Forest Lake

Vilas County, Wisconsin

Comprehensive Management Plan Update

November 2024

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Acknowledgements

This management planning effort was truly a team-based project and could not have been completed without the input of the following individuals:

Forest Lake Association Board Members

Linda Bogdala	Linda Coyle	Larry Heindl
Julie Heberer	Jeff Ladd	Donna Livingston
Kristie Ohberg	Dave Pushka	Rhonda Sprague
Tom Ward		

Lake Management Plan Committee Members

Mark Dryer	Tom Macak	Laura Gage
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1.0 INTRODUCTION

Forest Lake is a seepage lake situated in the Town of Land O'Lakes, in Vilas County, in northern Wisconsin with a maximum depth of 60 feet. Based on a 2022 aerial photograph, the lake was determined to be approximately 471 acres. Since Forest Lake is a seepage lake, water levels fluctuate every year based on weather patterns (Figure 1.0-1). This oligo-mesotrophic lake has a relatively small watershed when compared to the size of the lake. The Forest Lake ecosystem contains over 50 native plant species, of which fern-leaf pondweed and common waterweed were the most common plants.

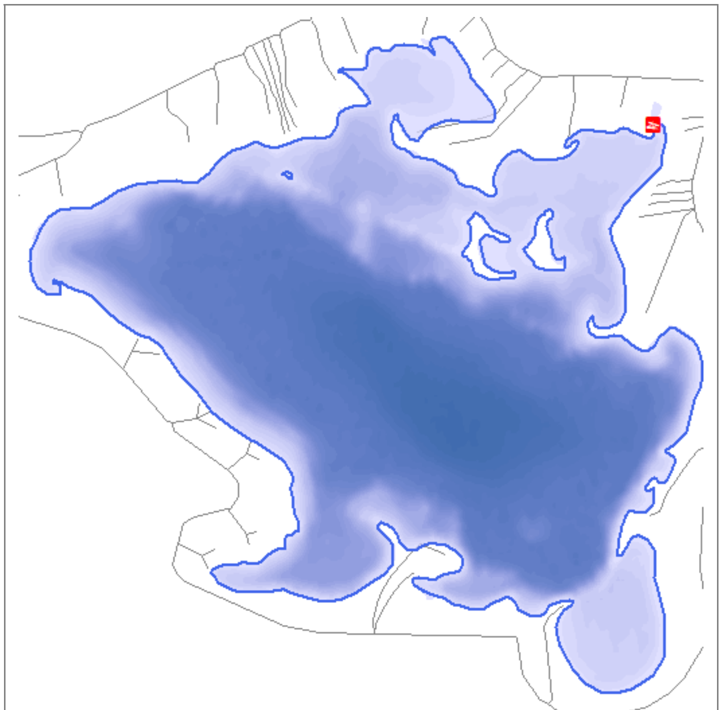


Figure 1.0-1. Forest Lake, Vilas County.

Forest Lake's primary management unit is the Forest Lake Association, Inc. (FLA) and has partnered with the Forest Lake Preservation Foundation Inc (FLPF) to sponsor recent WNDR grants. The FLA completed the lake's first comprehensive management plan in early 2019 and has been implementing that plan since that time. The group has continued to participate in the Citizens Lake Monitoring Network (CLMN), partner with other management units, conduct volunteer monitoring of AIS, manage EWM, facilitate periodic quantitative plant surveys, and perform watercraft inspections through Clean Boats Clean Waters Program (CBCW).

With Onterra's assistance, the FLPF successfully applied for a WDNR grant in November of 2022 to update the 2019 management plan for the lake as well as consider changes that have occurred on the lake since that time. This was completed by gathering and analyzing historical and current ecological data, identifying threats, determining goals and values of stakeholders, present feasible management actions, and increase the lake group's capacity to implement the management plan. Fieldwork for this effort was conducted during the summer of 2023, with planning discussions and public outreach occurring during the winter and spring of 2024.

2.0 STAKEHOLDER PARTICIPATION

Stakeholder participation is an important part of any management planning exercise. During this project, stakeholders were not only informed about the project and its results, but also introduced to important concepts in lake ecology. The objective of this component in the planning process is to accommodate communication between the planners and the stakeholders. The communication is educational in nature, both in terms of the planners educating the stakeholders and vice-versa. The planners educate the stakeholders about the planning process, the functions of their lake ecosystem, their impact on the lake, and what can realistically be expected regarding the management of the aquatic system. The stakeholders educate the planners by describing how they would like the lake to be, how they use the lake, and how they would like to be involved in managing it. All of this information is communicated through multiple meetings that involve the lake group as a whole or a focus group called a Planning Committee, the completion of a stakeholder survey seeking input from all lakeshore property owners and Association members.

The highlights of this component are described below. Materials used during the planning process can be found in Appendix A.

2.1 Board of Directors Planning Meeting

A single meeting was held on April 22, 2024 with the Forest Lake Association board and Todd Hanke, an aquatic ecologist with Onterra, to detail the findings of the current studies, discuss how the lake has changed or not changed since the last planning process, and assess the successes and failures the association has had with implementing the comprehensive management plan. Each goal and action were discussed and updated as necessary to utilize new information and Best Management Practices. Much of the focus was on the association's continued management of EWM within Forest Lake.

2.2 Management Plan Review and Adoption Process

After the Committee members approve the Implementation Plan (Section 5.0), a draft of the entire Lake Management Plan Update was provided to WDNR for agency review in late-July 2024. The draft Plan was made available via the Association's outreach and communication avenues for public comment for 21 days. The only public comments that were received were to update the acknowledged persons involved in this project and listed on the cover page. No agency review comments were received after the draft plan with exception of from WDNR water resource management specialist Ty Krajewski. Mr. Krajewski's response was received in November 2024 and stated having no substantial comments and that the project may move towards finalization. The final Plan was compiled and issued to WDNR and FLA in November 2024.

Stakeholder Survey

As a part of this project, a stakeholder survey was distributed to Forest Lake Association members and riparian property owners around Forest Lake. The survey was designed by Onterra staff and the Forest Lake Association planning committee and reviewed by a WDNR social scientist. During June 2023, the nine-page, 35-question survey was posted online through Survey Monkey for survey-takers to answer electronically. If requested, a hard copy was sent with a self-addressed stamped envelope for returning the survey anonymously. The returned hardcopy surveys were entered into the online version by a Forest Lake Association volunteer for analysis. Sixty-two percent of the surveys were returned. Since the survey reached above a 60% response rate, these results can be used to portray population projections accurately, and make conclusions with statistical validity. The data were analyzed and summarized by Onterra for use at the planning meetings and within the management plan. The full survey and results can be found in Appendix B, while discussion of those results is integrated within the appropriate sections of the management plan and a general summary is discussed below.

Based upon the results of the stakeholder survey, much was learned about the people who use and care for Forest Lake. 29% of respondents indicated that they live on the lake during the summer months only, while 39% visit on weekends through the year, 25% are year-round residents, and no stakeholders indicated they have undeveloped property. 18% of respondents have owned their property for over 11 years, and 45% have owned their property for over 25 years.

The primary activities that are important reasons for owning property on the lake include relaxing/entertaining, nature viewing, and open water fishing, and (Figure 2.2-1). Top concerns regarding Forest Lake listed by the stakeholders were current AIS within the lake, shoreline erosion, and water quality degradation (Figure 2.2-2).

Question 7: Please rank up to three activities that are important reasons for owning your property on or near the lake.

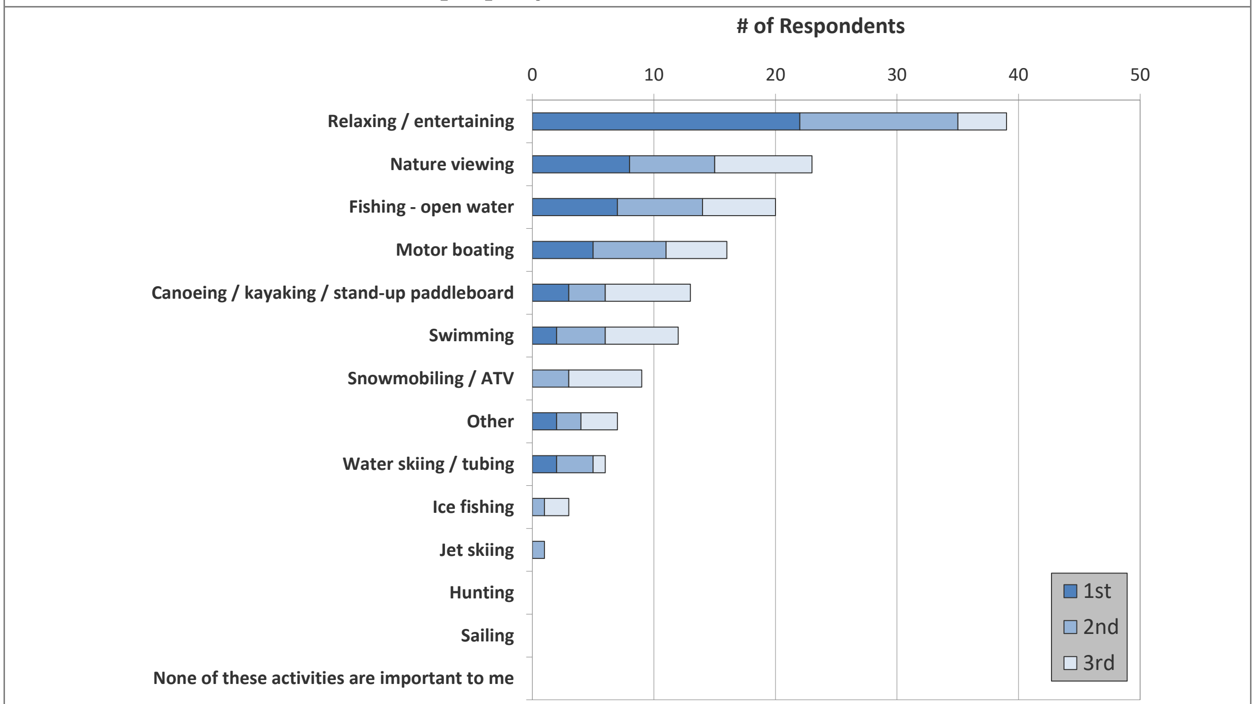


Figure 2.2-1. Select survey responses from the Forest Lake stakeholder survey. Additional questions and response charts may be found in Appendix B.

Question 15: Please rank your top three concerns regarding Forest Lake, with 1 being your greatest concern.

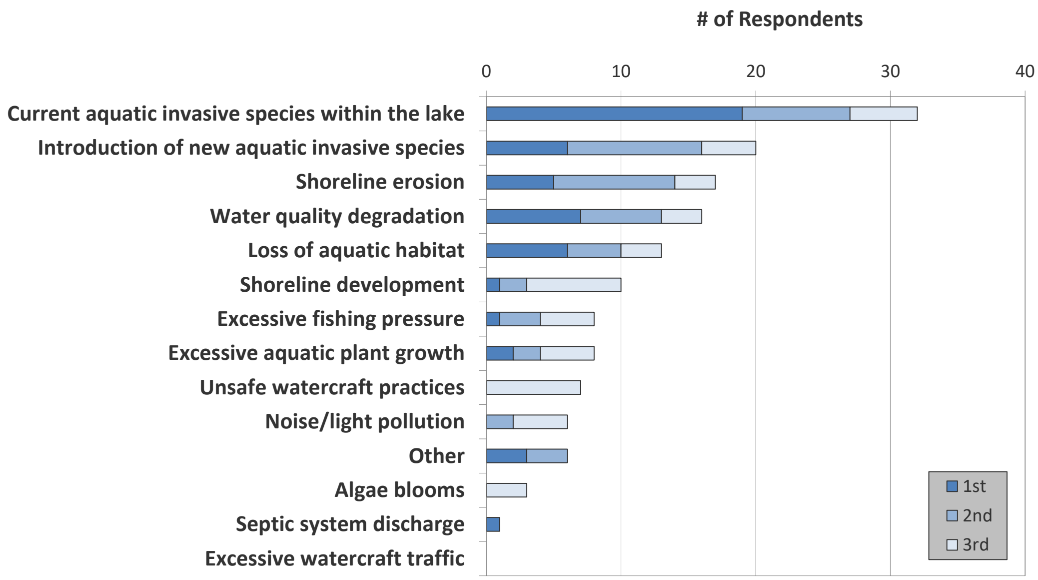


Figure 2.2-2. Select survey responses from the Forest Lake stakeholder survey, continued. Additional questions and response charts may be found in Appendix B.

3.0 RESULTS & DISCUSSION

3.1 Lake Water Quality

Water Quality Data Analysis and Interpretation

Reporting of water quality assessment results can often be a difficult and ambiguous task. Foremost is that the assessment inherently calls for a baseline knowledge of lake chemistry and ecology. Many of the parameters assessed are part of a complicated cycle and each element may occur in many different forms within a lake. Furthermore, water quality values that may be considered poor for one lake may be considered good for another because judging water quality is often subjective. However, focusing on specific aspects or parameters that are important to lake ecology, comparing those values to similar lakes within the same region and historical data from the study lake provides an excellent method to evaluate the quality of a lake's water.

Many types of analyses are available for assessing the condition of a particular lake's water quality. In this document, the water quality analysis focuses 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 here. Specific forms of water quality analyses are used to indicate not only the health of the lake, but also to provide a general understanding of the lake's ecology and assist in management decisions. Each type of available analysis is elaborated on below.

As mentioned above, chemistry is a large part of water quality analysis. In most cases, listing the values of specific parameters really does not lead to an understanding of a lake's water quality, especially in the minds of non-professionals. A better way of relating the information is to compare it to lakes with similar physical characteristics and lakes within the same regional area. In this document, a portion of the water quality information collected on Forest Lake is compared to other lakes in the state with similar characteristics as well as to lakes within the northern region. In addition, the assessment can also be clarified by limiting the primary analysis to parameters that are important in the lake's ecology and trophic state (see below). Three water quality parameters are focused upon in the Forest Lake water quality analysis:

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 disk 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 disk 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 quadrants (a Secchi disk) 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 disk 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. In addition, studies have shown that water clarity is used by most lake users to judge water quality – clear water equals clean water (Canter, Nelson, & Everett, 1994) (Dinius, 2007) (Smith, Cragg, & Croker, 1991).

Trophic State

Total phosphorus, chlorophyll-*a*, and water clarity values are directly related to the trophic state of the lake. As nutrients, primarily phosphorus, accumulate within a lake, its productivity increases and the lake progresses through three trophic states: oligotrophic, mesotrophic, and finally eutrophic. Every lake will naturally progress through these states and under natural conditions (i.e. not influenced by the activities of humans) this progress can take tens of thousands of years. Unfortunately, human influence has accelerated this natural aging process in many Wisconsin lakes. Monitoring the trophic state of a lake gives stakeholders a method by which to gauge the productivity of their lake over time. Yet, classifying a lake into one of three trophic states often does not give clear indication of where a lake really exists in its trophic progression because each trophic state represents a range of productivity. Therefore, two lakes classified in the same trophic state can actually have very different levels of production.

> Trophic states describe the lake's ability to produce plant matter (production) and include three continuous classifications: Oligotrophic lakes are the least productive lakes and are characterized by being deep, having cold water, and few plants. Eutrophic lakes are the most productive and normally have shallow depths, warm water, and high plant biomass. Mesotrophic lakes fall between these two categories.

However, through the use of a trophic state index (TSI), an index number can be calculated using phosphorus, chlorophyll-*a*, and clarity values that represent the lake's position within the eutrophication process. This allows for a clearer understanding of the lake's trophic state while facilitating clearer long-term tracking. (Carlson, 1977) presented a trophic state index that gained great acceptance among lake managers.

Limiting Nutrient

The limiting nutrient is the nutrient which is in shortest supply and controls the growth rate of algae and some macrophytes within the lake. This is analogous to baking a cake that requires four eggs, and four cups each of water, flour, and sugar. If the baker would like to make four cakes, he needs 16 of each ingredient. If he is short two eggs, he will only be able to make three cakes even if he has sufficient amounts of the other ingredients. In this scenario, the eggs are the limiting nutrient (ingredient).

In most Wisconsin lakes, phosphorus is the limiting nutrient controlling the production of plant biomass. As a result, phosphorus is often the target for management actions aimed at controlling plants, especially algae. The limiting nutrient is determined by calculating the nitrogen to phosphorus ratio within the lake. Normally, total nitrogen and total phosphorus values from the surface samples taken during the summer months are used to determine the ratio. Results of this ratio indicate if algal growth within a lake is limited by nitrogen or phosphorus. If the ratio is greater than 15:1, the lake is considered phosphorus limited; if it is less than 10:1, it is considered nitrogen limited. Values between these ratios indicate a transitional limitation between nitrogen and phosphorus.

Temperature and Dissolved Oxygen Profiles

Temperature and dissolved oxygen profiles are created simply by taking readings at different water depths within a lake. Although it is a simple procedure, the completion of several profiles over the course of a year or more provides a great deal of information about the lake. Much of this information relates to whether the lake thermally stratifies or not, which is determined primarily through the temperature profiles. Lakes that show strong stratification during the summer and winter months need to be managed differently than lakes that do not. Normally, deep lakes stratify to some extent, while shallow lakes (less than 17 feet deep) do not.

> **Lake stratification** occurs when temperature gradients are developed with depth in a lake. During stratification the lake can be broken into three layers: The epilimnion is the top layer of water which is the warmest water in the summer months and the coolest water in the winter months. The hypolimnion is the bottom layer and contains the coolest water in the summer months and the warmest water in the winter months. The metalimnion, often called the thermocline, is the middle layer containing the steepest temperature gradient.

Dissolved oxygen is essential in the metabolism of nearly every organism that exists within a lake. For instance, fish kills are often the result of insufficient amounts of dissolved oxygen. However, dissolved oxygen's role in lake management extends beyond this basic need by living organisms. In fact, its presence or absence impacts many chemical processes that occur within a lake. Internal nutrient loading is an excellent example that is described below.

Internal Nutrient Loading*

In lakes that support stratification, whether throughout the summer or periodically between mixing events, the hypolimnion can become devoid of oxygen both in the water column and within the sediment. When this occurs, iron changes from a form that normally binds phosphorus within the sediment to a form that releases it to the overlaying water. This can result in very high concentrations of phosphorus in the hypolimnion. Then, during turnover events, these high concentrations of phosphorus are mixed within the lake and utilized by algae and some macrophytes. In lakes that mix periodically during the summer (polymictic lakes), this cycle can *pump* phosphorus from the sediments into the water column throughout the growing season. In lakes that only mix during the spring and fall (dimictic lakes), this burst of phosphorus can support late-season algae blooms and even last through the winter to support early algal blooms the following spring. Further, anoxic conditions under the winter ice in both polymictic and dimictic lakes can add smaller loads of phosphorus to the water column during spring turnover that may support algae blooms long into the summer. This cycle continues year after year and is termed "internal phosphorus loading"; a phenomenon that can support nuisance algal blooms decades after external sources are controlled.

The first step in the analysis is determining if the lake is a candidate for significant internal phosphorus loading. Water quality data and watershed modeling are used to determine actual and predicted levels of phosphorus for the lake. When the predicted phosphorus level is well below the actual level, it may be an indication that the modeling is not accounting for all of the phosphorus sources entering the lake. Internal nutrient loading may be one of the additional contributors that may need to be assessed with further water quality analysis and possibly additional, more intense studies.

Non-Candidate Lakes

- Lakes that do not experience hypolimnetic anoxia.
- Lakes that do not stratify for significant periods (i.e. days or weeks at a time).
- Lakes with hypolimnetic total phosphorus values less than 200 μg/L.

Candidate Lakes

- Lakes with hypolimnetic total phosphorus concentrations exceeding 200 μg/L.
- Lakes with epilimnetic phosphorus concentrations that cannot be accounted for in watershed phosphorus load modeling.

Specific to the final bullet-point, during the watershed modeling assessment, the results of the modeled phosphorus loads are used to estimate in-lake phosphorus concentrations. If these estimates are much lower than those actually found in the lake, another source of phosphorus must be responsible for elevating the in-lake concentrations. Normally, two possibilities exist: 1) shoreland septic systems, and 2) internal phosphorus cycling. If the lake is considered a candidate for internal loading, modeling procedures are used to estimate that load.

Comparisons with Other Datasets

The WDNR document *Wisconsin 2020 Consolidated Assessment and Listing Methodology* (WDNR, 2019) is an excellent source of data for comparing water quality from a given lake to lakes with similar features and lakes within specific regions of Wisconsin. Water quality among lakes, even among lakes that are located in close proximity to one another, can vary due to natural factors such as depth, surface area, the size of its watershed and the composition of the watershed's land cover. For this reason, the water quality of Forest Lake will be compared to lakes in the state with similar physical characteristics. The WDNR groups Wisconsin's lakes into ten natural communities (Figure 3.1-1).

First, the lakes are classified into three main groups: (1) lakes and reservoirs less than 10 acres, (2) lakes and reservoirs greater than or equal to 10 acres, and (3) a classification that addresses special waterbody circumstances. The last two categories have several sub-categories that provide attention to lakes that may be shallow, deep, play host to cold water fish species or have unique hydrologic patterns. Overall, the divisions categorize lakes based upon their size, stratification characteristics, and hydrology. An equation developed by Lathrop and Lillie (Lathrop & Lillie, 1980), which incorporates the maximum depth of the lake and the lake's surface area, is used to predict whether the lake is considered a shallow (mixed) lake or a deep (stratified) lake. The lakes are further divided into classifications based on their hydrology and watershed size:

> **Seepage Lakes** have no surface water inflow or outflow in the form of rivers and/or streams.
>
> **Drainage Lakes** have surface water inflow and/or outflow in the form of rivers and/or streams.
>
> > Headwater drainage lakes have a watershed of less than 4 square miles.

Lowland drainage lakes have a watershed of greater than 4 square miles.

Because of its depth, small watershed and hydrology, Forest Lake is classified as a deep seepage lake (category 7 on Figure 3.1-1).

