waters, greater cutting depth may lead to increased persistence of vegetative control (Unmuth et al. 1998; Barton et al. 2013). Higher frequency of cutting, rather than the amount of plant that is cut, can result in larger reductions to propagules such as turions (Fox et al. 2002).

The timing of cutting operations, as for other management approaches, is important. For species dependent on vegetative propagules, control methods should be taken before the propagules are formed. However, for species with rhizomes, cutting too early in the season merely postpones growth while later-season cutting can better reduce plant abundance (Dall Armellina et al. 1996a, 1996b). Eurasian watermilfoil regrowth may be slower if cutting is conducted later in the summer (June or later). Cutting in the fall, rather than spring or summer, may result in the lowest amount of Eurasian watermilfoil regrowth the year after management (Kimbel and Carpenter 1981). However, managing early in the growing season may reduce non-target impacts to native plant populations when early-growing non-native plants are the dominant targets (Nichols and Shaw

1986). Depending on regrowth rate and management goals, multiple harvests per growing season may be necessary (Rawls 1975).

Vegetative fragments which are not collected after cutting can produce new localized populations, potentially leading to higher plant densities (Dall Armellina et al. 1996a). Eurasian watermilfoil and common waterweed (*Elodea canadensis*) biomass can be reduced by cutting (Abernethy et al. 1996), though Eurasian watermilfoil can maintain its growth rate following cutting by developing a more-densely branched form (Rawls 1975; Mony et al. 2011). Cutting and physical removal tend to be less expensive but require more effort than benthic barriers, so these approaches may be best used for small infestations or where non-native and native species inhabit the same stand (Bailey and Calhoun 2008).

Hand Pulling and Diver-Assisted Suction Harvesting

Hand-pulling and DASH involve removing rooted plants from the bottom sediment of the water body. The entire plant is removed and disposed of elsewhere. Hand-pulling can be done at shallower depths whereas DASH, in which SCUBA divers do the pulling, may be better suited for deeper aquatic plant beds. As a permit condition, DASH and hand-pulling may not result in lifting or removal of bottom sediment (i.e., dredging). Efforts should be made to preserve water clarity because turbid conditions reduce visibility for divers, slowing the removal process and making species identification difficult. When operated with the intent to distinguish between species and minimize disturbance to desirable vegetation, DASH can be selective and provide multi-year control (Boylen et al. 1996). One study found reduced cover of Eurasian watermilfoil both in the year of harvest and the following year, along with increased native plant diversity and reduced overall plant cover the year following DASH implementation (Eichler et al. 1993). However, hand harvesting or DASH may require a large time or economic investment for Eurasian watermilfoil and other aquatic vegetation control on a large-scale (Madsen et al. 1989; Kelting and Laxson 2010). Lake type, water clarity, sediment composition, underwater obstacles and presences of dense native plants, may slow DASH efforts or even prohibit the ability to utilized DASH. Costs of DASH per acre have been reported to typically range from approximately $5,060-8,100 (Cooke et al. 1993; Mattson et al. 2004). Additionally, physical removal of turions from sediments, when applicable, has been shown to greatly reduce plant abundance for multiple subsequent growing seasons (Caffrey and Monahan 2006), though this has not been implemented in Wisconsin due to the significant effort it requires.

Ecological Impacts of Physical Removal Techniques

Plants accrue nutrients into their tissues, and thus plant removal may also remove nutrients from waterbodies (Boyd 1970), though this nutrient removal may not be significant among lake types. Cutting and harvesting of aquatic plants can lead to declines in fish as well as beneficial zooplankton, macroinvertebrate, and native plant and mussel populations (Garner et al. 1996; Aldridge 2000; Torn et al. 2010; Barton et al. 2013). Many studies suggest leaving some vegetated areas undisturbed to reduce negative effects of cutting on fish and other aquatic organisms (Swales 1982; Garner et al. 1996; Unmuth et al. 1998; Aldridge 2000; Greer et al. 2012). Recovery of these populations to cutting in the long-term is understudied and poorly understood (Barton et al. 2013). Effects on water quality can be minimal but nutrient cycling may be affected in wetland systems

(Dall Armellina et al. 1996a; Martin et al. 2003). Cutting can also increase algal production, and turbidity temporarily if sediments are disturbed (Wile 1978; Bailey and Calhoun 2008).

Some changes to macroinvertebrate community composition can occur as a result of cutting (Monahan and Caffrey 1996; Bickel and Closs 2009). Studies have also shown 12-85% reductions in macroinvertebrates following cutting operations in flowing systems (Dawson et al. 1991; Kaenel et al. 1998). Macroinvertebrate communities may not rebound to pre-management levels for 4-6 months and species dependent on aquatic plants as habitat (such as simuliids and chironomids) are likely to be most affected. Reserving cutting operations for summer, rather than spring, may reduce impacts to macroinvertebrate communities (Kaenel et al. 1998).