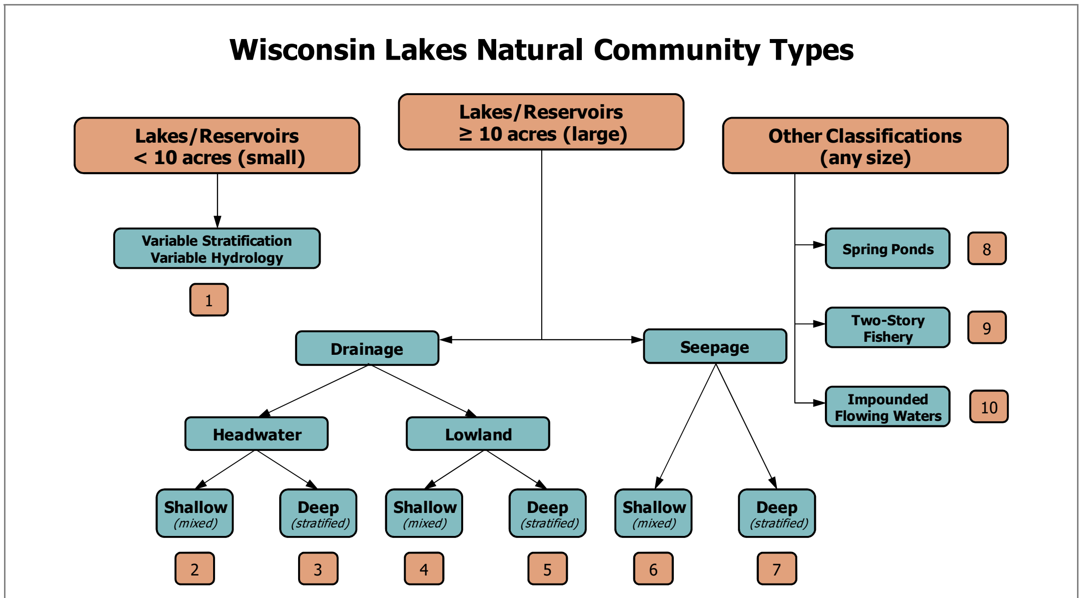


Figure 3.1-1. Wisconsin Lake Natural Communities. Adapted from WDNR 2017.

(Garrison, et al., 2008) developed statewide median values for total phosphorus, chlorophyll-*a*, and Secchi disk transparency for six of the lake classifications. Though they did not sample sufficient lakes to create median values for each classification within each of the state's ecoregions, they were able to create median values based on all of the lakes sampled within each ecoregion (Figure 3.1-2). Ecoregions are areas related by similar climate, physiography, hydrology, vegetation and wildlife potential. Comparing ecosystems in the same ecoregion is sounder than comparing systems within manmade boundaries such as counties, towns, or states. Forest Lake is within the Northern Lakes and Forests ecoregion.

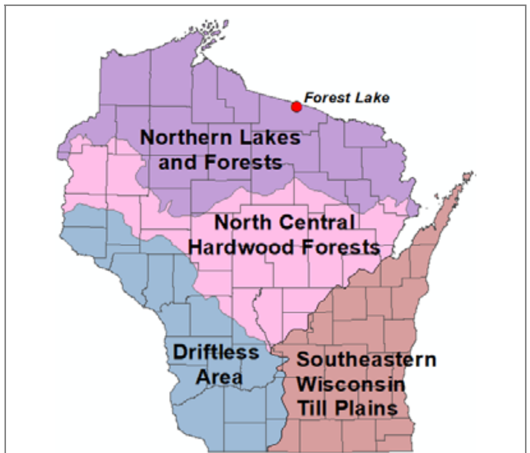


Figure 3.1-2. Location of Forest Lake within the ecoregions of Wisconsin. After (Nichols, 1999).

The Wisconsin 2020 Consolidated Assessment and Listing Methodology document also helps stakeholders understand the health of their lake compared to other lakes within the state. Looking at pre-settlement diatom population compositions from sediment cores collected from numerous lakes around the state, they were able to infer a reference condition for each lake's water quality prior to human development within their watersheds. Using these reference conditions and current water quality data, the assessors were able to rank phosphorus, chlorophyll-*a*, and Secchi disk transparency values for each lake class into categories ranging from excellent to poor.

These data along with data corresponding to statewide natural lake means, historic, current, and average data from Forest Lake is displayed in Figures 3.1-3 - 3.1-7. Please note that the data in these graphs represent concentrations and depths taken only during the growing season (April-October) or summer months (June-August). Furthermore, the phosphorus and chlorophyll-*a* data represent only surface samples. Surface samples are used because they represent the depths at which algae grow and depths at which phosphorus levels are not greatly influenced by phosphorus being released from bottom sediments.

Forest Lake Water Quality Analysis

Forest Lake Long-term Trends

The 2019 Forest Lake Comprehensive Management Plan included water quality data up to 2016. Total phosphorus, chlorophyll-*a*, and Secchi disk transparency were discussed with the overarching conclusion being that Forest Lake's water quality is excellent. That conclusion still holds true through the 2023 discussed here.

Since 2016, total phosphorus concentrations in Forest Lake fluctuated slightly within the *Excellent* and *Good* categories (Figure 3.1-3), but overall, the summer weighted mean for all of the available data remains in the *Excellent* category. Further the phosphorus levels are still lower than those found in most Wisconsin deep seepage lakes and all types of lakes in the Northern Lakes and Forests Ecoregion.

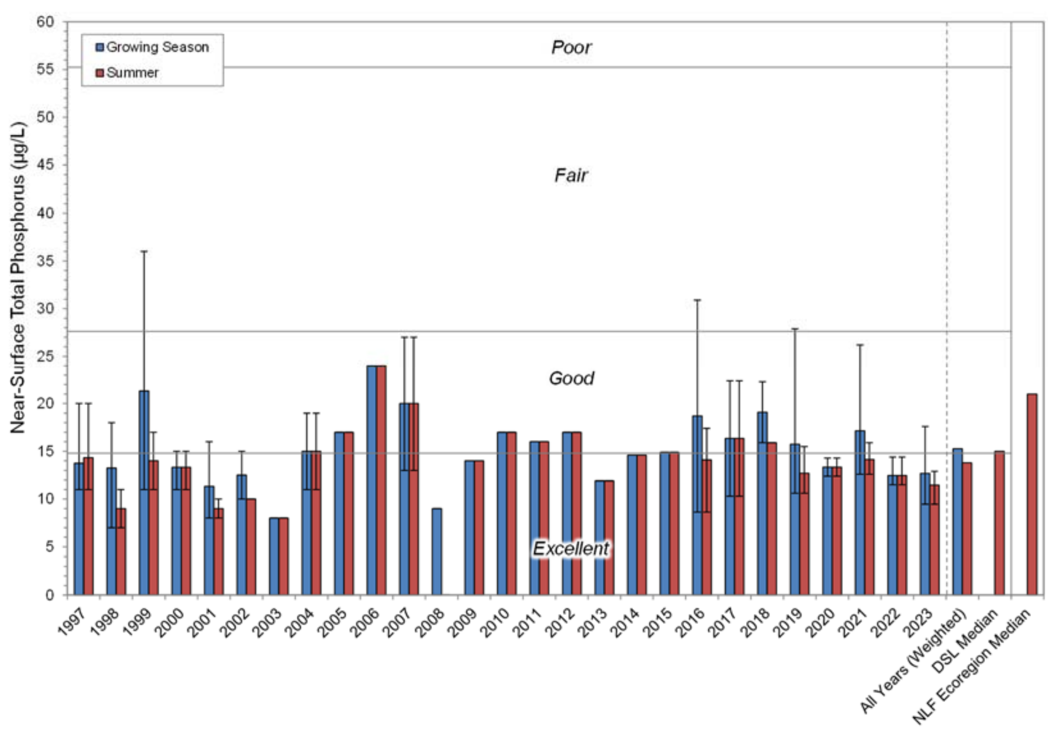


Figure 3.1-3. Forest Lake, statewide class 7 lakes, and regional total phosphorus concentrations. Mean values calculated with summer month surface sample data. Water Quality Index values adapted from WDNR PUB WT-913.

Like phosphorus concentrations, chlorophyll-*a* levels have remained low in Forest Lake. In fact, all but one value, a concentration of 6.71 µg/L collected in May 2023, were well within the Excellent category since 2016 (Figure 3.1-4). The weighted summer mean of 2.3 µg/L is much lower than similar lakes in the state and all lakes within the ecoregion.

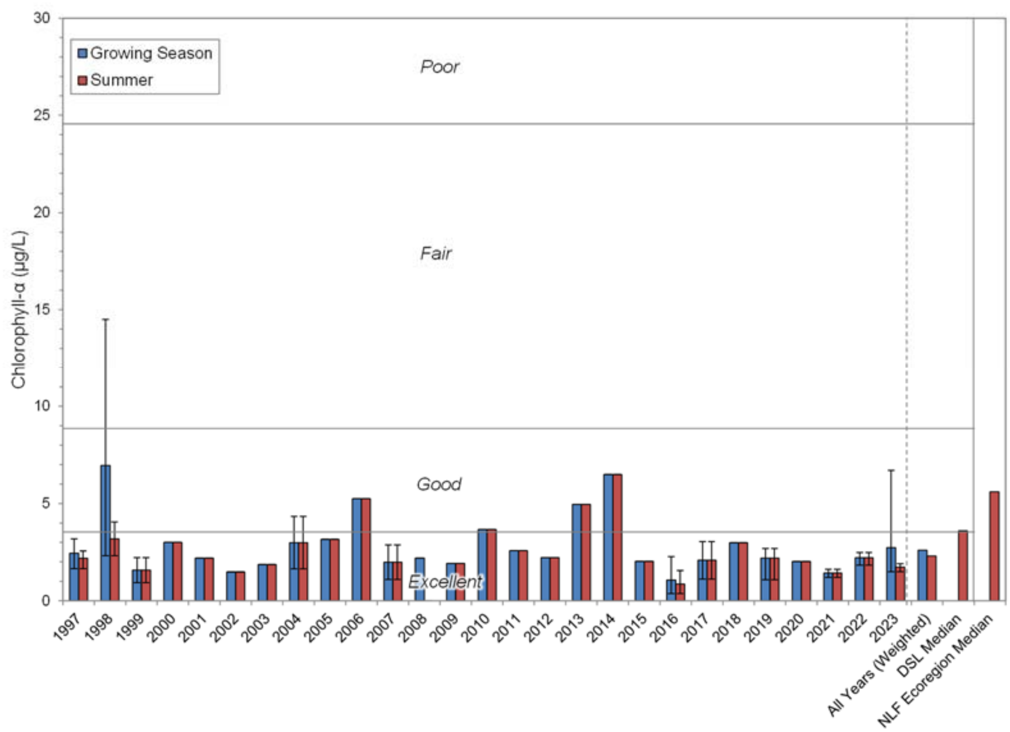


Figure 3.1-4. Forest Lake, statewide class 7 lakes, and regional chlorophyll-a concentrations. Mean values calculated with summer month surface sample data. Water Quality Index values adapted from WDNR PUB WT-913.

Algae account for the greatest amount of particulate matter in Wisconsin lakes, so with the incredibly low levels of chlorophyll-*a* found in Forest Lake, it is no surprise that the lake has very high water clarity (Figure 3.1-5). All values recorded at Forest Lake since 2016, with the exception of the May 2023 sample, were all within the *Excellent* category. During 2023, some of the deepest Secchi transparency values were recorded at the lake.

Overall, the trophic parameters (total phosphorus, chlorophyll-*a*, and Secchi clarity), at Forest Lake appear to have no trend over the course of the full dataset. Like all lakes, these parameters fluctuate from year-to-year and from season-to-season, but on average, all remain in the *Excellent* category and are better than the comparable datasets from the state and the ecoregion.

Internal nutrient loading is discussed in detail in the 2019 management plan. In that report, near-surface and near-bottom total phosphorus concentrations during stratification are discussed along with patterns seen in concentrations during the spring and fall turnover events. Like many lakes in Wisconsin, hypolimnetic (deep layer) phosphorus levels increase because the layer becomes anoxic during the summer. In August 2016, the near-bottom total phosphorus concentration was 392 µg/L, which was over 45 times higher than the near-surface concentration on the same day. In August 2023, the near-bottom sample concentration was 581 µg/L, which was almost 48 times the concentration found near the surface. It is apparent but not surprising that internal nutrient loading was also documented during 2023; however, as stated in the 2019 management plan, while the phenomenon occurs at the lake, it is not significant enough to be noticeable. To accurately estimate the extent of internal loading and its role in the overall phosphorus budget of Forest Lake, a moderately intense sampling schedule would need to be completed over 2-3 years, which is likely not necessary at this time.

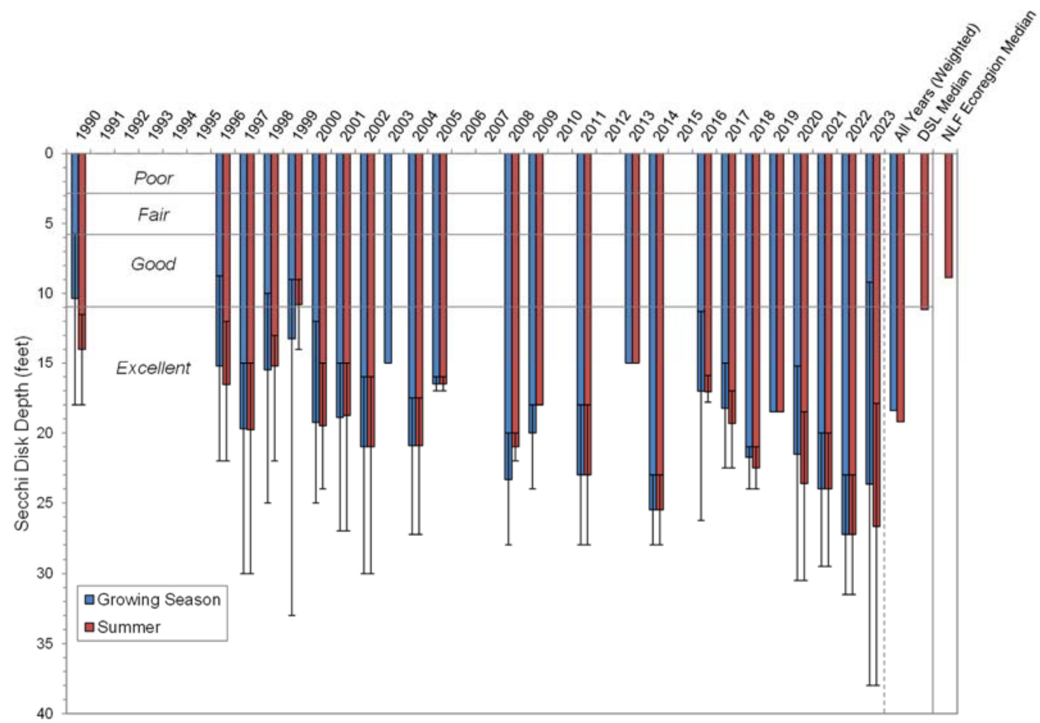


Figure 3.1-5. Forest Lake, statewide class 7 lakes, and regional Secchi disk clarity values. Mean values calculated with summer month surface sample data. Water Quality Index values adapted from WDNR PUB WT-913.

Limiting Plant Nutrient of Forest Lake

Using 2023 midsummer nitrogen and phosphorus concentrations from Forest Lake, a nitrogen:phosphorus ratio of 32:1 was calculated. This finding indicates that Forest Lake is indeed phosphorus limited as are the vast majority of Wisconsin lakes. A similar result was calculated during the 2019 management planning project.

Forest Lake Trophic State

Figure 3.1-6 contain the TSI values for Forest Lake. The TSI values calculated with Secchi disk, chlorophyll-*a*, and total phosphorus values range in values within the mesotrophic range. In general, the best values to use in judging a lake's trophic state are the biological parameters; therefore, relying primarily on total phosphorus and chlorophyll-*a* TSI values, it can be concluded that Forest Lake is oligo-mesotrophic. This same conclusion was drawn in the 2019 management plan.

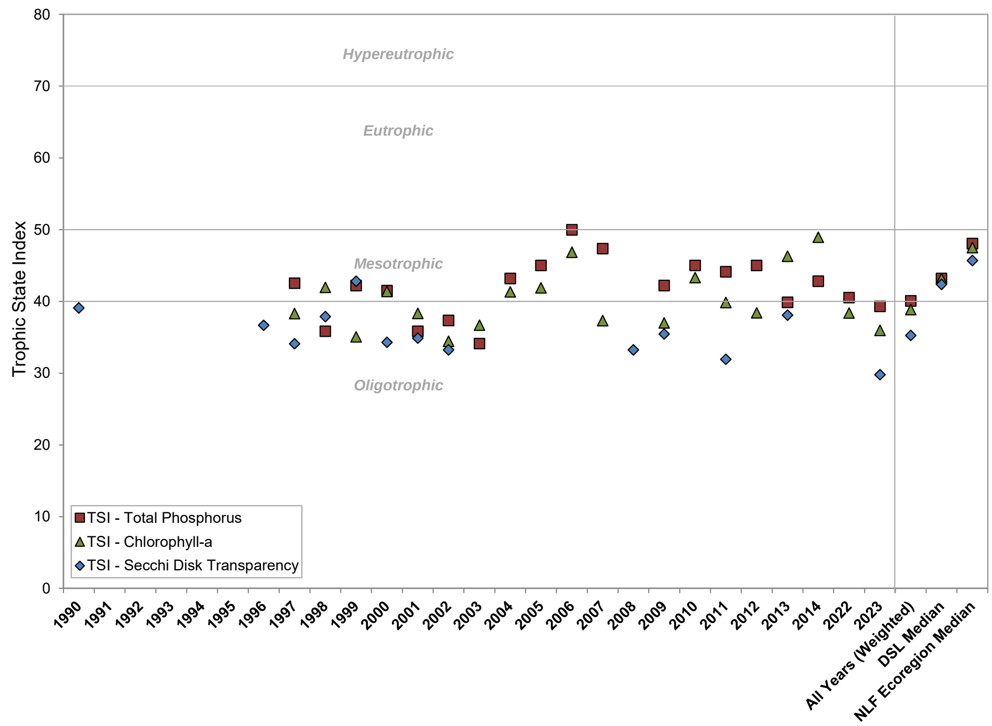


Figure 3.1-6. Forest Lake, statewide class 7 lakes, and regional Trophic State Index values. Values calculated with summer month surface sample data using WDNR PUB-WT-193.

Dissolved Oxygen and Temperature in Forest Lake

Dissolved oxygen and temperature were measured during two water quality sampling visits to Forest Lake by Onterra staff. Profiles depicting these data are displayed in Figure 3.1-7. Forest Lake was stratified during both visits, which is not unexpected due to the depth of the lake. These profiles look much the same as the profiles collected in 2016 during the development of the 2019 plan. Interestingly, the profile from August 16, 2023, shows a phenomenon called *metalimnetic oxygen maxima*, which is characterized by the mid-depth increase in dissolved oxygen in the profile. This type of profile occurs because there is a large algal community in the metalimnion. Lakes that exhibit this profile need to have good water clarity in the epilimnion so that sufficient light reaches the metalimnion to support photosynthesis. Algae thrive in this deeper water because there is sufficient light and higher amounts of nutrients, e.g. phosphorus, in these deeper waters. If the occurrence of this phenomenon disappears over time, it is an indication of declining water clarity in the lake.

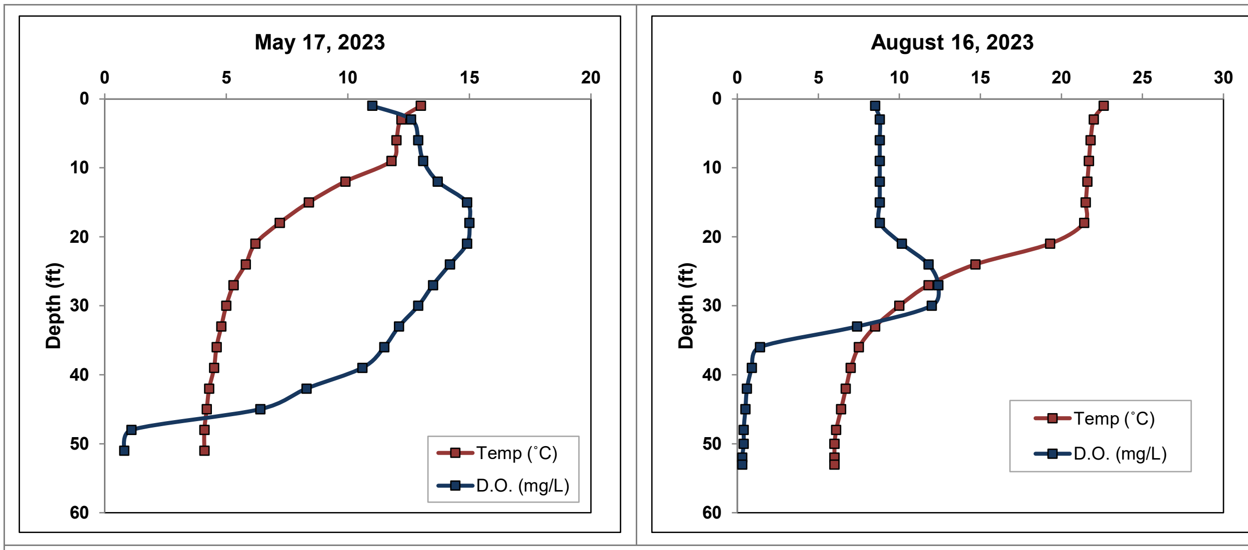


Figure 3.1-7. Forest Lake temperature and dissolved oxygen profiles. The mid-depth bulge in the August profile is called "metalimnetic oxygen maxima".

Additional Water Quality Data Collected at Forest Lake

The water quality section is centered on the trophic parameters and lake eutrophication. However, parameters other than water clarity, nutrients, and chlorophyll-*a* were collected as part of the project. These other parameters were collected to increase the understanding of Forest Lake's water quality and are recommended as a part of the WDNR long-term lake trends monitoring protocol. These parameters include pH, and alkalinity.

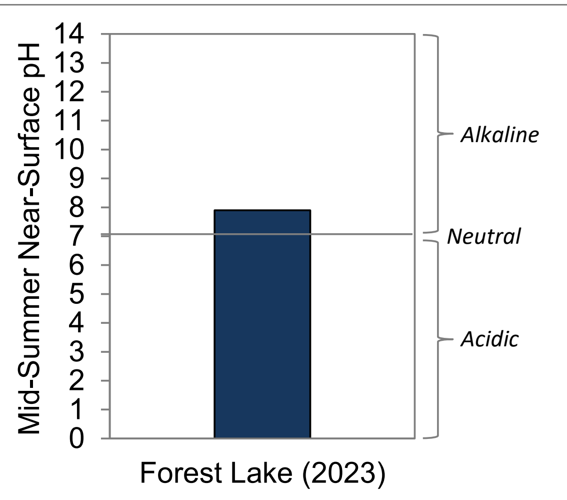


Figure 3.1-8. Forest Lake mid-summer near-surface pH value.

The pH scale ranges from 0 to 14 and indicates the concentration of hydrogen ions (H^+) within the lake's water and is an index of the lake's acidity. Water with a pH value of 7 has equal amounts of hydrogen ions and hydroxide ions (OH^-), and is considered to be neutral. Water with a pH of less than 7 has higher concentrations of hydrogen ions and is considered to be acidic, while values greater than 7 have lower hydrogen ion concentrations and are considered basic or alkaline. The pH scale is logarithmic; meaning that for every 1.0 pH unit the hydrogen ion concentration changes tenfold. The normal range for lake water pH in Wisconsin is about 5.2 to 8.4, though values lower than 5.2 can be observed in some acid bog lakes and higher than 8.4 in some marl lakes. In lakes with a pH of 6.5 and lower, the spawning of certain fish species such as walleye becomes inhibited (Shaw & Nimphius, 1985). The pH of the water in Forest Lake was found to be near neutral with a value of 8.2, and falls within the normal range for Wisconsin Lakes (Figure 3.1-8) and is similar to the value measured in 2016.

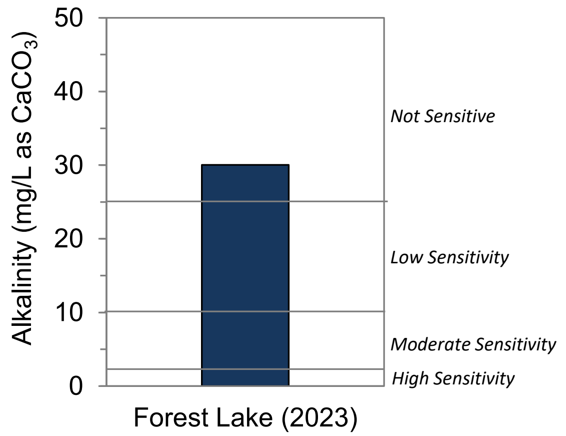


Figure 3.1-9. Forest Lake average growing season total alkalinity and sensitivity to acid rain. Samples collected from near-surface.

Alkalinity is a lake's capacity to resist fluctuations in pH by neutralizing or buffering against inputs such as acid rain. The main compounds that contribute to a lake's alkalinity in Wisconsin are bicarbonate (HCO_3^-) and carbonate (CO_3^-), which neutralize hydrogen ions from acidic inputs. These compounds are present in a lake if the groundwater entering it comes into contact with minerals such as calcite ($CaCO_3$) and/or dolomite ($CaMgCO_3)_2$). A lake's pH is primarily determined by the amount of alkalinity. Rainwater in northern Wisconsin is slightly acidic naturally due to dissolved carbon dioxide from the atmosphere with a pH of around 5.0. Consequently, lakes with low alkalinity have lower pH due to their inability to buffer against acid inputs. The alkalinity in Forest Lake was measured at an average of 30.1 (mg/L as $CaCO_3$) during the summer of 2023, indicating that the lake has a substantial capacity to resist fluctuations in pH and has a low sensitivity to acid rain (Figure 3.1-9). The 2019 management plan also determined the lake was not sensitive to acid rain as well.

A measure of water clarity once all of the suspended material (i.e., phytoplankton and sediments) have been removed, is termed *true color*, and measures how the clarity of the water is influenced by dissolved components. True color was measured at 5 SU (standard units) in May and 10 SU in August of 2023, indicating the lake's water was *clear* in 2023 (Figure 3.1-10).

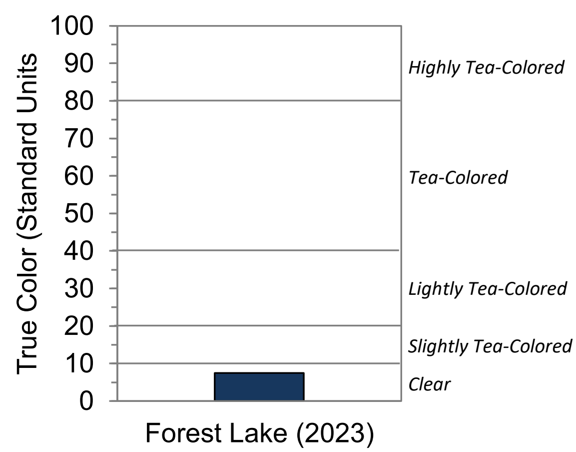


Figure 3.1-10. Forest Lake 2023 near-surface true color value.

Stakeholder Survey Responses to Forest Lake Water Quality

As discussed in section 2.0, the stakeholder survey asks many questions pertaining to perception of the lake and how it may have changed over the years. Figures 3.1-11 and 3.1-12 display the responses of members of Forest Lake stakeholders to questions regarding water quality and how it has changed over their years visiting Forest Lake.

The results of the 2023 survey are very similar to those of the 2016 survey reported on in the 2019 management plan; however, during 2016, some respondents listed the lake's water quality as being *Very Poor*, while in 2023, no one rated the lake below *Fair*. During 2023, more respondents rated the lake as *Good* and between the two surveys, about the same percentage of respondents rated the lake as *Excellent*. In 2016, slightly more respondents claimed the lake remained the same as those in 2023. In 2023, about the same percentage of respondents believe the lake's water quality *Somewhat improved* and *Somewhat degraded*, while in 2016, a similar percentage of respondents reported that the lake has *Somewhat degraded*, no one responded that it has *Somewhat improved*, and 6% believed the lake had *Severely degraded*.

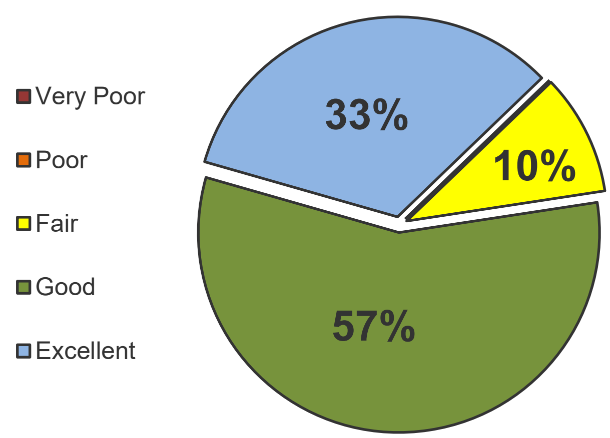


Figure 3.1-11. Stakeholder survey response Question #17. How would you describe the current water quality of Forest Lake?

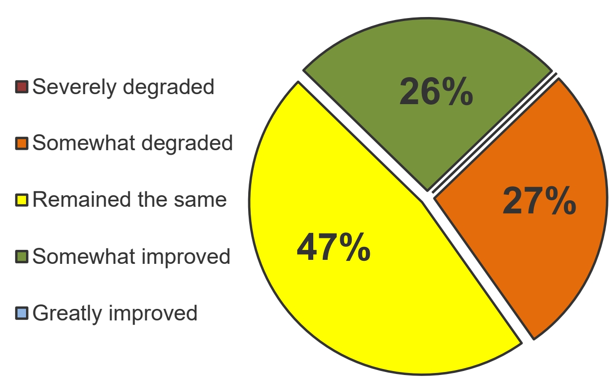


Figure 3.1-12. Stakeholder survey response Question #18. How has the water quality changed in Forest Lake since you first visited the lake?

3.2 Watershed Assessment

Watershed Modeling

A lake's **flushing rate** is simply a determination of the time required for the lake's water volume to be completely exchanged. **Residence time** describes how long a volume of water remains in the lake and is expressed in days, months, or years. The parameters are related and both determined by the volume of the lake and the amount of water entering the lake from its watershed. Greater flushing rates equal shorter residence times.

Two aspects of a lake's watershed are the key factors in determining the amount of phosphorus the watershed exports to the lake; 1) the size of the watershed, and 2) the land cover (land use) within the watershed. The impact of the watershed size is dependent on how large it is relative to the size of the lake. The watershed to lake area ratio (WS:LA) defines how many acres of watershed drains to each surface-acre of the lake. Larger ratios result in the watershed having a greater role in the lake's annual water budget and phosphorus load.

The type of land cover that exists in the watershed determines the amount of phosphorus (and sediment) that runs off the land and eventually makes its way to the lake. The actual amount of pollutants (nutrients, sediment, toxins, etc.) depends greatly on how the land within the watershed is used. Vegetated areas, such as forests, grasslands, and meadows, allow the water to permeate the ground and do not produce much surface runoff. On the other hand, agricultural areas, particularly row crops, along with residential/urban areas, minimize infiltration and increase surface runoff. The increased surface runoff associated with these land cover types leads to increased phosphorus and pollutant loading; which, in turn, can lead to nuisance algal blooms, increased sedimentation, and/or overabundant macrophyte populations. For these reasons, it is important to maintain as much natural land cover (forests, wetlands, etc.) as possible within a lake's watershed to minimize the amount runoff (nutrients, sediment, etc.) from entering the lake.

In systems with lower WS:LA ratios, land cover type plays a very important role in how much phosphorus is loaded to the lake from the watershed. In these systems, the occurrence of agriculture or urban development in even a small percentage of the watershed (less than 10%) can unnaturally elevate phosphorus inputs to the lake. If these land cover types are converted to a cover that does not export as much phosphorus, such as converting row crop areas to grass or forested areas, the phosphorus load and its impacts to the lake may be decreased. In fact, if the phosphorus load is reduced greatly, changes in lake water quality may be noticeable, (e.g. reduced algal abundance and better water clarity) and may even be enough to cause a shift in the lake's trophic state.

In systems with high WS:LA ratios, like those 10-15:1 or higher, the impact of land cover may be tempered by the sheer amount of land draining to the lake. Situations actually occur where lakes with completely forested watersheds have sufficient phosphorus loads to support high rates of plant production. In other systems with high ratios, the conversion of vast areas of row crops to vegetated areas (grasslands, meadows, forests, etc.) may not reduce phosphorus loads sufficiently to see a change in plant production. Both of these situations occur frequently in impoundments.

Regardless of the size of the watershed or the makeup of its land cover, it must be remembered that every lake is different and other factors, such as flushing rate, lake volume, sediment type, and many others, also influence how the lake will react to what is flowing into it. For instance, a deeper lake with a greater volume can dilute more phosphorus within its waters than a less voluminous lake and as a result, the production of a lake is kept low. However, in that same lake, because of its low flushing rate (a residence time of years), there may be a buildup of phosphorus in the sediments that may reach sufficient levels over time and lead to a problem such as internal nutrient loading. On the contrary, a lake with a higher flushing rate (low residence time, i.e., days or weeks) may be more productive early on, but the constant flushing of its waters may prevent a buildup of phosphorus and internal nutrient loading may never reach significant levels.

A reliable and cost-efficient method of creating a general picture of a watershed's effect on a lake can be obtained through modeling. The WDNR created a useful suite of modeling tools called the Wisconsin Lake Modeling Suite (WiLMS). Certain morphological attributes of a lake and its watershed are entered into WiLMS along with the acreages of different types of land cover within the watershed to produce useful information about the lake ecosystem. This information includes an estimate of annual phosphorus load and the partitioning of those loads between the watershed's different land cover types and atmospheric fallout entering through the lake's water surface. WiLMS also calculates the lake's flushing rate and residence times using county-specific average precipitation/evaporation values or values entered by the user. Predictive models are also included within WiLMS that are valuable in validating modeled phosphorus loads to the lake in question and modeling alternate land cover scenarios within the watershed. Finally, if specific information is available, WiLMS will also estimate the significance of internal nutrient loading within a lake and the impact of shoreland septic systems.

Forest Lake Watershed - 2019 Comprehensive Management Plan

Forest Lake is classified as a deep seepage lake and does not possess a tributary inlet or outlet. Forest Lake's total watershed encompasses approximately 1,087 acres (1.7 square miles), yielding a watershed to lake area ratio of 1:1 (Map 2). Approximately 43% of Forest Lake's watershed was composed of the lake surface, 41% of forest, 12% of wetlands, and 4% of pasture/grass (Figure 3.2-1, left frame).

According to modeling completed for the 2019 comprehensive management plan, the lake's water is completely replaced approximately once every 12 years (residence time) or 0.08 times a year (flushing rate); however, the residence time is likely shorter than estimated as Forest Lake is primarily fed by groundwater and WiLMS only uses surface runoff to estimate residence time.

It was estimated that approximately 189 pounds of phosphorus are delivered to the lake from its watershed on an annual basis. Phosphorus loading from septic systems was also estimated using data obtained from the 2016 stakeholder survey of riparian property owners. Of the estimated 189 pounds of phosphorus being delivered annually to Forest Lake, 66% is estimated to originate from direct atmospheric deposition into the lake, 19% from forest, 11% from wetlands, and 3% from riparian septic systems (Figure 3.2-1).

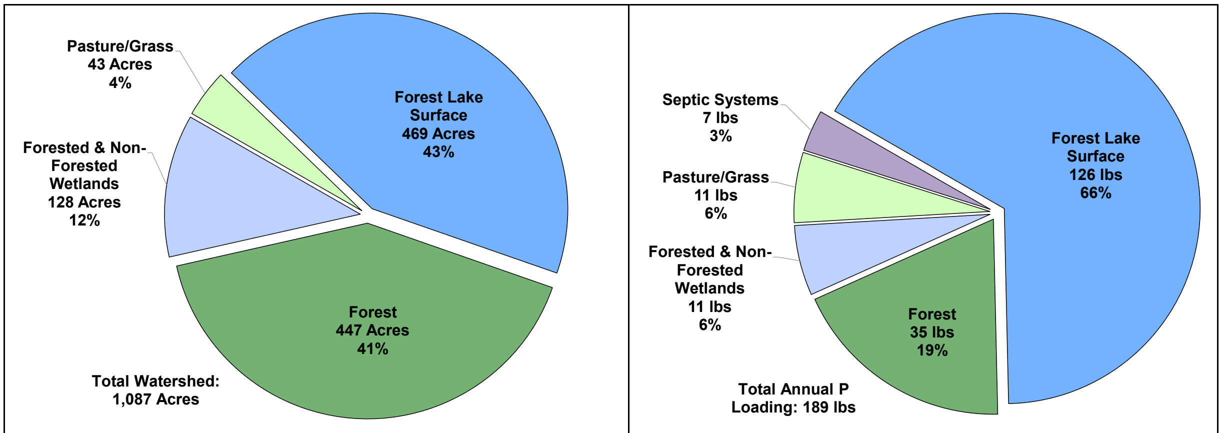


Figure 3.2-1. Forest Lake watershed land cover types in acres and phosphorus loading in pounds. Watershed land cover type based upon National Land Cover Database 2011 (NLCD – Fry et. al 2011). Phosphorus loading based upon Wisconsin Lake Modeling Suite (WiLMS).

Using predictive equations, WiLMS estimated that based on the potential annual phosphorus load, Forest Lake should have a growing season mean (GSM) total phosphorus concentration of approximately 14 µg/L. This predicted concentration is relatively similar to the measured GSM total phosphorus concentration of 15 µg/L. Which indicated that the lake's watershed and phosphorus inputs were modeled fairly accurately and the measured phosphorus concentrations in Forest Lake are near expected levels based on the lake's watershed size and land cover composition.

Forest Lake Watershed – Changes in Land Cover Between 2019 & 2024

In this project, the most current land cover information available was used to update the general description of the watershed from the 2019 plan. This land cover assessment found that there are no significant differences in the Forest Lake watershed boundary since the creation of the comprehensive management plan in 2019. However, discussion is provided in this report on how the minor changes found in the land cover type delineations may impact the lake and how these changes occurred.

The National Land Cover Database (NLCD) is a spatial reference and descriptive database of the land cover for the conterminous United States, provided by the U.S. Geological Survey (U.S. Geological Survey, 2018). While the watershed assessments were conducted in 2019 and 2024, the land cover is determined using data from NLCD 2011 and NLCD 2021, respectively. The NLCD is typically updated every three years, with the most recent land cover data available being the NLCD 2021. The land cover acreages from the NLCD 2011 and 2021 datasets are compared in Table 3.2-1.

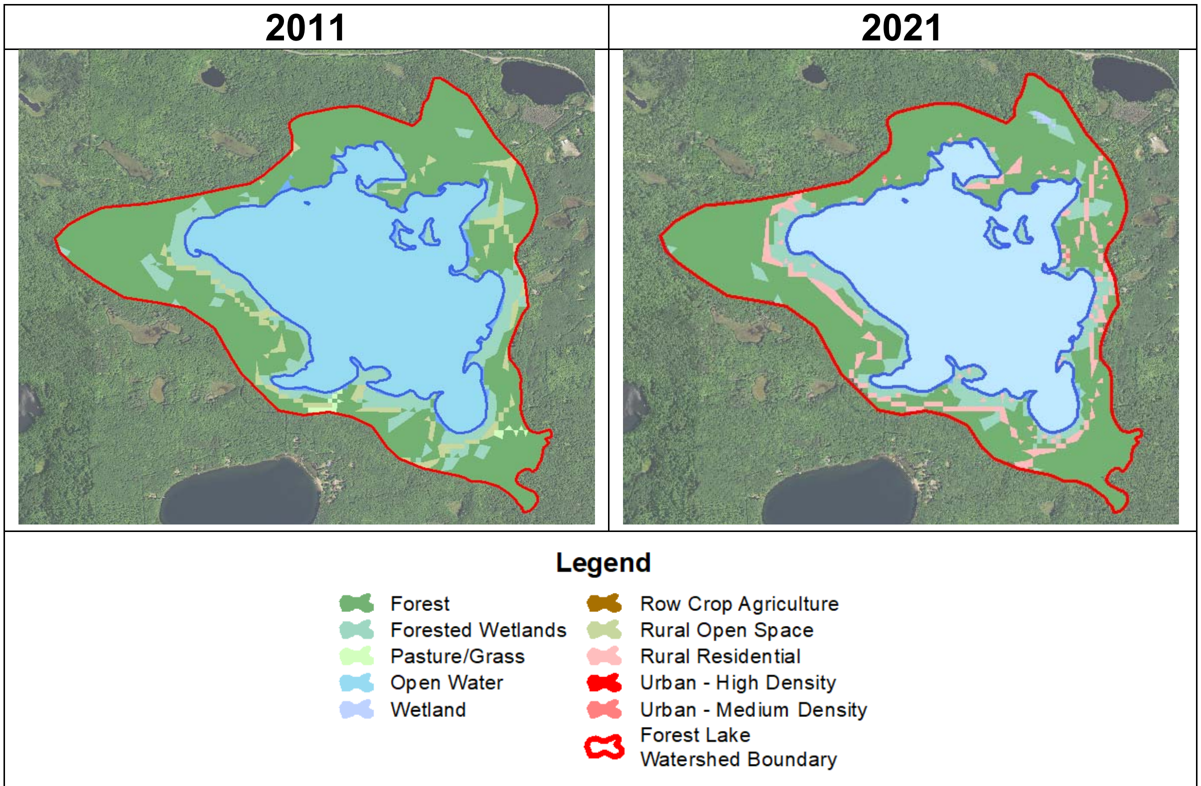


Figure 3.2-2. Comparison of the Forest Lake land cover delineation (NLCD 2011) and (NLCD 2021).

To discuss the minor differences, how they occurred, and what it means for water quality in Forest Lake, the classification and phosphorous loading themselves need be examined within the watershed. The differences in Forest Lake's watershed land cover types between the 2011 and 2021 NLCDs are largely a reflection of the increased resolution in the 2021 NLCD screening. And, ultimately, has little to no impact on the modeled phosphorous loading from the watershed.

As the NLCD is updated, land covers may be categorized differently based upon what the increased resolution of the screening can detect. For example, in the Forest Lake watershed, much of the rural open space in 2011 was reclassified as rural residential in the 2021 NLCD. That, and some of the wetland from 2011 is now rural residential (Table 3.2-1). It is unlikely that wetlands were developed in the decade between the datasets. Further examination would need to be completed to determine if rural open space was developed to rural residential during that time span. However, as mentioned above, these slight reclassifications do not make a difference in the output of the phosphorus load modeling.

Table 3.2-1. Forest Lake of land cover acreage from the NLCD 2011 and NLCD 2021

Landcover Classification	2011 Acres	2021 Acres	Percent Change
Forest	447.2	450.0	1%
Wetland	125.5	100.3	-20%
Open Water	0.0	0.0	No Change
Forest Lake	471.1	471.4	No Change
Rural Residential	0.0	56.7	100%*
Pasture Grass	42.9	5.3	-88%
Urban - High Density	0.0	0.0	No Change
Ubran - Medium Density	0.0	0.4	No Change
Row Crops	0.0	0.0	No Change
Total Acreage	**1087**	**1084**	

* Percent change of increase cannot be divided by zero. How ever, since the acerage is more than zero, it is counted as a 100% increase from the original for purposes of this analysis.

The WiLMs phosphorous loading estimate for rural residential is slightly lower than that of pasture grass, meaning that the phosphorous loading for the 2021 delineation would only be slightly less than the 2011. Because a large majority of phosphorous loading is contributed from the surrounding forest (19%) and the lake itself (66%), and only 6% from pasture grass (Figure 3.2-1, Right), the changes made by the reclassification of pasture and grass is insignificant.