Mechanical harvesting can also incidentally remove fish and turtles inhabiting the vegetation and lead to shifts in aquatic plant community composition (Engel 1990; Booms 1999). Studies have shown mechanical harvesting can remove between 2%-32% of the fish community by fish number, with juvenile game fish and smaller species being the primary species removed (Haller et al. 1980; Mikol 1985). Haller et al. (1980) estimated a 32% reduction in the fish community at a value of $6000/hectare. However, fish numbers rebounded to similar levels as an unmanaged area within 43 days after harvesting in the Potomac River in Maryland (Serafy et al. 1994). In addition to direct impacts to fish populations, reductions in fish growth rates may correspond with declines in zooplankton populations in response to cutting (Garner et al. 1996). Because divers are physically uprooting plants from the lake bed, hand removal may disturb benthic organisms. Additionally, DASH may also result in some accidental capture of fish and invertebrates, small amounts of sediment removal, or increased turbidity. It is possible that equipment modifications could help minimize some of these unintended effects.

C

APPENDIX C

Comment Response Document for the Official First Draft

- Onterra Response Comments to WDNR Comments
- LLPLD Planning Committee Response to WDNR Comments

Comments to Long Lake Draft Aquatic Plant Management Plan (4/29/19) – Comments Received 5/21/2019

Response by Eddie Heath (Onterra, LLC)

WDNR Official Comments: Carol Warden – Team Leader
(UW Trout Lake Station Center for Limnology Aquatic Invasive Species Specialist)

Contributing comments by:
Susan Knight, UW Trout Lake Station Center for Limnology Research Scientist
Michelle Nault, State-Wide Lakes & Reservoir Ecologist

1) Pages 24 and 25: Can we discuss the disparity in figures 2.3-3 and 2.3-4? I see you say that figure 2.3-3 is only depicting mapped beds and may not include everything found on a PI, still it may be worth more explanation that in 2011 46 acres were mapped and 91 acres were treated, in 2012 only 3.3 acres were mapped yet 65.9 acres were treated, and so on. As a reader, I don't grasp the reason for the larger treatment areas. An additional paragraph below Fig 2.3-3 was added to help with clarity.

2) Relating to previous comment, could an appendix be made showing maps of mapped EWM overlaid with treatment areas for each year treatment occurred? The report now includes a link/url where the annual reports from this grant-funded project can be accessed. These reports provide maps as suggested within this review comment as well as context to their design specifics.

3) Page 28 middle of first paragraph: should be "extracted relevant chapters."? Change made.

4) Page 29: you discuss a 15% threshold that BSLPOA adopted. It is unclear as to whether this is something LLPOA may be interested in doing as well or if it's just being used to define lake-wide mgmt. As outlined in the proceeding paragraphs, this text was provided to the LLPLD's planning committee and work team prior to development of their own management goals/actions. Therefore, this information is provided as a local example of how another lake group arrived at their management perspective.

5) Page 29, Nuisance control: missing a parenthesis toward the end of the of first paragraph. Change made.

6) Page 33, second to last paragraph: should read "this section provides AN update…" not AND. Change made.

7) When you simply use the word 'survey' or even 'pre-treatment survey' it can get a bit confusing. When you use this term the reader may not be completely sure if this is indicative of a quantitative sub-PI survey, or if it's one of your early season AIS meander surveys, or their late-season AIS meander survey (or a hydroacoustic survey…which you also do). Usually I can deduce what type of survey you are talking about by other context within the paragraph (i.e., if you state a pre-treatment survey following DNR protocols I

assume it's a sub-PI since we don't really have a DNR protocol for meander AIS mapping). For example, in one place you state that a "focused pre-treatment survey….would be used to finalize the permit." I am pretty sure this is the early season AIS meander survey, but it would be helpful if that was super clear as I could also see someone thinking this might be a sub-PI or even lakewide PI Expanded discussion of survey types was provided within the first two paragraphs under the sub-heading *Long Lake Historic EWM Management*. Attention was given to subsequent use of *survey* to ensure it clearly explains what surveys is being referenced. Additional details of what a pretreatment survey were included within the Implementation Plan Section.

8) EWM FOO is very low now. It was at about 15% in 2006, The EWM population in 2006 was 26.3% and then was very low in 2012. They want a "trigger" of 20%, but we have no idea where that number came from. Some additional text about how thresholds are selected has been added to the "Lake-wide Population Management" sub-section of "Long Lake Future Management Discussions." Context of how the LLPLD arrived at this threshold has been included within the Implementation Plan Section. Some reviewers would not support a whole-lake herb treatment even if FOO hits 20%, because of the geometry, morphometry of the lake. Additional text has been added to make it clearer that this threshold is not for implementing whole-lake herbicide treatment. In earlier communications with the WDNR review team as the trigger was being developed, this response was given: "We agreed that if 20% was a threshold to start conversation that would be fine, even in smaller areas such as the southern bay, but it should not be a hard line when crossed that treatment automatically occurs." It's hard to believe the narrow band of EWM along the east and west shores is much of an impediment to recreation. Statement of opinion.