3.3 Aquatic Plants

Introduction

Although the occasional lake user may consider aquatic macrophytes to be "weeds" and a nuisance to the recreational use of the lake, the plants are actually an essential element in a healthy and functioning lake ecosystem. It is very important that lake stakeholders understand the importance of lake plants and the many functions they serve in maintaining and protecting a lake ecosystem. With increased understanding and awareness, most lake users will recognize the importance of the aquatic plant community and their potential negative effects on it.

Photograph 3.3-1. Example of emergent and floating-leaf communities.

Diverse aquatic vegetation provides habitat and food for many kinds of aquatic life, including fish, insects, amphibians, waterfowl, and even terrestrial wildlife. For instance, wild celery (*Vallisneria americana)* and wild rice (*Zizania aquatica* and *Z. palustris*) both serve as excellent food sources for ducks and geese. Emergent stands of vegetation provide necessary spawning habitat for fish such as northern pike (*Esox lucius*) and yellow perch (*Perca flavescens*). In addition, many of the insects that are eaten by young fish rely heavily on aquatic plants and the periphyton attached to them as their primary food source. The plants also provide cover for feeder fish and zooplankton, stabilizing the predator-prey relationships within the system. Furthermore, rooted aquatic plants prevent shoreland erosion and the resuspension of sediments and nutrients by absorbing wave energy and locking sediments within their root masses. In areas where plants do not exist, waves can resuspend bottom sediments decreasing water clarity and increasing plant nutrient levels that may lead to algae blooms. Lake plants also produce oxygen through photosynthesis and use nutrients that may otherwise be used by phytoplankton, which helps to minimize nuisance algal blooms.

Under certain conditions, a few species may become a problem and require control measures. Excessive plant growth can limit recreational use by deterring navigation, swimming, and fishing activities. It can also lead to changes in fish population structure by providing too much cover for feeder fish resulting in reduced predation by predator fish, which could result in a stunted pan-fish population. Exotic plant species, such as Eurasian watermilfoil (*Myriophyllum spicatum*) and curly-leaf pondweed (*Potamogeton crispus*) can also upset the delicate balance of a lake ecosystem by out competing native plants and reducing species diversity. These species will be discussed further in depth in the Aquatic Invasive Species section. These invasive plant species can form dense stands that are a nuisance to humans and provide low-value habitat for fish and other wildlife.

When plant abundance negatively affects the lake ecosystem and limits the use of the resource, plant management and control may be necessary. The management goals should always include the control of invasive species and restoration of native communities through environmentally sensitive and economically feasible methods. No aquatic plant management plan should only contain methods to control plants, they should also contain methods on how to protect and possibly enhance the important plant communities within the lake. Unfortunately, the latter is often neglected and the ecosystem suffers as a result.

Aquatic Plant Management and Protection

Many times, an aquatic plant management plan is aimed at only controlling nuisance plant growth that has limited the recreational use of the lake, usually navigation, fishing, and swimming. It is important to remember the vital benefits that native aquatic plants provide to lake users and the lake ecosystem, as described above. Therefore, all aquatic plant management plans also need to address the enhancement and protection of the aquatic plant community. Below are general descriptions of the many techniques that can be utilized to control and enhance aquatic plants. Each alternative has benefits and limitations that are explained in its description. Please note that only legal and commonly used methods are included. For instance, the herbivorous grass carp (*Ctenopharyngodon idella*) is illegal in Wisconsin and rotovation, a process by which the lake bottom is tilled, is not a commonly accepted practice. Unfortunately, there are no "silver bullets" that can completely cure all aquatic plant problems, which makes planning a crucial step in any aquatic plant management activity. Many of the plant management and protection techniques commonly used in Wisconsin are described below.

Important Note:
Even though most of these techniques are not applicable to Forest Lake, it is still important for lake users to have a basic understanding of all the techniques so they can better understand why particular methods are or are not applicable in their lake. The techniques applicable to Forest Lake are discussed in Summary and Conclusions section and the Implementation Plan found near the end of this document.

Permits

The signing of the 2001-2003 State Budget by Gov. McCallum enacted many aquatic plant management regulations. The rules for the regulations have been set forth by the WDNR as NR 107 and 109. A major change includes that all forms of aquatic plant management, even those that did not require a permit in the past, require a permit now, including manual and mechanical removal. Manual cutting and raking are exempt from the permit requirement if the area of plant removal is no more than 30 feet wide and any piers, boatlifts, swim rafts, and other recreational and water use devices are located within those 30 feet. This action can be conducted up to 150 feet from shore. Please note that a permit is needed in all instances if wild rice is to be removed. Furthermore, installation of aquatic plants, even natives, requires approval from the WDNR.

Permits are required for chemical and mechanical manipulation of native and non-native plant communities. Large-scale protocols have been established for chemical treatment projects covering >10 acres or areas greater than 10% of the lake littoral zone and more than 150 feet from shore. Different protocols are to be followed for whole-lake scale treatments (≥160 acres or ≥50% of the lake littoral area). Additionally, it is important to note that local permits and U.S. Army Corps of Engineers regulations may also apply. For more information on permit requirements, please contact the WDNR Regional Water Management Specialist or Aquatic Plant Management and Protection Specialist.

Manual Removal (Hand-Harvesting & DASH)

Manual removal methods include hand-pulling, raking, and hand-cutting. Hand-pulling involves the manual removal of whole plants, including roots, from the area of concern and disposing them out of the waterbody. Raking entails the removal of partial and whole plants from the lake by dragging a rake with a rope tied to it through plant beds. Specially designed rakes are available from commercial sources or an asphalt rake can be used. Hand-cutting differs from the other two manual methods because the entire plant is not removed, rather the plants are cut similar to mowing a lawn; however, Wisconsin law states that all plant fragments must be removed.

Photograph 3.3-2. Example of aquatic plants that have been removed manually.

Manual removal or hand-harvesting of aquatic invasive species has gained favor in recent years as an alternative to herbicide control programs. 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.

Cost

Contracting aquatic invasive species removal by third-party firm can cost approximately $1,500+ per day for traditional hand-harvesting methods whereas the costs can be closer to $2,500 when DASH technology is used. Additional disposal, travel, and permitting fees may also apply.

Advantages	*Disadvantages*
• Very cost effective for clearing areas around docks, piers, and swimming areas. • Relatively environmentally safe if treatment is conducted after June 15th. • Allows for selective removal of undesirable plant species. • Provides immediate relief in localized area. • Plant biomass is removed from waterbody.	• Labor intensive. • Impractical for larger areas or dense plant beds. • Subsequent treatments may be needed as plants recolonize and/or continue to grow. • Uprooting of plants stirs bottom sediments making it difficult to conduct action. • May disturb benthic organisms and fish-spawning areas. • Risk of spreading invasive species if fragments are not removed.

Bottom Screens

Bottom screens are very much like landscaping fabric used to block weed growth in flowerbeds. The gas-permeable screen is placed over the plant bed and anchored to the lake bottom by staking or weights. Only gas-permeable screen can be used or large pockets of gas will form under the mat as the result of plant decomposition. This could lead to portions of the screen becoming detached from the lake bottom, creating a navigational hazard. Normally the screens are removed and cleaned at the end of the growing season and then placed back in the lake the following spring. If they are not removed, sediments may build up on them and allow for plant colonization on top of the screen. Please note that depending on the size of the screen a Wisconsin Department of Natural Resources permit may be required.

Cost

Material costs range between $.20 and $1.25 per square-foot. Installation cost can vary largely, but may roughly cost $750 to have 1,000 square feet of bottom screen installed. Maintenance costs can also vary, but an estimate for a waterfront lot is about $120 each year.

Advantages	*Disadvantages*
• Immediate and sustainable control. • Long-term costs are low. • Excellent for small areas and around obstructions. • Materials are reusable. • Prevents fragmentation and subsequent spread of plants to other areas.	• Installation may be difficult over dense plant beds and in deep water. • Not species specific. • Disrupts benthic fauna. • May be navigational hazard in shallow water. • Initial costs are high. • Labor intensive due to the seasonal removal and reinstallation requirements. • Does not remove plant biomass from lake. • Not practical in large-scale situations.

Water Level Drawdown

The primary manner of plant control through water level drawdown is the exposure of sediments and plant roots/tubers to desiccation and either heating or freezing depending on the timing of the treatment. Winter drawdowns are more common in temperate climates like that of Wisconsin and usually occur in reservoirs because of the ease of water removal through the outlet structure. An important fact to remember when considering the use of this technique is that only certain species are controlled and that some species may even be enhanced. Furthermore, the process will likely need to be repeated every two or three years to keep target species in check.

Cost

The cost of this alternative is highly variable. If an outlet structure exists, the cost of lowering the water level would be minimal; however, if there is not an outlet, the cost of pumping water to the desirable level could be very expensive. If a hydro-electric facility is operating on the system, the costs associated with loss of production during the drawdown also need to be considered, as they are likely cost prohibitive to conducting the management action.

Advantages	*Disadvantages*
• Inexpensive if outlet structure exists. • May control populations of certain species, like Eurasian watermilfoil for a few years. • Allows some loose sediment to consolidate, increasing water depth. • May enhance growth of desirable emergent species. • Other work, like dock and pier repair may be completed more easily and at a lower cost while water levels are down.	• May be cost prohibitive if pumping is required to lower water levels. • Has the potential to upset the lake ecosystem and have significant effects on fish and other aquatic wildlife. • Adjacent wetlands may be altered due to lower water levels. • Disrupts recreational, hydroelectric, irrigation and water supply uses. • May enhance the spread of certain undesirable species, like common reed and reed canary grass. • Permitting process may require an environmental assessment that may take months to prepare. • Non-selective.

Mechanical Harvesting

Photograph 3.3-3. Mechanical harvester.

Aquatic plant harvesting is frequently used in Wisconsin and involves the cutting and removal of plants much like mowing and bagging a lawn. Harvesters are produced in many sizes that can cut to depths ranging from 3 to 6 feet with cutting widths of 4 to 10 feet. Plant harvesting speeds vary with the size of the harvester, density and types of plants, and the distance to the off-loading area. Equipment requirements do not end with the harvester. In addition to the harvester, a shore-conveyor would be required to transfer plant material from the harvester to a dump truck for transport to a landfill or compost site. Furthermore, if off-loading sites are limited and/or the lake is large, a transport barge may be needed to move the harvested plants from the harvester to the shore in order to cut back on the time that the harvester spends traveling to the shore conveyor. Some lake organizations contract to have nuisance plants harvested, while others choose to purchase their own equipment. If the latter route is chosen, it is especially important for the lake group to be very organized and realize that there is a great deal of work and expense involved with the purchase, operation, maintenance, and storage of an aquatic plant harvester. In either case, planning is very important to minimize environmental effects and maximize benefits.

Cost

Equipment costs vary with the size and features of the harvester, but in general, standard harvesters range between $45,000 and $100,000. Larger harvesters or stainless-steel models may cost as much as $200,000. Shore conveyors cost approximately $20,000 and trailers range from $7,000 to $20,000. Storage, maintenance, insurance, and operator salaries vary greatly.

Advantages	***Disadvantages***
• Immediate results. • Plant biomass and associated nutrients are removed from the lake. • Select areas can be treated, leaving sensitive areas intact. • Plants are not completely removed and can still provide some habitat benefits. • Opening of cruise lanes can increase predator pressure and reduce stunted fish populations. • Removal of plant biomass can improve the oxygen balance in the littoral zone. • Harvested plant materials produce excellent compost.	• Initial costs and maintenance are high if the lake organization intends to own and operate the equipment. • Multiple treatments are likely required. • Many small fish, amphibians and invertebrates may be harvested along with plants. • There is little or no reduction in plant density with harvesting. • Invasive and exotic species may spread because of plant fragmentation associated with harvester operation. • Bottom sediments may be re-suspended leading to increased turbidity and water column nutrient levels.

Herbicide Treatment

Photograph 3.3-4. Liquid herbicide application. Photo credit: Amy Kay, Clarke.

The use of herbicides to control aquatic plants and algae is a technique that is widely used by lake managers. Traditionally, herbicides were used to control nuisance levels of aquatic plants and algae that interfere with navigation and recreation. While this practice still takes place in many parts of Wisconsin, the use of herbicides to control aquatic invasive species is becoming more prevalent. Resource managers employ strategic management techniques towards aquatic invasive species, with the objective of reducing the target plant's population over time; and an overarching goal of attaining long-term ecological restoration. For submergent vegetation, this largely consists of implementing control strategies early in the growing season; either as spatially-targeted, small-scale spot treatments or low-dose, large-scale (whole lake) treatments. Treatments occurring roughly each year before June 1 and/or when water temperatures are below 60°F can be less impactful to many native plants, which have not emerged yet at this time of year. Emergent species are targeted with foliar applications at strategic times of the year when the target plant is more likely to absorb the herbicide.

While there are approximately 300 herbicides registered for terrestrial use in the United States, only 13 active ingredients can be applied into or near aquatic systems. All aquatic herbicides must be applied in accordance with the product's US Environmental Protection Agency (EPA) approved label. There are numerous formulations and brands of aquatic herbicides and an extensive list can be found in Appendix F of (Gettys, 2009).

Applying herbicides in the aquatic environment requires special considerations compared with terrestrial applications. WDNR administrative code states that a permit is required if, "you are standing in socks and they get wet." In these situations, the herbicide application needs to be completed by an applicator licensed with the Wisconsin Department of Agriculture, Trade and Consumer Protection. All herbicide applications conducted under the ordinary high-water mark require herbicides specifically labeled by the United States Environmental Protection Agency

Aquatic herbicides can be classified in many ways. Organization of this section follows Netherland (2009) in which mode of action (i.e., how the herbicide works) and application techniques (i.e., foliar or submersed treatment) group the aquatic herbicides. The table below provides a general list of commonly used aquatic herbicides in Wisconsin and is synthesized from (Netherland, 2009).

The arguably clearest division amongst aquatic herbicides is their general mode of action and fall into two basic categories:

1. Contact herbicides act by causing extensive cellular damage, but usually do not affect the areas that were not in contact with the chemical. This allows them to work much faster, but in some plants does not result in a sustained effect because the root crowns, roots, or rhizomes are not killed.
2. Systemic herbicides act slower than contact herbicides, being transported throughout the entire plant and disrupting biochemical pathways which often result in complete mortality.

Table 3.3-1. Common herbicides used for aquatic plant management.

	General Mode of Action	Compound	Specific Mode of Action	Most Common Target Species in Wisconsin
Contact		Copper	plant cell toxicant	Algae, including macro-algae (i.e. muskgrasses & stoneworts)
		Endothall	Inhibits respiration & protein synthesis	Submersed species, largely for curly-leaf pondweed; invasive watermilfoil control when mixed with auxin herbicides
		Diquat	Inhibits photosynthesis & destroys cell membranes	Nusiance species including duckweeds, targeted AIS control when exposure times are low
		Flumioxazin	Inhibits photosynthesis & destroys cell membranes	Nusiance species, targeted AIS control when exposure times are low
Systemic	Auxin Mimics	2,4-D	auxin mimic, plant growth regulator	Submersed species, largely for invasive watermilfoil
		Triclopyr	auxin mimic, plant growth regulator	Submersed species, largely for invasive watermilfoil
		Florpyrauxifen-benzyl	arylpicolinate auxin mimic, growth regulator, different binding afinity than 2,4-D or triclopyr	Submersed species, largely for invasive watermilfoil
	In Water Use Only	Fluridone	Inhibits plant specific enzyme, new growth bleached	Submersed species, largely for invasive watermilfoil
	Enzyme Specific (ALS)	Penoxsulam	Inhibits plant-specific enzyme (ALS), new growth stunted	Emergent species with potential for submergent and floating-leaf species
		Imazamox	Inhibits plant-specific enzyme (ALS), new growth stunted	New to WI, potential for submergent and floating-leaf species
	Enzyme Specific (foliar use only)	Glyphosate	Inhibits plant-specific enzyme (ALS)	Emergent species, including purple loosestrife
		Imazapyr	Inhibits plant-specific enzyme (EPSP)	Hardy emergent species, including common reed

Both types are commonly used throughout Wisconsin with varying degrees of success. The use of herbicides is potentially hazardous to both the applicator and the environment, so all lake organizations should seek consultation and/or services from professional applicators with training and experience in aquatic herbicide use.

Herbicides that target submersed plant species are directly applied to the water, either as a liquid or an encapsulated granular formulation. Factors such as water depth, water flow, treatment area size, and plant density work to reduce herbicide concentration within aquatic systems. Understanding concentration and exposure times are important considerations for aquatic herbicides. Successful control of the target plant is achieved when it is exposed to a lethal concentration of the herbicide for a specific duration of time. Much information has been gathered in recent years, largely as a result of an ongoing cooperative research project between the Wisconsin Department of Natural Resources, US Army Corps of Engineers Research and Development Center, and private consultants (including Onterra). This research couples quantitative aquatic plant monitoring with field-collected herbicide concentration data to evaluate efficacy and selectivity of control strategies implemented on a subset of Wisconsin lakes and flowages. Based on their preliminary findings, lake managers have adopted two main treatment strategies: 1) whole-lake treatments, and 2) spot treatments.

Spot treatments are a type of control strategy where the herbicide is applied to a specific area (treatment site) such that when it dilutes from that area, its concentrations are insufficient to cause significant affects outside of that area. Spot treatments typically rely on a short exposure time (often hours) to cause mortality and therefore are applied at a much higher herbicide concentration than whole-lake treatments. This has been the strategy historically used on most Wisconsin systems.

Whole-lake treatments are those where the herbicide is applied to specific sites, but when the herbicide reaches equilibrium within the entire volume of water (entire lake, lake basin, or within the epilimnion of the lake or lake basin); it is at a concentration that is sufficient to cause mortality to the target plant within that entire lake or basin. The application rate of a whole-lake treatment is dictated by the volume of water in which the herbicide will reach equilibrium. Because exposure time is so much longer, target herbicide levels for whole-lake treatments are significantly less than for spot treatments.

Cost

Herbicide application charges vary greatly between $400 and $1,500 per acre depending on the chemical used, who applies it, permitting procedures, and the size/depth of the treatment area.

Advantages	***Disadvantages***
• Herbicides are easily applied in restricted areas, like around docks and boatlifts. • Herbicides can target large areas all at once. • If certain chemicals are applied at the correct dosages and at the right time of year, they can selectively control certain	• All herbicide use carries some degree of human health and ecological risk due to toxicity. • Fast-acting herbicides may cause fish kills due to rapid plant decomposition if not applied correctly.

invasive species, such as Eurasian watermilfoil. • Some herbicides can be used effectively in spot treatments. • Most herbicides are designed to target plant physiology and in general, have low toxicological effects on non-plant organisms (e.g., mammals, insects)	• Many people adamantly object to the use of herbicides in the aquatic environment; therefore, all stakeholders should be included in the decision to use them. • Many aquatic herbicides are nonselective. • Some herbicides have a combination of use restrictions that must be followed after their application. • Overuse of same herbicide may lead to plant resistance to that herbicide.

Biological Controls

There are many insects, fish and pathogens within the United States that are used as biological controls for aquatic macrophytes. For instance, the herbivorous grass carp has been used for years in many states to control aquatic plants with some success and some failures. However, it is illegal to possess grass carp within Wisconsin because their use can create problems worse than the plants that they were used to control. Other states have also used insects to battle invasive plants, such as water hyacinth weevils (*Neochetina spp.*) and hydrilla stem weevil (*Bagous spp.*) to control water hyacinth (*Eichhornia crassipes*) and hydrilla (*Hydrilla verticillata)*, respectively.

However, Wisconsin, along with many other states, is currently experiencing the expansion of lakes infested with Eurasian watermilfoil and as a result has supported the experimentation and use of the milfoil weevil (*Euhrychiopsis lecontei*) within its lakes. The milfoil weevil is a native weevil that has shown promise in reducing Eurasian watermilfoil stands in Wisconsin, Washington, Vermont, and other states. Research is currently being conducted to discover the best situations for the use of the insect in battling Eurasian watermilfoil. Milfoil weevils are not currently available for purchase. Some lake groups have investigated rearing weevils on their own. Groups may measure for weevil population density in the lake or document weevil herbivory impacts to EWM. A manual that is authored by Golden Sands RC&D, and is referenced by WDNR, explains weevil biocontrol considerations for Wisconsin Lakes (Golden Sands, 2017).

Cost

Stocking with adult weevils costs about $1.20/weevil and they are usually stocked in lots of 1000 or more.

Advantages	***Disadvantages***
• Milfoil weevils occur naturally in Wisconsin. • Likely environmentally safe and little risk of unintended consequences.	• Stocking and monitoring costs are high. • This is an unproven and experimental treatment. • There is a chance that a large amount of money could be spent with little or no change in Eurasian watermilfoil density.

Wisconsin has approved the use of two species of leaf-eating beetles (*Galerucella calmariensis* and *G. pusilla*) to battle purple loosestrife. These beetles were imported from Europe and used as a biological control method for purple loosestrife. Many cooperators, such as county conservation departments or local UW-Extension locations, currently support large beetle rearing operations. Beetles are reared on live purple loosestrife plants growing in kiddy pools surrounded by insect netting. Beetles are collected with aspirators and then released onto the target wild population. For more information on beetle rearing, contact your local UW-Extension location.

In some instances, beetles may be collected from known locations (cella insectaries) or purchased through private sellers. Although no permits are required to purchase or release beetles within Wisconsin, application/authorization and release forms are required by the WDNR for tracking and monitoring purposes.

Cost

The cost of beetle release is very inexpensive, and in many cases is free.

Advantages	***Disadvantages***
• Extremely inexpensive control method. • Once released, considerably less effort than other control methods is required. • Augmenting populations may lead to long-term control.	• Although considered "safe," reservations about introducing one non-native species to control another exist. • Long range studies have not been completed on this technique.

Analysis of Current Aquatic Plant Data

Aquatic plants are an important element in every healthy lake. Changes in lake ecosystems are often first seen in the lake's plant community. Whether these changes are positive, such as variable water levels or negative, such as increased shoreland development or the introduction of an exotic species, the plant community will respond. Plant communities respond in a variety of ways. For example, there may be a loss of one or more species. Certain life forms, such as emergent or floating-leaf communities, may disappear from specific areas of the lake. A shift in plant dominance between species may also occur. With periodic monitoring and proper analysis, these changes are relatively easy to detect and provide very useful information for management decisions.

Multiple aquatic plant surveys were completed on Forest Lake; the first looked strictly for the exotic plant curly-leaf pondweed, while the others that followed assessed both native and non-native species. Combined, these surveys produce a great deal of information about the aquatic vegetation of the lake. These data are analyzed and presented in numerous ways; each is discussed in more detail below.

Primer on Data Analysis & Data Interpretation

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 Forest Lake. 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

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 Forest 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. 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 Forest Lake to be compared to other lakes within the region and state.

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

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. A lake with a diverse plant community is also better suited to compete against exotic infestations than a lake with a lower diversity. 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 and
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 Forest 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.

Forest Lake Aquatic Plant Survey Results

The point intercept survey has occurred on Forest Lake in 2005, 2013, 2016, 2019, and 2022. Emergent and floating-leaf plant communities were specifically mapped in a 2016 survey which documents several additional species growing around the margins of Forest Lake. Approximately 57 species of plants have been documented in aquatic plant surveys in Forest Lake (Table 3.3-2).

Table 3.3-2. Aquatic plant species located during aquatic plant surveys.

Growth Form	Scientific Name	Common Name	Status in Wisconsin	Coefficient of Conservatism	2005	2013	2016	2019	2022
Emergent	*Bolboschoenus fluviatilis*	River bulrush	Native	5			I		
	Carex comosa	Bristly sedge	Native	5			I		
	Carex lasiocarpa	Narrow-leaved woolly sedge	Native	9			I		
	Carex pseudocyperus	Cypress-like sedge	Native	8			I		
	Carex stricta	Common tussock sedge	Native	7			I		
	Carex utriculata	Common yellow lake sedge	Native	7			I		
	Dulichium arundinaceum	Three-way sedge	Native	9			X		
	Eleocharis palustris	Creeping spikerush	Native	6		X	X		
	Glyceria borealis	Northern manna grass	Native	8			I		
	Glyceria canadensis	Rattlesnake grass	Native	7			I		
	Iris versicolor	Northern blue flag	Native	5			I		
	Sagittaria latifolia	Common arrowhead	Native	3			I		
	Sagittaria rigida	Stiff arrowhead	Native	8		X	I		
	Schoenoplectus tabernaemontani	Softstem bulrush	Native	4		X	X		
	Scirpus pedicellatus	Stalked woolgrass	Native	6			I		
	Sparganium americanum	American bur-reed	Native	8			X		
	Sparganium androcladum	Shining bur-reed	Native	8			X		
	Typha spp.	Cattail spp.	Unknown (Sterile)	N/A		X			
FL	*Brasenia schreberi*	Watershield	Native	7			X		X
	Nuphar variegata	Spatterdock	Native	6			I		X
	Nymphaea odorata	White water lily	Native	6	X	X	X	X	X
	Persicaria amphibia	Water smartweed	Native	5			X		
	Sparganium angustifolium	Narrow-leaf bur-reed	Native	9	X		X	X	
Submergent	*Bidens beckii*	Water marigold	Native	8	X	X	X	X	X
	Chara spp.	Muskgrasses	Native	7	X	X	X	X	X
	Elodea canadensis	Common waterweed	Native	3	X	X	X	X	X
	Elodea nuttallii	Slender waterweed	Native	7	X			X	X
	Heteranthera dubia	Water stargrass	Native	6		X	X	X	X
	Isoetes spp.	Quillwort spp.	Native	8	X	X	I		X
	Myriophyllum sibiricum	Northern watermilfoil	Native	7	X	X	X	X	X
	Myriophyllum spicatum	Eurasian watermilfoil	Non-Native - Invasive	N/A		X		X	X
	Najas flexilis	Slender naiad	Native	6	X	X	X	X	X
	Najas guadalupensis	Southern naiad	Native	7					X
	Nitella spp.	Stoneworts	Native	7		X		X	X
	Potamogeton alpinus	Alpine pondweed	Native	9		X	I		
	Potamogeton amplifolius	Large-leaf pondweed	Native	7	X	X	X	X	X
	Potamogeton amplifolius X P. illinoensis	Large-leaf X illinois pondweed	Native	N/A		X			
	Potamogeton berchtoldii	Slender pondweed	Native	7			X	X	X
	Potamogeton foliosus	Leafy pondweed	Native	6			I		X
	Potamogeton gramineus	Variable-leaf pondweed	Native	7	X	X	X	X	X
	Potamogeton illinoensis	Illinois pondweed	Native	6		X	I	X	X
	Potamogeton natans	Floating-leaf pondweed	Native	5			I	X	
	Potamogeton obtusifolius	Blunt-leaved pondweed	Native	9				X	X
	Potamogeton praelongus	White-stem pondweed	Native	8			X	X	X
	Potamogeton praelongus X P. richardsonii	White-stem X clasping-leaf pondweed	Native	N/A			X		
	Potamogeton pusillus	Small pondweed	Native	7	X	X	X	X	X
	Potamogeton richardsonii	Clasping-leaf pondweed	Native	5				X	
	Potamogeton richardsonii X P. gramineus	Clasping-leaf X variable-leaf pondweed	Native	N/A	X				
	Potamogeton robbinsii	Fern-leaf pondweed	Native	8	X	X	X	X	X
	Potamogeton spirillus	Spiral-fruited pondweed	Native	8	X	X	X	X	X
	Potamogeton zosteriformis	Flat-stem pondweed	Native	6	X	X	X	X	X
	Sagittaria sp. (rosette)	Arrowhead sp. (rosette)	Native	N/A			X		
	Utricularia vulgaris	Common bladderwort	Native	7			I	X	X
	Vallisneria americana	Wild celery	Native	6		X	X	X	X
S/E	*Eleocharis acicularis*	Needle spikerush	Native	5	X	X	X	X	X
	Sagittaria graminea	Grass-leaved arrowhead	Native	9		X			
FF	*Lemna minor*	Lesser duckweed	Native	5				I	

X = Located on rake during point-intercept survey; I = Incidentally located; not located on rake during point-intercept survey
FL = Floating-leaf; F/L = Floating-leaf & Emergent; S/E = Submergent and/or Emergent; FF = Free-floating

The sediment within littoral areas of Forest Lake is conducive for supporting lush aquatic plant growth within its large bays. Data from the point-intercept survey indicates that approximately 50% of the sampling locations located within the littoral zone contained fine organic sediment (muck), 34% contained sand, and 16% contained rock (Figure 3.3-1).

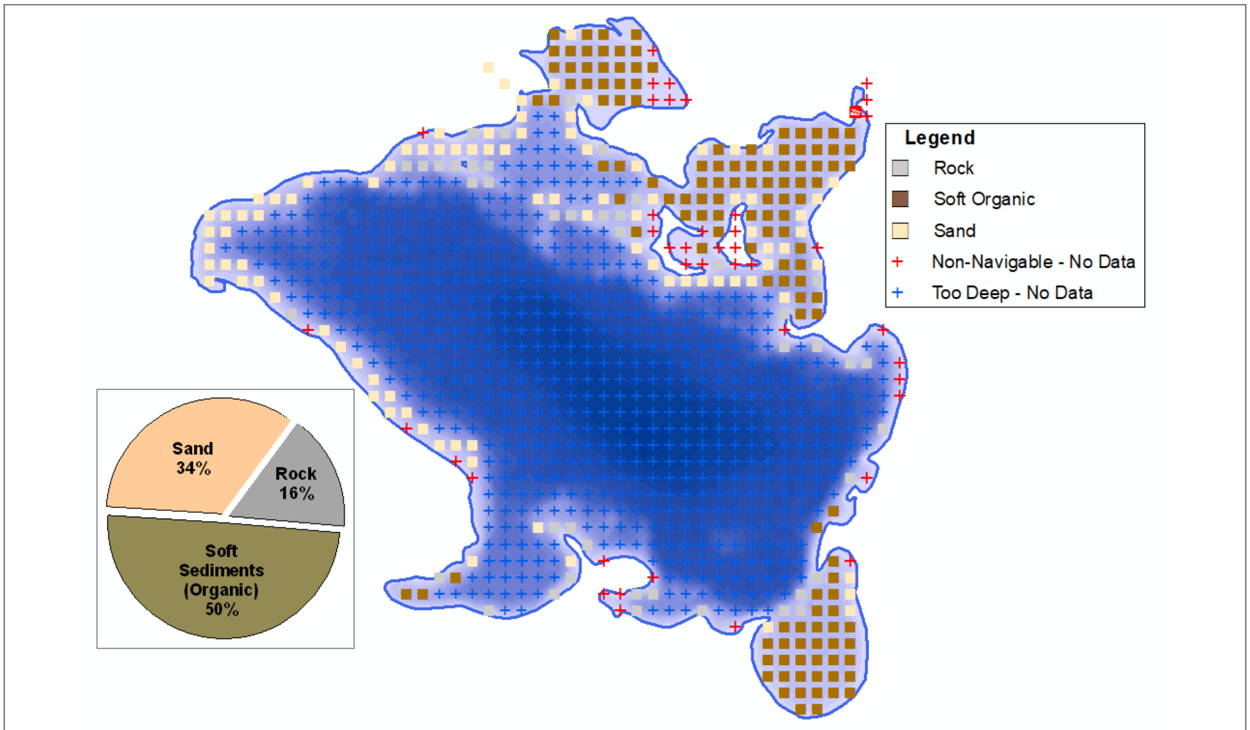


Figure 3.3-1. Proportion of substrate types within littoral areas. Created using data from 2022 point-intercept survey.

A total of 28 aquatic plant species were encountered directly on the rake during the 2022 whole-lake point-intercept survey with fern-leaf pondweed (*Potamogeton robbinsii*, 37.9%), common waterweed (*Elodea canadensis*, 20.3%), muskgrasses (*Chara spp.* 9.0%), and northern watermilfoil (*Myriophyllum sibiricum*, 9.0%) being the most frequently encountered species (Figure 3.3-2). Eurasian watermilfoil was not encountered during the survey resulting in an occurrence of 0%.

Fern-leaf pondweed (*Potamogeton robbinsii*)	Common & slender waterweed (*Elodea canadensis & E. nuttalli*)	Muskgrass (*Charra spp.*)
		

Photograph 3.3-5. Common plant species found during the 2022 surveys. Photo credit Onterra.

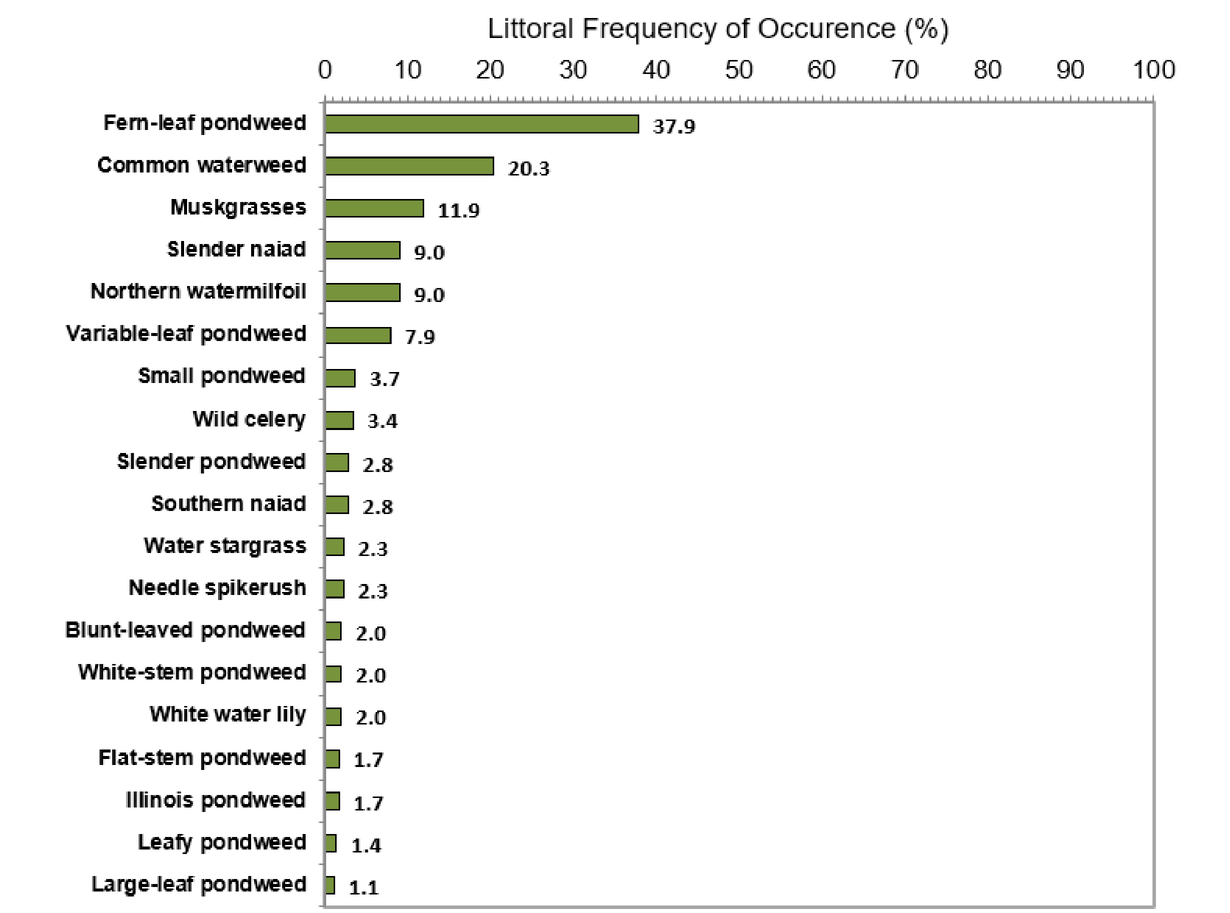


Figure 3.3-2. Forest Lake littoral frequency of occurrence of aquatic plant species. Created using data from the 2022 whole-lake point-intercept survey. Species with 1% or greater occurrence are displayed.

Figure 3.3-3 shows five aquatic plants that exhibited statistically valid increases in occurrence from the 2019-2022 survey. Three native species showed valid decreases in occurrence during the same timeframe and are displayed on Figure 3.3-4. The occurrence of all species recorded from past point-intercept surveys are included in Appendix C.

Due to their morphological similarity and often difficulty in differentiating between them, the occurrences of small pondweed (*Potamogeton. pusillus*) and slender pondweed (*P. berchtoldii*) and common waterweed (*Elodea canadensis*) and slender waterweed (*E. nuttalii)* were combined for this analysis.

As fern-leaf pondweed's name suggests, the arrangement of leaves along the stem give this plant a fern-like appearance. Fern-leaf pondweed typically develops large colonies over soft sediments which grow close to the lake bottom, and it is one of the deepest-growing vascular plants in Wisconsin. Large beds of fern-leaf pondweed provide excellent structural habitat for aquatic wildlife and help to prevent the suspension of the soft bottom sediments in which they grow. In Forest Lake, fern-leaf pondweed was present between 4 and 20 feet of water. The occurrence of fern-leaf pondweed was 37.9% in 2022 which continues an increasing trend when compared to previous years (Figure 3.3- 3 and Photograph 3.3-5).

Common and slender waterweeds are interesting plants in that although it sometimes produces root-like structures that bury themselves into the sediment, it is largely an unrooted plant that can obtain nutrients directly from the water. As a result, this plant's location in a lake can be dependent upon water movement. In Forest Lake, common and slender waterweeds were present between 3 and 17 feet of water. The occurrence of common and slender waterweeds was 20.3% in 2022 and has been relatively stable over the period of study (Figure 3.3-3 and Photograph 3.3-5).

Muskgrasses are a genus of macroalgae, of which there are ten documented species that occur in Wisconsin. Dominance of the aquatic plant community by muskgrasses is common in hardwater lakes and these macroalgae have been found to be more competitive against vascular plants (e.g., pondweeds, milfoils, etc.) in lakes with higher concentrations of calcium carbonate in the sediment (Kufel and Kufel 2002); (Wetzel 2001). Muskgrasses 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 encrustations which form on these plants, aiding in improving water quality by making the phosphorus unavailable to phytoplankton (Coops 2002). Muskgrasses can be easily identified by their strong skunk-like odor. As well as providing a food source for waterfowl, muskgrasses often serves as a sanctuary for small fish and other aquatic organisms. In Forest Lake, muskgrasses were present between 3 and 16 feet of water. The occurrence of muskgrasses was 11.9% in 2022 which has fluctuated over the years Forest Lake has been surveyed (Figure 3.3-3 and Photograph 3.3-5).

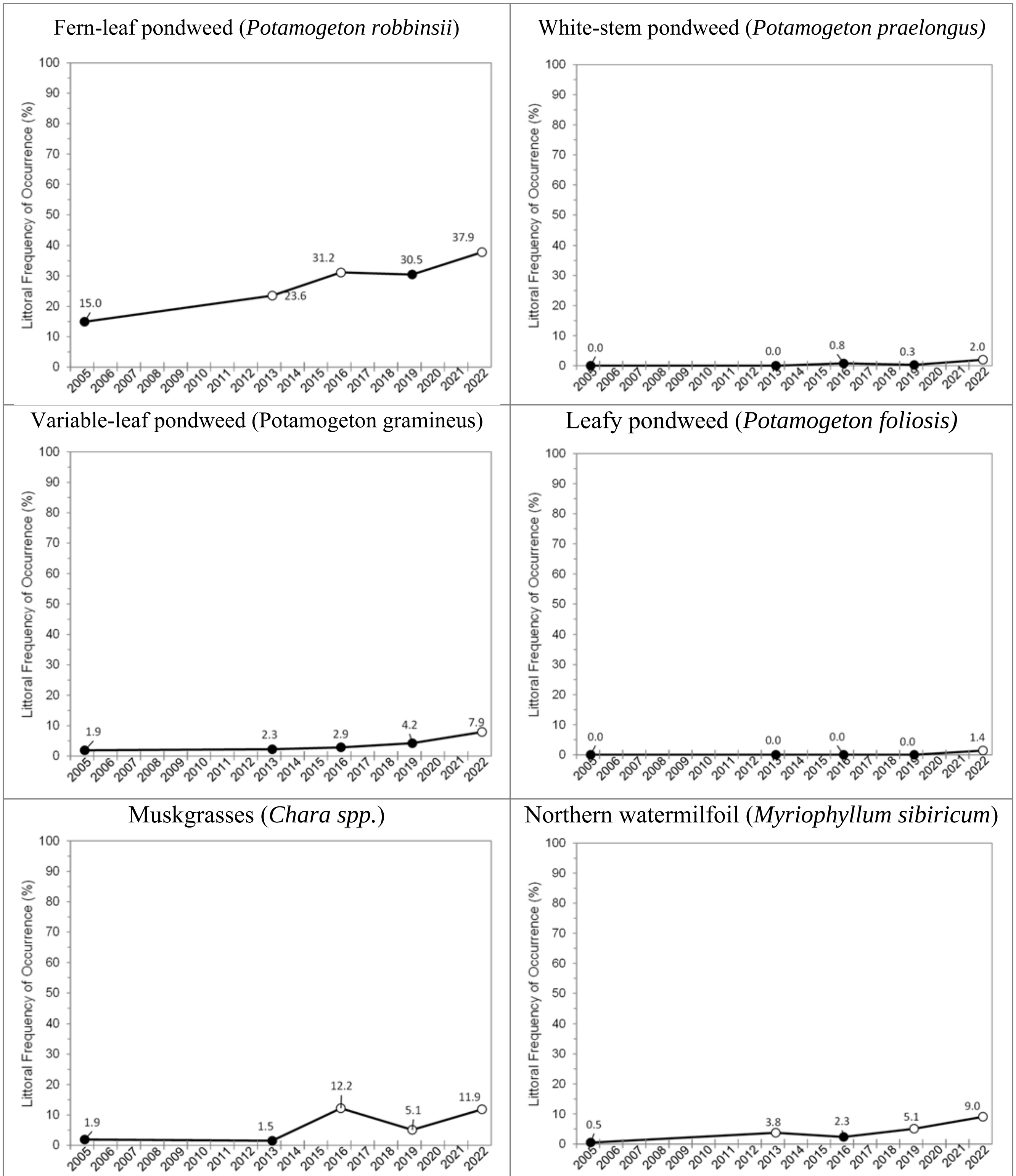
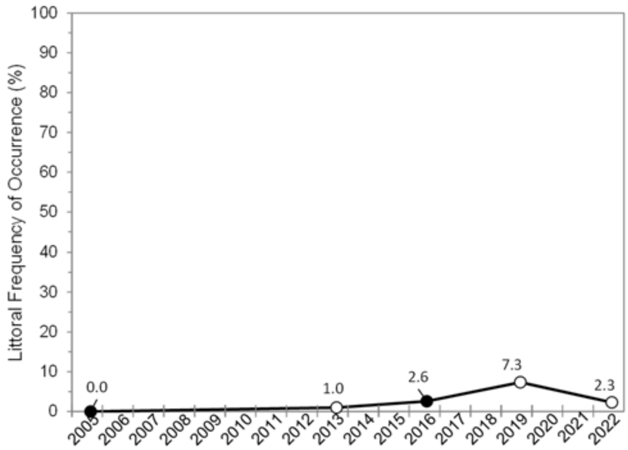
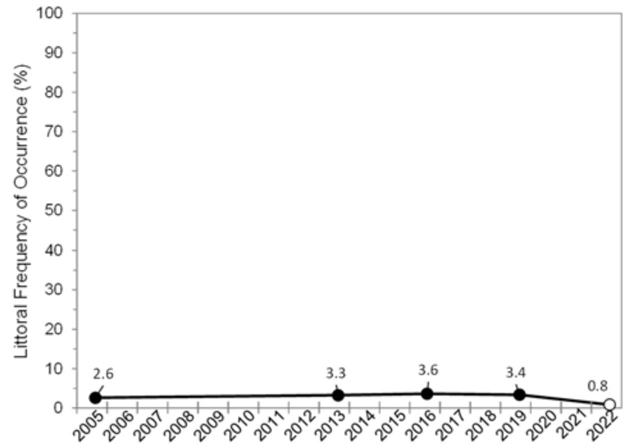


Figure 3.3-3. Littoral frequency of occurrence of native aquatic plant that exhibited statistically valid <u>increases</u> in occurrence from 2019-2022 in Forest Lake. Open circle represents statistically valid change in occurrence from previous survey.