9) The report indicates they will not do EWM treatments unless the population is in a bay (and not in a narrow band), the polygon is >5 acres, and it is very dominant. That suggests to only propose to treat in the south bay, and probably never along the shore along most of the lake. Likely a correct interpretation based upon current herbicide tools and BMPs. How narrow is narrow? No formal definition exists and would have to rely on management experience followed by monitoring to evaluate outcomes.

10) Why was there a treatment in 2013? As discussed above, a large quantity of herbicide was used when EWM was very low. More explanation on this strategy was added to this document. It brings into question whether the history of treatments on Long were not thoughtfully planned. This statement of opinion does not add value to this review and may be viewed by some as offensive. Onterra and the LLPLD have been active participants in research that has resulted in state-wide changes in BMPS over time. Decisions in 2013 did not have the luxury of today's advanced understanding of the subject.

11) Possibly, the lake group or Onterra is hoping to propose ProcellaCor for spot treatments. It is unclear the basis for this review comment, as this product is not specifically discussed within this document. Unless new ProcellaCor studies demonstrate it sticks like glue where applied, the majority of Long Lake will not be a candidate for spot treatments. The implementation plan was constructed to account for potential changes in available management options over time.

12) It is understood the lake group is dedicated to getting rid of EWM, and Onterra does a decent job saying that will never happen. On the other hand, Onterra reports the strategy to treat all colonized areas was "approved at the July 2012 annual meeting (40 in favor, 0 against, 0 abstain)." Why give us the vote? The vote was provided to demonstrate the district's level of acceptance for the recently WDNR-approved management plan and associated strategy at that time. Maybe Onterra wants us to know how much pressure they are under, but the vote doesn't reflect what may actually be an achievable goal. This statement of opinion does not offer value to this review and may be viewed by some as offensive.

13) It would be interesting to know how the Najas guadalupensis behaves in Long. Does it form rafts? Southern naiad was found causing nuisance conditions in Big Sand Lake years ago that loosely corresponded with years of lower water levels. Southern naiad has not been noted as causing nuisance conditions on Long Lake. The report suggests Naj guad may have come in because of herbicides. That is possible, but it came in heavily in other lakes that were never treated with herbicides. Increases in southern naiad populations following select herbicides treatment has been hypothesized by Dr. Donald Les (a researcher of the genetics of this plant, now at UConn), but the data from Long Lake (whole-lake PI and sub-PI) do <u>not</u> really support the connection. But we believe it was important to include for discussion.

LLPLD Planning Committee Response to WDNR Comments
Dan Anderson – June 9, 2019

RE: Comments to Carol Warden and colleagues comments concerning the Long Lake Aquatic Plant Management Plan Update dated May 21, 2019.

As a full time resident of Long Lake and a former Long Lake of Phelps Lake District Board Commissioner and Chairman, I feel obligated to respond to some of the comments in WDNR correspondence/memorandum. I have been actively involved in the Long Lake Association and Long Lake of Phelps Lake District since 2005. I have always felt that we have had a good working relationship with the WDNR, specifically Kevin Gauthier and Steve Gilbert. Personally, I feel that without their guidance and support we would not have accomplished the goals we have established in our current and past Lake Management Plans. After some reading some of the comments in the May 21, 2019 document, I believe they imply that the district is not judicious and thoughtful as they should be concerning the ecosystem of our lake when addressing the issue of EWM.

- Starting in 2006 we and the Big Sand Lake Association partnered with the WDNR to include Big Sand and Long Lakes in the WDNR Scientific Lake Research Program. At that time Big Sand had the largest infestation of EWM in Vilas County. Long being connected to Big Sand Lake also had a large amount of EWM.
- Over the next 7 years the Long Lake District worked with the WDNR and the USACE to research and evaluate the different chemical treatments of EWM. We took water samples after specific time lapses to evaluate the efficacy of the chemical based on the unique littoral zone and water flow of Long Lake. These samples were sent to University of Florida for analysis. From the onset, we were sensitive about the impact of the application of the chemicals.
- The district researched the various application mythologies and found the applicator from the state of Oregon who could provide the best use of computer technologies to apply the chemicals where they needed to be at the right amount.
- I found the comment in point 12: "Why give us vote" perplexing and lacking understanding. From our initial comments to our riparian owners about managing EWM we stated it will not be totally removed. We want to restrict, control, monitor and mitigate EWM. We need our riparian owners' comments and support and that is why we take a vote. Candidly, the comment about Onterra was inappropriate and not warranted.
- As with many lakes, Long Lake has some unique characteristics in dealing with EWM in the littoral around the lake and how to manage the issue. Over the past 12 years we have attempted be thoughtful and careful in our approach to managing EWM. From chemical treatments, DASH, hand harvesting and letting nature take its course. Each approach has its benefits and drawbacks. The riparian owners have been supportive of the efforts of the Lake District Board and the WDNR. We need to keep this positive collaboration going forward and understand it is a partnership in protecting the ecosystem of Long Lake.