Water stargrass (*Heteranthera dubia*)



Water marigold (*Bidens beckii)*



Flat-stem pondweed (*Potamogeton zosteriformis)*

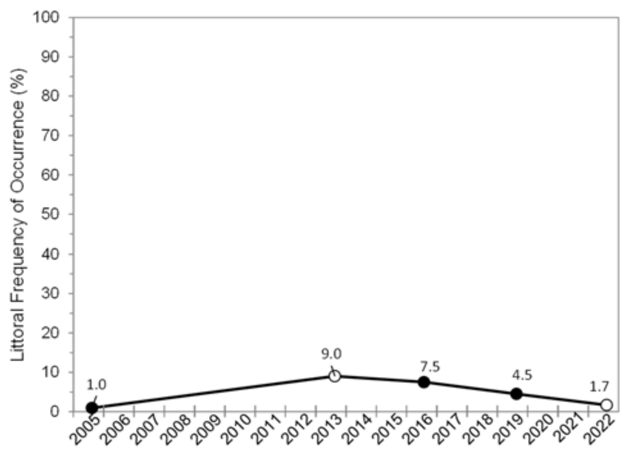


Figure 3.3-4. Littoral frequency of occurrence of native aquatic plant that exhibited statistically valid decreases in occurrence from 2019-2022 in Forest Lake. Open circle represents statistically valid change in occurrence from previous survey.

Whole-lake point-intercept surveys are used to quantify the abundance of individual plant species within the lake. Of the 354 point-intercept sampling locations that fell at or shallower than the maximum depth of plant growth (the littoral zone) in Forest Lake in 2022, aquatic plant rake fullness data was also collected. The 2022 data indicates that 38% of the sampling locations contained vegetation with a total rake fullness rating (TRF) of 1, 18% had a TRF rating of 2, and 14% had a TRF rating of 3 (Figure 3.3-5). The TRF data indicates that where aquatic plants are present in Forest Lake, they are at a moderate abundance. Total rake fullness levels were not recorded in 2005 and have shown a very consistent ratio of each fullness level from surveys completed between 2013-2022 (Figure 3.3-5). These data indicate a stable amount of plant biomass since 2013.

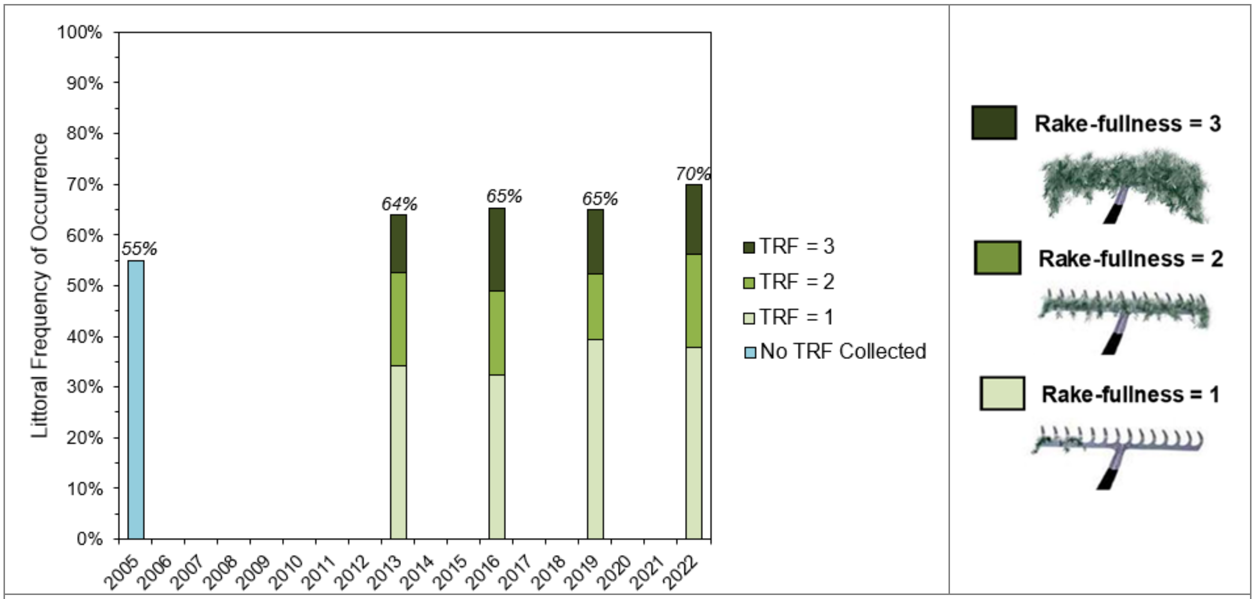


Figure 3.3-5. Aquatic vegetation total rake fullness ratings.

Data collected during the aquatic plant surveys was also used to complete a Floristic Quality Assessment (FQA) which incorporates the number of native aquatic plant species recorded on the rake during the point-intercept survey and their average conservatism. The data used for these calculations does not include any incidental species (visual observations) but only considers plants that were sampled on the rake during the survey. For instance, while a total of 29 native species were located in Forest Lake in the 2022 survey, 27 were physically encountered on the rake while the remaining 1 species was located incidentally and 1 other (Eurasian watermilfoil) is an invasive species. Figure 3.3-6 displays the species richness, average conservatism, and floristic quality of Forest Lake along with ecoregion and state median values.

Forest Lake's native plant species richness value has averaged 26.4 over the course of the last five point-intercept surveys. This falls above the median values for the ecoregion (21) and state (19). Forest Lake's average species conservatism of 6.7 over the last five surveys equals that of the ecoregion, and is slightly higher than the state (6.3) medians as well. This indicates that Forest Lake has a comparable number of environmentally sensitive species (higher C-values) when compared to other lakes within the Northern Lakes and Forests ecoregion. Using the species richness and average conservatism values, Forest Lake's Floristic Quality Index was 34.8 in 2022, and averaged 34.5 over the last five surveys which falls above the median value for lakes in the NLFL ecoregion (30.8) and for lakes statewide (27.2).

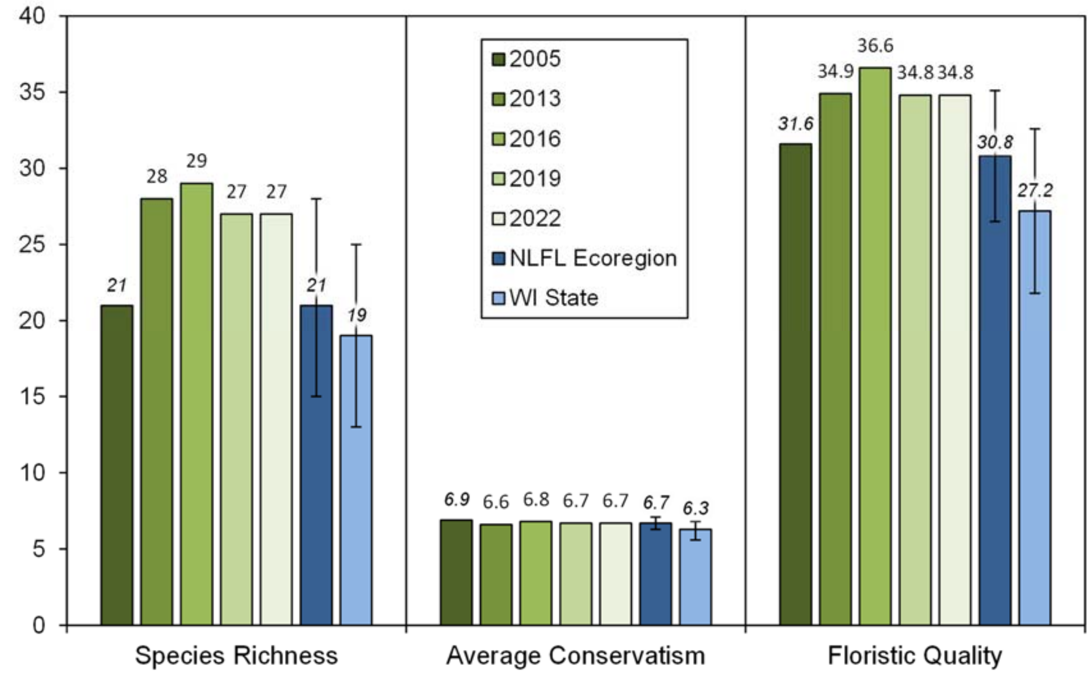


Figure 3.3-6. Floristic Quality Assessment. Error bars represent interquartile range of comparable. Regional and state medians calculated with Onterra and WDNR data. Analysis follows (Nichols, 1999).

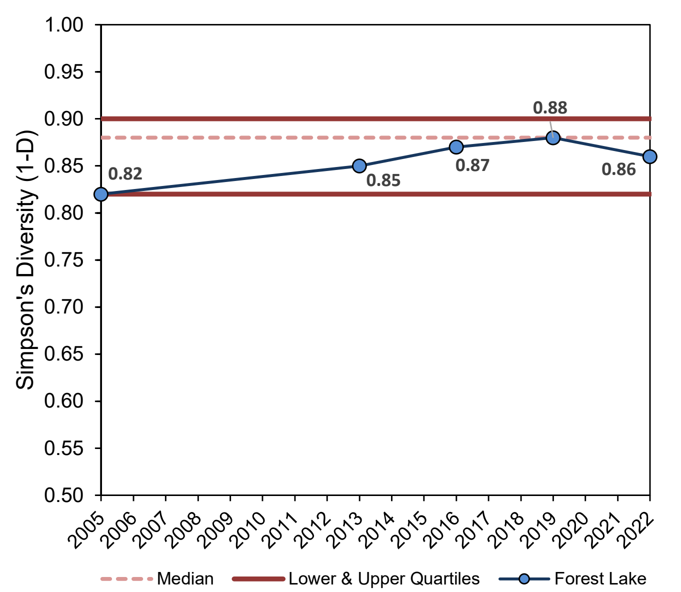


Figure 3.3-7. Simpson's Diversity Index. Solid lines indicate 25th and 75th percentiles for NLFL lakes; dashed line indicates median for NLFL lakes.

Simpson's Diversity Index is a measure of both the number of aquatic plant species in a given community and their abundance. This measurement is important because plant communities with higher diversity are believed to be more resilient to disturbances and natural fluctuations that affect plant growth (e.g., changes water clarity, water levels, etc.). Plant communities with higher diversity also provide more diversity in habitat types and food sources for invertebrates, fish, and other wildlife. Higher species diversity leads to a healthier and more adaptive system that is resistant to disturbance and more stable over time. Unlike species richness which is simply the number of aquatic plant species within the community, species diversity considers how evenly those species are distributed throughout the community.

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 Forest Lake's diversity values rank. Using data collected by Onterra, quartiles were calculated for 212 lakes within the NLFL ecoregion (Figure 3.3-7). The Simpson's Diversity Index values were calculated using past years point-intercept survey data. Forest Lake's Simpson's Diversity Index value has been stable at 0.82-0.88 over the course of the point-intercept surveys spanning 2005-2022, which is near the ecoregion median.

3.4 Non-native Aquatic Species in Forest Lake

Eurasian watermilfoil (Myriophyllum spicatum)

Eurasian watermilfoil (*Myriophyllum spicatum*; EWM) was first documented in Forest Lake during the summer of 2001. Eurasian watermilfoil is an invasive species, native to Europe, Asia and North Africa, that has spread to most Wisconsin counties (Figure 3.4-1). 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, Eurasian watermilfoil 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 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.

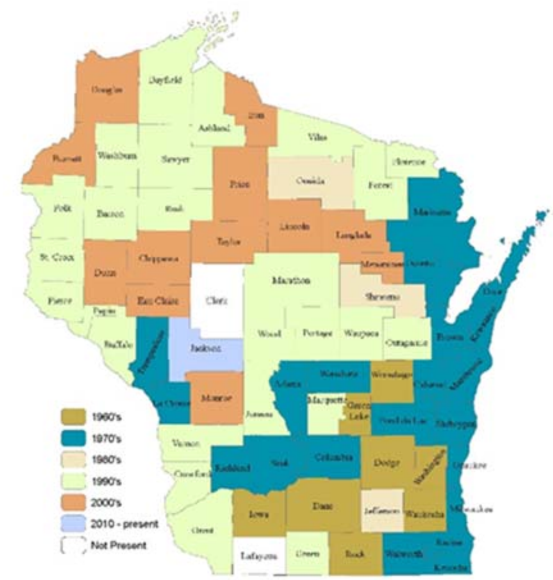


Figure 3.4-1. Spread of Eurasian watermilfoil within WI counties. WDNR Data 2022 mapped by Onterra.

Fragmentation

It is true that EWM fragments transferred from one lake to another is the cause of essentially every new EWM population. It is also true that EWM fragments are the vector of population spread within a lake. Everyone has been conditioned that EWM fragments are bad. But in reality, it is much more complex.

There are two types of EWM fragments, auto-fragments and allo-fragments. Auto-fragmentation is the purposeful fragmentation of EWM for the purposes of asexual reproduction. This plant has evolved a mechanism to increase its population in this manner. The parent plant actually sends

Photograph 3.4-1. EWM fragment with adventitious roots. Photo credit Onterra.

carbohydrate reserves to the growing tip (apical meristem) before the fragment separates. Also, before separation, the fragment will start growing root-like structures (adventitious roots, Photograph 3.4-1). Applying an analogy, that plant has packed its bags and is ready to endure floating around in the lake for a few days and then trying to grow in a new place in the lake. This naturally happens in all lakes. Onterra's experience is that there are two main events – once in late-spring and again towards the end of the growing season. Allo-fragments are those fragments that break off by mechanical breakage by boats, wind, mechanical harvesting, etc. These fragments have a smaller chance of producing a new plant – continuing with the analogy, because they did not get to pack their bags and have to try to make it with what they have on hand.

For a new infestation, lake managers are concerned with all types of fragments. But for an established population with auto fragmentations occurring naturally, a few additional allo-fragments are insignificant to worry about from a population management perspective. However, fragments of any plant species can be unwelcomed by riparians when they accumulate on their shoreline.

Frankly, for established populations, lake managers are not really concerned with EWM fragments at all (either kind). The footprint of EWM is everywhere conducive for the plant under the current environmental conditions. If it is not growing in a part of the lake, it is not because it has never been exposed to that area. It is because the conditions are not favorable at this time. Conditions change from year to year and the footprint and density of EWM will also, even if unmanaged. This concept is factored into the creation of short and long-term EWM management strategies.

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.

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 clearest for unmanaged lakes in the Northern Lakes and Forests Ecoregion (NLF) and the North Central Hardwood Forests Ecoregion (NCHF) (Figure 3.4-2).

The results of the study clearly indicate that EWM populations in unmanaged lakes can fluctuate greatly between years (Figure 3.4-2). 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. 2019 also experienced record rainfall which may have had an impact on the EWM population indirectly through a decrease in water clarity.

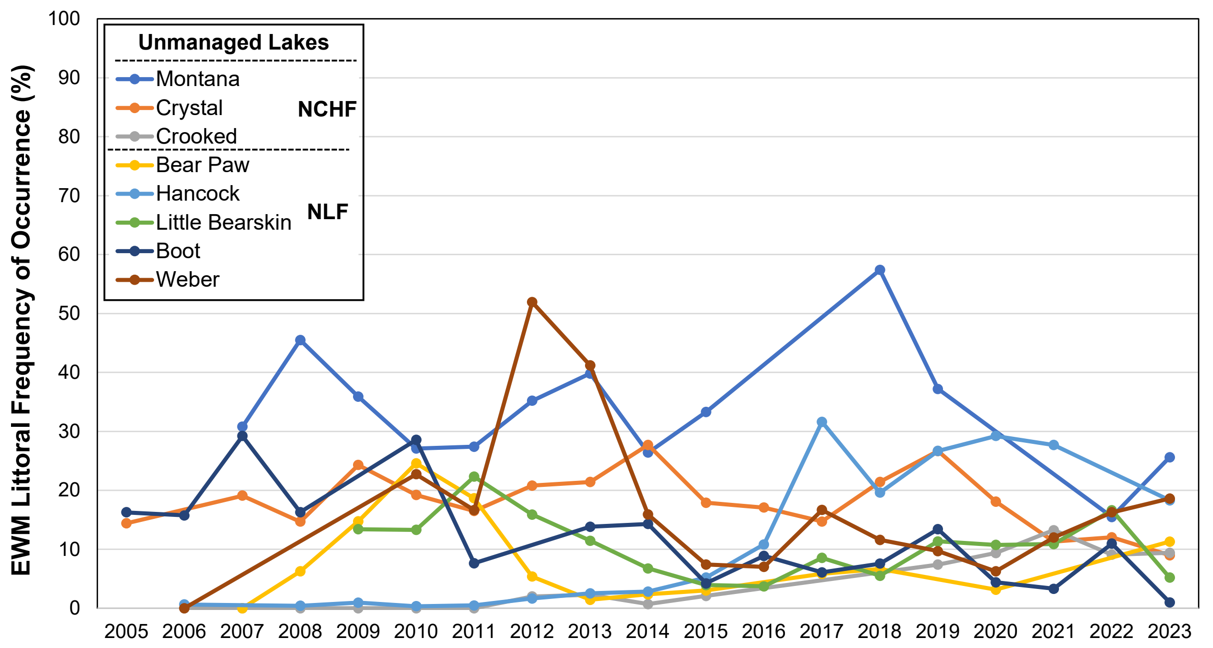
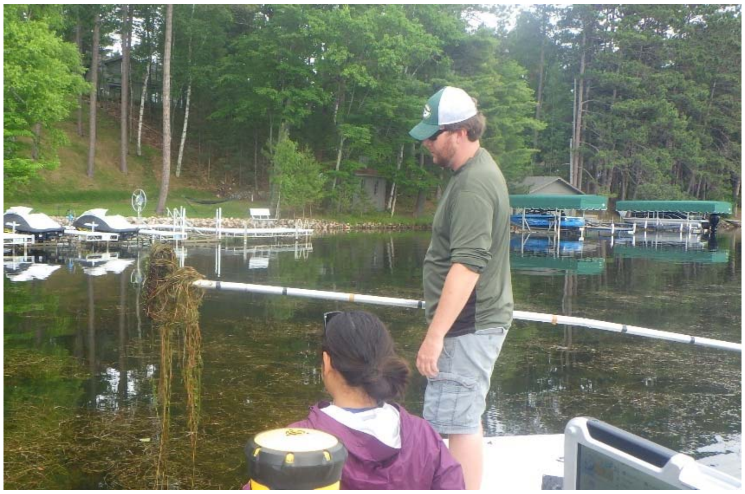


Figure 3.4-2. LFOO of EWM in northern ecoregions without management. Data provided by and used with permission from WDNR. LFOO = littoral frequency of occurrence.

It is important to note that two types of surveys are discussed in the subsequent materials: 1) whole lake point-intercept surveys and 2) EWM mapping survey. Overall, each survey has its strengths and weaknesses, which is why both are utilized in different ways as part of this project.

The point-intercept survey provides a standardized way to gain quantitative information about a lake's aquatic plant population through visiting predetermined locations and using a rake sampler to identify all the plants at each location. The point-intercept survey can be applied at various scales. Most commonly, the point-intercept survey is applied at the whole-lake scale to provide a lake-wide assessment of the overall plant community. More focused point-intercept surveys, called sub-sample point-intercept surveys, may be conducted over specific areas to monitor an active management strategy such as herbicide treatments. These types of sub-sample point-intercept surveys have also been applied on Forest Lake in the past in monitoring programs related to the 2020 and 2022 herbicide spot treatments.

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 EWM mapping survey, the entire littoral area of the lake is surveyed through visual observations from the boat (Photograph 3.4-2). 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 are qualitatively attributed to a density rating based upon a five-tiered scale from *highly scattered* to *surface matting*. Point-based techniques were applied to AIS locations that were considered as *small plant colonies* (<40 feet in diameter), *clumps of plants*, or *single or few plants*.



Photograph 3.4-2. Point-intercept survey on a WI lake. Photo credit Onterra.

Photograph 3.4-3. EWM mapping survey on a Wisconsin lake. Photo credit Onterra.

EWM population of Forest Lake

Using data from the point-intercept surveys that have been completed over the years, the littoral frequency of occurrence of EWM can be compared for Forest Lake (Figure 3.4-3). The frequency of occurrence of EWM has remained relatively low in all surveys. In 2019, the occurrence of EWM reached 3.4% and declined to 0.6% by 2022 as the population continued to be suppressed by management activities. In 2005 and 2016, EWM was below detection limits of this survey methodology (0% occurrence), although EWM was known to be present in the lake at relatively modest levels.

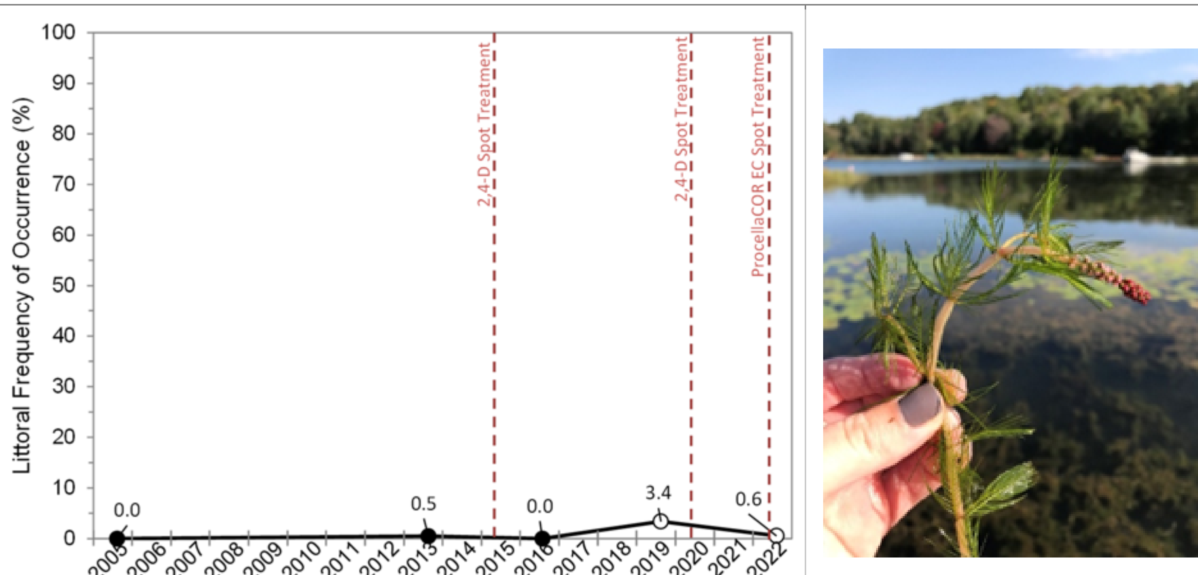


Figure 3.4-3. EWM littoral frequency of occurrence within Forest Lake. Data from available point-intercept surveys. Photo credit Onterra.

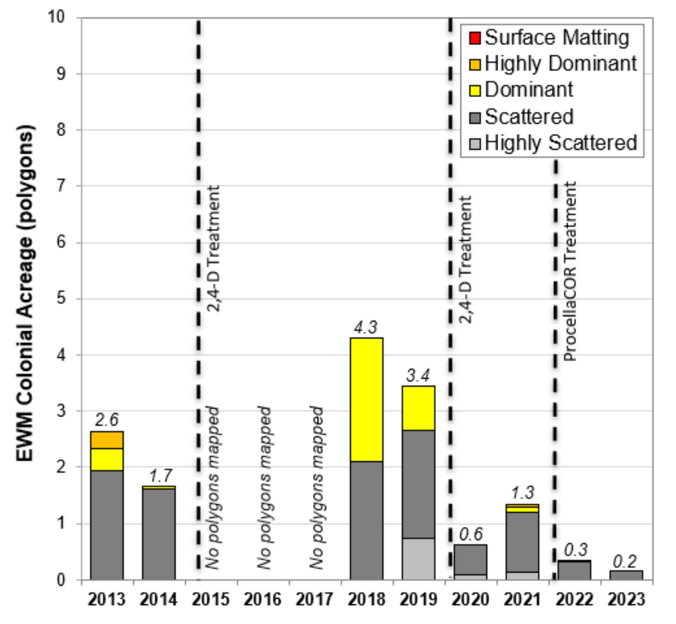


Figure 3.4-4. Acres of colonized EWM (polygons) from 2013-2023 in Forest Lake.

The EWM population in Forest Lake has been monitored since 2013 through the completion of annual Late-Summer EWM Mapping Surveys by Onterra ecologists allowing for a good historical record of the EWM population dynamics. Colonized acreage has not exceeded 4.3 acres in any year dating back to 2013. The figure demonstrates the reduction in EWM acres following herbicide management events in 2015, 2020, and 2022. The late-summer 2023 mapping survey delineated just 0.2 acres of *scattered* density EWM in the lake which is the lowest number of acres since 2017. It is important to note that the acreage only accounts for EWM occurrences that were mapped with area-based (polygons) mapping methodologies. Many additional EWM occurrences were mapped with point-based methodologies throughout the system and are described as either *single or few plants*, *clumps of plants*, or *small plant colonies*. Any EWM mapped with point-based methods do not contribute to the acreages displayed on Figure 3.4-4.

Forest Lake Historic EWM Management

Initial management efforts following detection included volunteer-based hand-harvesting activities and a spot 2,4-D treatment in 2001. The herbicide treatment was determined to be highly effective and continued volunteer-based hand-harvesting occurred in subsequent years, seemingly maintaining the EWM population at low levels. In 2013 and 2014, FLA supplied over 575 volunteer hours monitoring and hand-harvesting the EWM population. During those same years, the group paid for 230 hours of harvesting by professionals. Sufficient EWM was located in the northern portion of the lake to warrant an 8-acre 2,4-D treatment during the spring of 2015, which met control expectations.

No active EWM management occurred during 2016-2017, and expanding EWM at the time resulted in professional hand harvesting efforts during 2018-2019.

In 2020, a 2,4-D herbicide treatment strategy was developed to target two areas of the lake that held largest populations of EWM. These bays, located on the northern shoreline of the lake, are described as the boat landing bay and the northwest bay. Ultimately, EWM control in the boat landing bay was high and met treatment expectations. EWM control in the northwest bay did not meet treatment expectations for the year of treatment.

Table 3.4-1. EWM Management History for Forest Lake from 2013 – present.

Year	AIS Management	Acres Treated	Chemical
2013	DASH/HH		
2014	DASH/HH		
2015	Herbicide Treatment	8.1	2,4-D
2016	None		
2017	None		
2018	DASH/HH		
2019	DASH/HH		
2020	Herbicide Treatment	14.34	2,4-D
2021	DASH/HH		
2022	Herbicide Treatment and DASH/HH	2.8	ProcellaCOR
2023	DASH/HH		

In 2021, the FLA developed a professional hand-harvest strategy to target multiple areas of the lake that had low density areas of EWM. In 2021, divers spent 63 hours underwater and removed 218 cubic feet of EWM.

The results from the 2021 late-season survey also served to assess *year-after treatment* efficacy of the two 2,4-D treatment areas. Only a handful of single EWM plants were observed in 2021 in the boat landing bay and met control expectations. Monitoring of the northwest bay following both herbicide and subsequent DASH efforts showed the results did not meet the control expectations for either control strategy. Given the limited success of past management efforts in the northwestern bay, the FLA planned for and carried out a 2.8-acre ProcellaCOR treatment in 2022 that was designed with the expectation that the herbicide would mix within the waters of the northwest bay and impact EWM throughout this area of the lake.

The 2022 ProcellaCOR™ treatment looked promising during the *year of treatment* (2022), as very little EWM was detected in the application sites or within the area of potential impact. The *year-after-treatment* (2023) results indicated some EWM population rebound in the June 2023 EWM mapping survey while the population was still below pre-treatment levels with no colonized areas present in the bay. Professional hand harvesting efforts took place during 2022 with approximately 228.5 cubic feet of EWM harvested from four main sites around the lake with the boat landing bay site receiving the greatest amount of effort.

The late-summer 2022 EWM mapping survey indicated a modest population within the lake as a results of recent management activities. No areas of EWM met the FLA's trigger within their management plan for considering herbicide treatment in 2023. The FLA elected to continue an aggressive EWM management strategy during 2023 through a coordinated professional hand harvesting program that would target much of the known population in the lake.

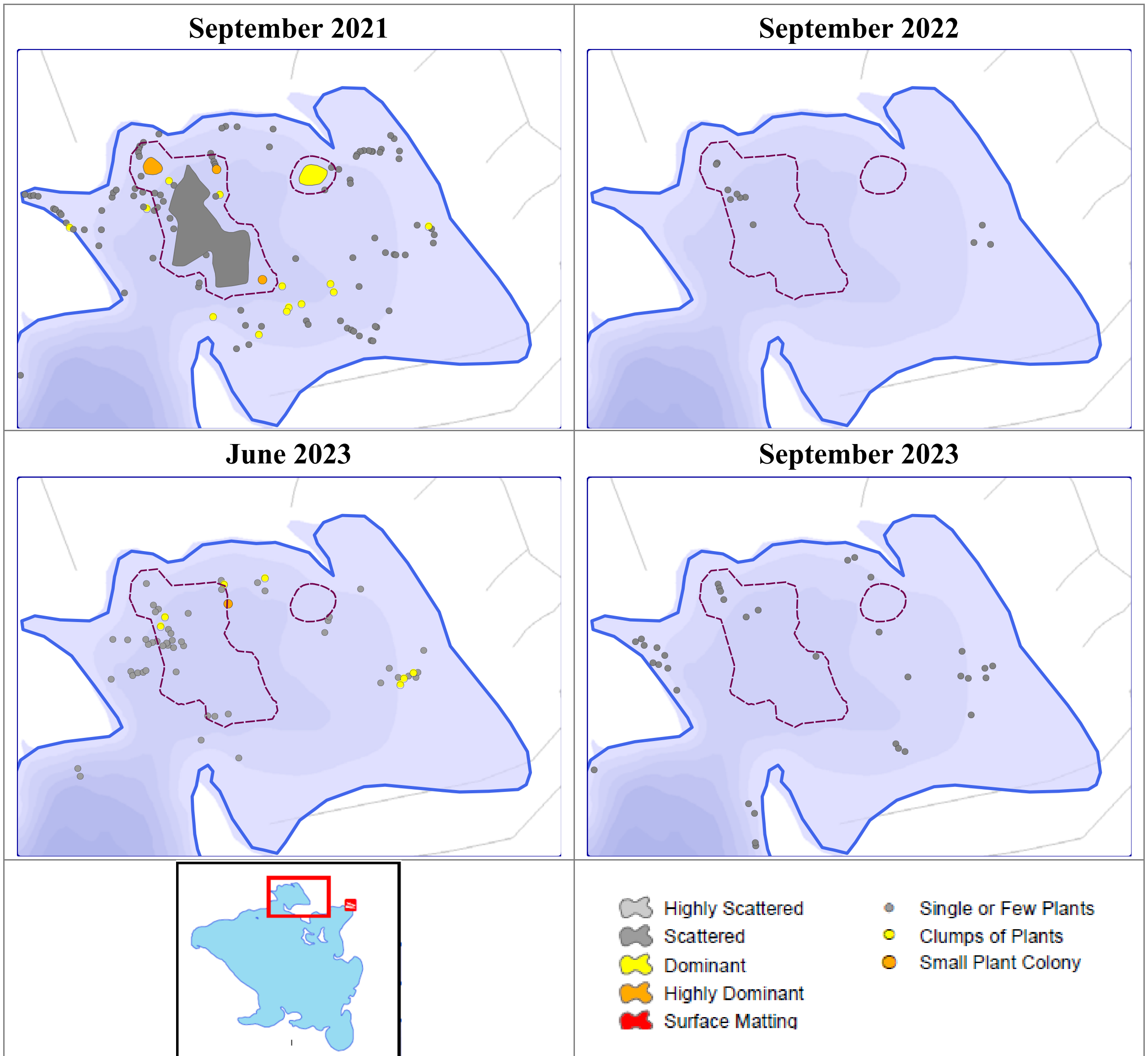


Figure 3.4-5. EWM population progression in northwestern bay of Forest Lake. 2022 ProcellaCOR herbicide treatment area in purple outline. Professional hand harvesting efforts occurred in July-August 2023.

2023 EWM Management and Monitoring

Onterra ecologists completed an Early-Season EWM Mapping Survey on Forest Lake on June 14, 2023 with one purpose being to update the professional hand harvesting strategy for the remainder of the summer as well as to search the lake for potential occurrences of curly-leaf pondweed. During this survey, EWM was found to have expanded in the bay which was treated in 2020 but the overall lake-wide levels were still low (Map 2). This meander-based visual survey did not locate any occurrences of curly-leaf pondweed. At present, curly-leaf pondweed either does not occur in Forest Lake or exists at an undetectable level.

The FLA contracted with Aquatic Plant Management, LLC in 2023 to provide professional hand harvesting efforts throughout the lake. Professional harvesting activities included five days utilizing Diver Assisted Suction Harvesting (DASH) and 13 days of traditional hand harvesting during July, August, and early September. A total of 283.5 cubic feet of EWM was harvested from 15 sites around the lake during the professional harvesting activities in 2023. Details of the professional hand harvesting efforts are included in Appendix D.

Onterra ecologists completed a Late-Summer EWM Mapping Survey on Forest Lake on September 14, 2023. The only colonized EWM that was delineated during the survey was a relatively small (0.2-acres) *scattered* colony located between the two islands on the northeast part of the lake. The majority of the EWM population in the lake is comprised of *single plants* or *clumps of plants*. The population is widely scattered in littoral areas of the lake with loose concentrations of occurrences in some of the bays of the lake. The current population is well below levels that would impact the recreational use or the ecology of the lake. Aggressive hand harvesting efforts during 2022-2023 have suppressed the EWM population in the lake and inhibited expansion of the species from forming larger colonized areas.

Future AIS Management Philosophy

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. BMPs for aquatic plant management change rapidly, as new information about effectiveness, non-target impacts, and risk assessment emerges. One of the primary purposes of completing an APM Update is to ensure that the group's goals and actions align with what is considered to be the current BMP for AIS management. Materials included within the text below serve to provide an overview of current BMP's for EWM management for the FLA to review and consider when creating their updated APM Plan.

During the Planning Committee meetings held as part of this project, Onterra outlined three broad EWM population management perspectives for consideration, including a generic potential action plan for each (Figure 3.4-6). During these discussions, conversation regarding risk assessment of the various management actions was also discussed. Onterra provided extracted relevant chapters from the WDNR's *APM Strategic Analysis Document* to serve as an objective baseline for the FLA to weigh the benefits of the management strategy with the collateral impacts each management action may have on lake ecosystem. These chapters are included as Appendix E. The FLA Planning Committee also reviewed these management perspectives in the context of perceived riparian stakeholder support.

1. **No Coordinated Active Management**
   (Let Nature Take its Course)
   - Focus on education of manual removal methods for property owners
   - Lake organization does not oppose contracted efforts, but does not organize or pay for them
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)
   - Hand-harvesting alone is not likely able to accomplish this goal and herbicides or a mechanical harvester may be required

Figure 3.4-6. Potential EWM Management Perspectives

Let Nature Take its Course: In some instances, the EWM population of a lake may plateau or reduce without conducting active management, as shown in the WDNR Long-Term EWM Trends Monitoring Research Project on Figure 3.4-2. Some lake groups decide to periodically monitor the EWM population, typically through a 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. Individual riparians may choose to hand-remove the EWM within their recreational footprint, but most often the lake group chooses not to assist financially or with securing permits. In some instances, the lake group may select this management goal, but also set an EWM population threshold or management *trigger* where they would revisit their management strategy if the population reached that level. Said another way, the lake group would let nature take its course up until populations reached a certain lake-wide level or site-specific density threshold. At that time, the lake group would investigate whether active management measures may be justified.

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 system 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 as management actions have unintended collateral impacts.

In early EWM populations, the entire population may be targeted through hand-harvesting or spot treatments. 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. In areas of the state that contain highly established and prevalent EWM populations, lake-wide population management is often considered too aggressive by local WDNR regulators. In these instances, the nuisance conditions are targeted for management and other areas are tolerated or avoided. In recent years the FLA has managed the lake-wide population through a combination of herbicide treatments and coordinated hand harvesting efforts.

Nuisance Control: 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 these cultural ecosystem services.

There has been a change in preferred strategy amongst many lake managers and regulators when it comes to established EWM population in recent years. Instead of chasing the entire EWM population with management, perhaps focusing on the areas that are causing the largest impacts can be more economical and cause less ecological stress. The majority of EWM management in Wisconsin would be considered nuisance management, where dense areas that are causing navigation or recreation issues are prioritized for management and dense areas not meeting these criteria being left unmanaged. Mechanical harvesting and herbicide spot treatments are most typically employed to reach nuisance management goals, although hand-harvesting/DASH is sometimes employed to target small footprints.

Spot vs Whole-Lake or Whole-Basin Treatment Approaches

Spot treatments are a type of control strategy where the herbicide is applied to a specific area (treatment site) such that when it dilutes from that area, its concentrations are insufficient to cause significant affects outside of that area. Spot treatments typically rely on a short exposure time to cause mortality as the herbicide dissipates out of the spots rapidly. 2,4-D was historically the most commonly used spot-treatment herbicide for EWM control in Wisconsin, with more recent strategies incorporating florpyrauxifen-benzyl (ProcellaCOR) or herbicide combinations such as 2,4-D and endothall, or endothall and diquat for example. Studies have confirmed that it is extremely rare that 2,4-D concentrations are maintained within most spot treatments long enough to cause EWM mortality. Spot-treatment designs that tend to be the most successful often require larger sized sites (>5.0 acres) or are otherwise somewhat protected from dissipation by being located in a semi-protected bay for example.

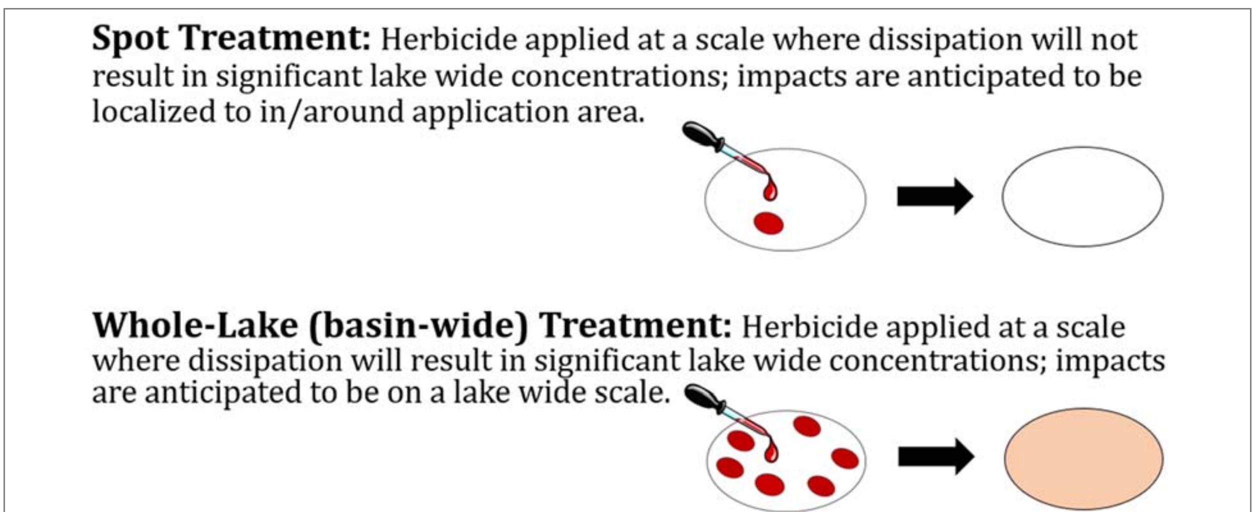


Figure 3.4-7. Ecological definitions of herbicide treatment. Graphics created in conjunction with WDNR.

Whole-lake or whole-basin treatments are a collective of spot-treatments around that lake that are expected to mix into a uniform lake-wide concentration that is sufficient to impact EWM. During 2010-2020, whole-lake and whole-basin herbicide treatments gained popularity, as it was easier to predict EWM control goals and understand levels of collateral native plant impacts. This type of whole-basin treatment has been utilized in the past in Forest Lake in treatments targeting the northwest bay and the boat landing bay.

Florpyrauxifen-benzyl (ProcellaCOR)

The active ingredient florpyrauxifen-benzyl is sold exclusively by SePRO under the tradename ProcellaCOR™. ProcellaCOR™ has been the state's most popular herbicide for EWM management in recent years, supplanting the use of 2,4-D in many applications. This herbicide has largely been used in spot treatment scenarios, but has recently been adopted as a whole-lake treatment option on a number of Wisconsin lakes. The FLA has reviewed available information about ProcellaCOR in the past as they considered its use in 2020 and utilized it in a 2022 treatment in the northwestern bay.

Onterra has monitored numerous ProcellaCOR™ treatments in Wisconsin since 2019 with data analysis related to herbicide concentration monitoring and native aquatic plant impacts being investigated in the majority of treatments. Analysis of these data have allowed lake managers to better understand the ways in which the herbicide dissipates or mixes within a lake in the hours and days after application. Additionally, aquatic plant monitoring data provides insights as to which native species are typically impacted with ProcellaCOR™ treatments. From Onterra' s monitoring experience, most treatments have demonstrated generally high levels of EWM control efficacy with a lower degree of impacts to non-target aquatic plant species compared to other commonly employed herbicides.

Lake managers continue to learn how to successfully implement this form of treatment after being registered for use in Wisconsin relatively recently. ProcellaCOR™ is in a new class of synthetic auxin mimic herbicides with reportedly short concentration and exposure time (CET) requirements compared to other systemic herbicides. Auxin-mimic herbicides are translocated throughout the plant and suppress growth regulation hormones, so the plant grows uncontrollably at the cellular level which causes mortality.

Herbicide Resistance

While understood in terrestrial herbicide applications for years, herbicide resistance is an emerging topic amongst aquatic herbicide applicators, lake management planners, regulators, and researchers. Herbicide resistance is when a population of a given species develops reduced susceptibility to an herbicide over time, such that an herbicide use pattern that once was effective no longer produces the same level of effect. 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 over time.

Repetitive treatments with the same herbicide mode-of-action may cause a shift towards increased herbicide tolerance in the population. Rotating herbicide use-patterns can help avoid population-level herbicide tolerance evolution from occurring.

Stakeholder Survey Responses to Eurasian Watermilfoil Management

As discussed in Section 2.0, the stakeholder survey asks many questions pertaining to perception of the lake and how it may have changed over the years. The return rate of the 2023 survey was 62%. Because the response rate was above 60% in 2023, the stakeholder survey results can be interpreted in the context of the overall population offered to participate in the survey.

Question 23 gauged stakeholder support for various EWM management techniques (Figure 3.4-8). Stakeholders were largely not supportive of the no active management option. Hand harvesting including DASH was highly supported as was an integrated management approach. Herbicide treatment was supported by the 75% of stakeholders (pooled as either *highly supportive* or *somewhat supportive*) with 16% opposed (pooled as either *not supportive* or *somewhat unsupportive*). Mechanical harvesting received mixed support with more stakeholders not supportive than supportive.

23. The Forest Lake Association is currently assessing future techniques for continuing to manage the EWM population. What is your level of support for the future use of the following EWM management techniques in Forest Lake? Please select one response for each control technique.

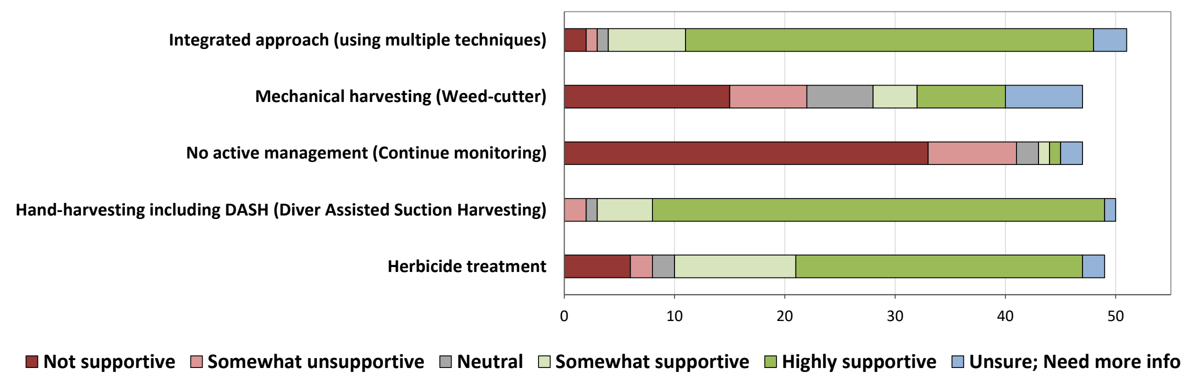


Figure 3.4-8. Select survey responses from the Forest Lake Stakeholder Survey. Additional questions and response charts may be found in Appendix B.

When asked about concerns for using certain active EWM management techniques in Forest Lake, use of aquatic herbicides received the most responses for the choices given, which DASH/hand harvesting was the most selected option for concern of potential costs (Figure 3.4-9).

24. What concerns, if any, do you have for the future use of aquatic herbicides, DASH/hand harvesting, or Mechanical Harvesting to target EWM in Forest Lake?

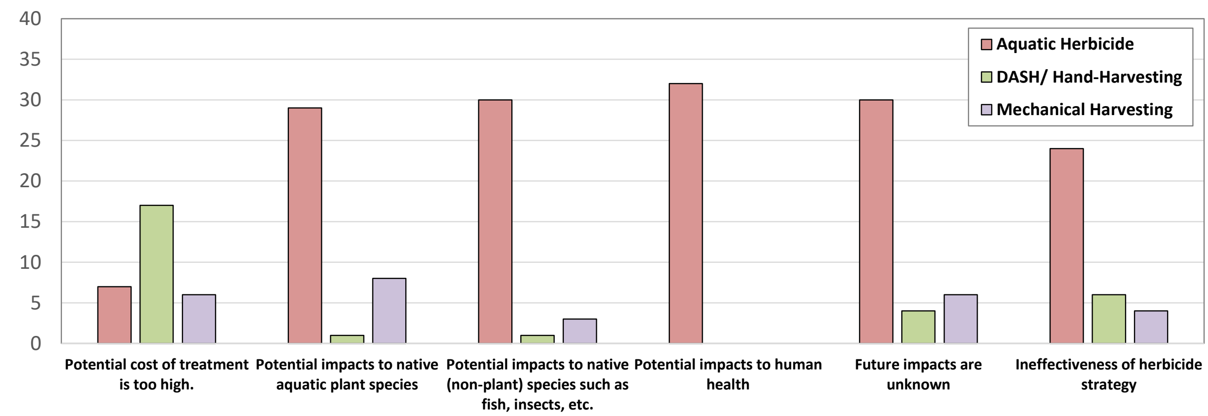


Figure 3.4-9. Select survey responses from the Forest Lake Stakeholder Survey. Additional questions and response charts may be found in Appendix B.

4.0 SUMMARY & CONCLUSIONS

The objective of this project was for the FLA to review aspects of their Comprehensive Management Plan and update the Plan to account for changes in best management practices and lessons learned since the creation of the Plan. The Implementation Plan detailed below includes many of the same goals and actions as were originally included in the FLA's 2019 Comprehensive Management Plan. Each of these goals were re-evaluated during this project and updated if needed. The FLA continues to strive to protect Forest Lake and preserve its high-quality condition through promoting good lake stewardship and by providing outreach and educational materials amongst property owners and FLA membership.

An evaluation of the available water quality data indicates that Forest Lake continues to have excellent water quality based on trophic parameters including phosphorus, chlorophyll-a, and Secchi disk transparency. Data collected since the completion of the 2019 Comprehensive Plan largely mirrors past conclusions with no negative trends detected.

In this project, the most current land cover information available was used to update the general description of the watershed from the 2019 plan. This land cover assessment found that there are no significant differences in the Forest Lake watershed boundary since the creation of the comprehensive management plan in 2019. Higher resolution aerial photography allows for some refinements of land cover types within the watershed, however the differences between the previous dataset do not lead to significant changes to nutrient inputs through the watershed modeling process.

An evaluation of the aquatic plant community indicates that Forest Lake continues to harbor a high-quality plant community as demonstrated by floristic quality assessment values well above ecoregion and state median values. Some native aquatic plants have shown variability in occurrence between surveys, while others have remained relatively stable over time. When comparing the 2022 survey to the prior one conducted in 2019, five species showed statistically valid increases in occurrence, and three species showed valid declines while many other species were not statistically different.

Since the FLA's 2019 Comprehensive Management Plan was completed, the FLA has enacted their aquatic plant management strategy including carrying out an integrated approach to EWM management largely through a combination of professional hand harvesting or herbicide treatments. The FLA re-evaluated their recent EWM management strategy during this project and has chosen to continue with a similar integrated approach to EWM management going forward as outlined within the Implementation Plan below.

5.0 IMPLEMENTATION PLAN

The Implementation Plan presented below was created through the collaborative efforts of a Planning Committee comprised of members of the Forest Lake Association (FLA) and ecologist/planners from Onterra. It represents the path the FLA will follow in order to meet their lake management goals. The goals detailed within the plan are realistic and based upon the findings of the studies completed in conjunction with this planning project and the needs of the Forest Lake stakeholders as portrayed by the returned stakeholder surveys, and communications between FLA Board members, WDNR partners, and lake ecologists/planners. 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.

The FLA's Comprehensive Management Plan was finalized in January 2019 and this project served to evaluate the goals and objectives within that plan and update any items accordingly. The Goals below include those originally developed during the 2019 Plan with updated descriptions and other modifications made throughout.

Management Goal 1: Control Existing and Prevent Further Aquatic Invasive Species Infestations within Forest Lake

Management Action:	Continue Clean Boats Clean Waters watercraft inspections at public access location
Timeframe:	Continuation of current effort
Facilitator:	FLA Board of Directors
Potential Grant:	WDNR AIS-Clean Boats Clean Waters Grant
Description:	Currently the FLA monitors the Forest Lake public boat landing using training provided by the Clean Boats Clean Waters program. Forest 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 infestation of other waterways with invasive species that are present in Forest Lake. The goal is to cover the 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. The FLA has set a goal of 200 hours of annual watercraft inspections utilizing a combination of volunteer and paid inspectors. Volunteers would focus upon high-use periods such as weekends and holidays. The long-term goal is permanent monitoring, given the financial capacity to do so. The FLA continues to seek more participation in the CBCW program.

Management Action:	Coordinate annual professional monitoring of EWM.
Timeframe:	Continuation of current effort
Facilitator:	FLA Board of Directors
Description:	The FLA will rely on annual professional EWM monitoring surveys to guide their management strategies while in the past Plan there was also a volunteer-based monitoring component that has been discontinued. As the name implies, the Late-Season EWM Mapping Survey is a professionally contracted survey completed towards the end of the growing season when the plant is at its anticipated peak growth stage, allowing for a true assessment of the amount of this exotic within the lake. For Forest Lake, this survey would likely take place in August or September, dependent on the growing conditions of the particular year. This survey would include a complete or focused meander survey of the lake's littoral zone by professional ecologists and mapping using GPS technology (sub-meter accuracy is preferred). Late Season EWM Mapping Surveys have been conducted annually since 2011 with consistent methodology being used. These data allow lake stakeholders to understand annual EWM populations in response to natural variation and directed management activities. The mapping data that is provided from this survey is instrumental in monitoring active EWM management activities and in developing a management strategy for the following year. When the late-summer EWM mapping survey is not already covered within a grant funded project, the FLA would pay out-of-pocket for this monitoring to ensure the continuity of this dataset. When coordinating the FLA's professional hand harvesting strategy, the late-summer EWM mapping survey is often used to guide the following year's work. However, in some cases it may be preferable to solicit an Early-Season EWM AIS Survey (often during June) as well to refine the upcoming season's harvesting plan, adjust prioritization schemes, or identify new sites with AIS. The FLA has contracted for early-season mapping surveys in the past including during 2024 for this purpose and will consider this monitoring activity on a year-to-year basis.
Action Steps:	
	See description above as.

<u>Management Action:</u>	Conduct EWM population control using hand-harvesting (including DASH) and/or herbicide spot treatments.
Timeframe:	Continuation of current effort
Facilitator:	FLA Board of Directors
Description:	The proactive EWM management strategy that has occurred in Forest Lake since its detection has kept the EWM population at low levels. At these low levels, the EWM population is likely not causing measurable negative ecological impacts to the system nor diminishing the navigability, recreation, or aesthetics of the lake. After discussing EWM management perspectives during this project, the FLA favors a population management approach for EWM. To facilitate this approach, the FLA will utilize an integrated pest management approach which will include professional hand removal by divers as well as herbicide treatments. This type of strategy has been employed since detection of EWM including in the years since the last comprehensive management plan was completed. *<u>Hand-Harvesting</u>* Hand-harvesting would occur during roughly mid-June to mid-September. Conducting hand-harvesting earlier or later in the year can reduce the effectiveness of the strategy, as plants are more brittle and extraction of the roots more difficult. All known EWM occurrences in the lake will be considered within the hand harvesting effort, while a prioritization of efforts will result in some sites not being managed in a given year. If a Diver Assisted Suction Harvest (DASH) component is utilized, the FLA and contracted firm would be responsible for the WDNR permit procedures. The contracted firm would be guided with GPS data from the most recent EWM mapping survey and would track their efforts (when, where, time spent, quantity removed) for post assessments. The FLA has created a prioritization strategy for hand harvesting efforts each year based on various factors including strategic location, scale of infestation, available financial resources, and past management history of the site. This prioritization typically includes discussion between FLA leadership, the lake management consultant, and the contracted professional harvesting firm. This important aspect works towards meeting the FLA's overall EWM management objectives and ensures the actions take place in a coordinated, and intentional manner. The FLA has developed a strong understanding of how hand harvesting (including DASH) can be applied in managing EWM in Forest Lake. This understanding also pertains to expectations and limits of the technique along with associated costs. The FLA understands that the potential for future WDNR AIS Control Grants may provide funding assistance to carry out the FLA's EWM management strategy, however

	the grant program is highly competitive and the FLA will be prepared for the possibility of self-funding their management efforts if grant funding is not received. *<u>Herbicide Spot Treatment</u>* Considerations for conducting an herbicide treatment would be made utilizing the current understanding of best management practices for this technique. While some herbicide spot treatments show promise, the unpredictability of spot treatments state-wide has resulted in less favorability of this strategy with some WDNR regulators and lake managers. This is particularly true in areas of increased water exchange via flow, exposed and offshore EWM colonies, or when traditional weak-acid herbicides like 2,4-D are used. Any herbicide spot-treatments on Forest Lake would consider herbicides thought to be effective under short exposure situations. At the time of this writing, florpyrauxifen-benzyl (ProcellaCOR™), a combination of 2,4-D/endothall (Chinook®), and a combination of diquat/endothall (Aquastrike™) are examples of herbicides with reported short exposure time requirements that are employed for spot treatments of hybrid/Eurasian watermilfoil control in Wisconsin. 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. Any herbicide treatment design for Forest Lake will also consider the potential for meaningful basin-wide dosing calculations similar to the type of designs that have been implemented in Forest Lake in the past. The FLA understands that any herbicide treatment will not eradicate EWM from the lake, but would ideally result in multiple years of a reduced population that could potentially be extended longer through follow-up management efforts such as hand harvesting. When asked to state their level of support for the future use of herbicide use to manage EWM, herbicide treatment was supported by the 75% of stakeholders (pooled as either *highly supportive* or *somewhat supportive*) with 16% opposed (pooled as either *not supportive* or *somewhat unsupportive*). If the following trigger is met, the FLA would initiate pretreatment monitoring and begin discussions, including consultation with WDNR staff, regarding conducting herbicide spot treatments: "*colonized (polygons) areas where a sufficiently large treatment area can be constructed to hold concentration and exposure times*."

	It is believed that these areas are too large to be controlled using hand-harvesting techniques. The FLA would give preference to dominant or greater density EWM populations as at these levels, impacts to recreational use of the lake becomes apparent. In practice, spot-treatments require a minimum size of approximately 5 acres to be able to hold concentration exposure times long enough to achieve EWM mortality. Sites that are somewhat protected from dissipation, such as being located in a bay of a lake, and sites that are broader in shape rather than narrow, would have a greater likelihood for success in a spot-treatment design scenario compared to offshore sites. It is likely that these areas would need to be targeted with herbicides that require short exposure times (diquat, florpyrauxifen-benzyl [ProcellaCOR™]) or herbicide combinations (diquat/endothall, 2,4-D/endothall, etc.). If populations exceed spot-treatment thresholds, larger-scale herbicide strategies may be given consideration. If the trigger is met and the FLA is considering herbicide treatment, early consultation with WDNR would occur along with the following set of bullet points: • Create a Control and Monitoring Plan. The Control and Monitoring Plan would likely be created based on the results of a late-summer EWM mapping survey or in combination with the results of a whole-lake point-intercept survey. These data would be used to create a specific EWM control strategy for the following year including information such as the herbicide to be used, dosing strategy, targeted areas, and an accompanying monitoring strategy. The annual Control and Monitoring Plan would include applicable risk assessment materials for the FLA to review. This might include a summary of available research, toxicity, selectivity, etc. • Monitoring for EWM efficacy at the scale of likely impact. If the treatment is a true spot treatment, the application area should be monitored. If the Area of Potential Impact (AOPI) is larger, such as a bay of the lake or the entire lake, monitoring would occur on that level. • EWM control efficacy would occur 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 the *Draft Aquatic Plant Treatment Evaluation Protocol (October 1, 2016).* This generally consists of collecting quantitative point-intercept the *late-summer prior to treatment* (pre) and the summer following the treatment (post) at the scale of AOPI.

	• Herbicide concentration monitoring may also occur surrounding the treatment if grant funds are being used or the FLA believes important information would be gained from the effort. An herbicide applicator firm would be selected and a permit application would be applied to the WDNR as early in the calendar year as practical, allowing interested parties sufficient time to review the control plan as well as review the permit application. Unless specified otherwise by the manufacturer of the herbicide, an early-season use-pattern would occur. This would consist of the herbicide treatment occurring towards the beginning of the growing season (typically in June), after active growth tissue is confirmed on the target plants. A focused pretreatment survey would take place approximately a week or so prior to treatment. This site visit would evaluate the growth stage of the EWM (and native plants) and confirm the proposed treatment area extents and water depths. This information would be used to finalize and confirm the treatment specifics and dictate approximate ideal treatment timing. Additional aspects of the treatment may also be investigated, depending on the use pattern being considered, such as the role of stratification. In order to meet herbicide spot-treatment control expectations, little to no EWM would be expected to persist in treated areas during the year of treatment, with minimal sign of recovery during the year after treatment as well. Basin-wide treatment strategies would be expected to result in 3 or more years of reduced EWM populations.

Management Action:	Conduct periodic quantitative vegetation monitoring on Forest Lake.
Timeframe:	Point-Intercept Survey every 3-5 years, Community Mapping every 7-10 years
Facilitator:	FLA Board of Directors
Description:	As part of the ongoing aquatic plant monitoring program, a whole-lake point-intercept survey will be conducted at a minimum once every 3-5 years. This will allow a continued understanding of the submergent aquatic plant community dynamics within Forest Lake. The WDNR indicates that repeating a point-intercept survey every five years will generally suffice to meet WDNR planning requirements and grant eligibility requirements. If large-scale aquatic plant management is taking place, more frequent monitoring may be required. A point-intercept survey was conducted on Forest Lake in 2016; 2019, and 2022. The FLA will plan to have the next survey completed between 2025-2027.

	In order to understand the dynamics of the emergent and floating-leaf aquatic plant community in Forest Lake, a community mapping survey would be conducted every 7-10 years. A community mapping survey was conducted on Forest Lake in 2016 as a part of this management planning effort. The FLA will consider replicating the community mapping survey within the next 2-3 years, or at the time of the next management plan update. This survey involves mapping the extents of these communities with GPS guidance during mid or late-summer. Comparisons would be made to the past survey in terms of species composition and the physical extents of the communities around the lake.

Management Action:	Initiate rapid response plan following detection of new AIS
Timeframe:	If/When Necessary
Facilitator:	FLA Board of Directors
Description:	If volunteer or professional surveys locate a suspected new AIS within Forest Lake, the location would be marked (e.g. GPS, maker buoy) and a specimen would be taken to the WDNR Regional AIS Coordinator, or to the Vilas County Land Conservation Department for verification. If the suspected specimen is indeed a non-native species, the WDNR will fill out an incident form and develop a strategy to determine the population level within the lake. The lake would be professionally surveyed, either by agency personnel or a private consulting firm during that species' peak growth phase. If the AIS is a NR40 prohibited species (i.e. red swamp crayfish, starry stonewort, hydrilla, etc.), the WDNR may take an active role in the response. If the AIS is a NR40 restricted species (i.e. purple loosestrife, curly-leaf pondweed, etc.), the FLA would need to reach out to a consultant to develop a formal monitoring and/or control strategy. The WDNR would be able to help financially through the AIS Grant Program's Early Detection and Response program. This grant program is non-competitive and doesn't have a specific application deadline, but is offered on a first-come basis to the sponsor of project waters that contain new infestations (found within less than 3% of the lake and officially documented less than 5 years from grant application date). Currently this program will fund up to 75% percent of monitoring and control costs, and up to $25,000 is available per project.

Management Goal 2: Maintain Current Water Quality Conditions

Management Action:	Monitor water quality through WDNR Citizens Lake Monitoring Network.

Timeframe:	Continuation of current effort.	
Facilitator:	FLA Board of Directors	
Description:	Monitoring water quality is an important aspect of every lake management planning activity. Collection of water quality data at regular intervals aids in the management of the lake by building a database that can be used for long-term trend analysis. Early discovery of negative trends may lead to the reason of why the trend is occurring. Volunteer water quality monitoring is currently being completed annually by Forest 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. The FLA currently monitors the deep hole site within as a part of the advanced CLMN program. This includes collecting Secchi disk transparency and sending in water chemistry samples (chlorophyll-a, and total phosphorus) to the Wisconsin State Laboratory of Hygiene for analysis. The samples are collected three times during the summer and once during the spring. It is important to note that as a part of this program, the data collected are automatically added to the WDNR database and available through their Surface Water Integrated Monitoring System (SWIMS). It will be the Board of Directors responsibility to ensure that a volunteer is prepared to communicate with WDNR representatives and collect water quality samples each year. This action has been implemented since initially being included within the FLA's 2019 Comprehensive Plan and will remain in place going forward.	
Action Steps:		
1.	Trained CLMN volunteer(s) collects data and report results to WDNR and to association members during annual meeting.	
2.	CLMN volunteer and/or FLA Board of Directors would facilitate new volunteer(s) as needed	
3.	Coordinator contacts Sandra Wickman (715.365.8951) to acquire necessary materials and training for new volunteer(s)	

Management Goal 3: Improve Lake and Fishery Resource by Protection and Restoring Shoreland Condition

<u>Management Action:</u>	Educate stakeholders on the importance of shoreland condition and shoreland restoration on Forest Lake.
Timeframe:	Initiated 2019, Ongoing
Facilitator:	FLA Board of Directors

Description:	The shoreland zone of a lake is highly important to the ecology of a lake. When shorelands are developed, the resulting impacts on a lake range from a loss of biological diversity to impaired water quality. Because of its proximity to the waters of the lake, even small disturbances to a natural shoreland area can produce ill effects. Approximately 11% of Forest Lake's shoreline is considered completely urbanized or developed unnatural. This limits shoreland habitat, but it also reduces natural buffering of shoreland runoff and allows nutrients to enter the lake. Because property owners may have little experience with or be uncertain about restoring a shoreland to its natural state, the FLA has decided to take the following steps to increase shoreland restoration on Forest Lake: 1. Educate riparians about the importance of healthy and natural shorelands. 2. Solicit 3 or more riparians to allow shoreland restoration and storm water runoff designs for their property. 3. The FLA will work with Vilas County (Quita Sheehan) or private entity to create design work. Small-scale WDNR grants may be sought to offset design costs. 4. Designs can be shared with FLA members to provide further education of shoreland restoration projects. 5. Move forward with implementing shoreland restoration per the designs that were developed for those riparians that wish to. Project funding would be available through the WDNR's Healthy Lakes Initiative Grants (see below). 6. The FLA's goal would be to have at least 1 shoreland restoration sites to serve as demonstrations sites to encourage other riparians to follow same path of shoreland restoration. The WDNR's Healthy Lakes Initiative Grant program allows partial cost coverage for native plantings in transition areas. This reimbursable grant program is intended for relatively straightforward and simple projects. More advanced projects that require advanced engineering design may seek alternative funding opportunities, potentially through Oconto County. • 75% state share grant with maximum award of $25,000; up to 10% state share for technical assistance • Maximum of $1,000 per 350 ft^2 of native plantings (best practice cap)

	• Implemented according to approved technical requirements (WDNR, County, Municipal, etc.) and complies with local shoreland zoning ordinances • Must be at least 350 ft^2 of contiguous lakeshore; 10 feet wide • Landowner must sign Conservation Commitment pledge to leave project in place and provide continued maintenance for 10 years • Additional funding opportunities for water diversion projects and rain gardens (maximum of $1,000 per practice) also available
Action Steps:	
1.	Recruit facilitator from Planning Committee
2.	Facilitator contacts the Vilas County Land Conservation department to gather information on initiating and conducting shoreland restoration projects. If able, the county staff member would be asked to speak to FLA members about shoreland restoration at their annual meeting.
3.	The FLA would encourage property owners that have restored their shorelines to serve as demonstration sites.

<u>Management Action:</u>	Coordinate with WDNR and private landowners to expand coarse woody habitat in Forest Lake
Timeframe:	Initiated 2019, Ongoing
Facilitator:	FLA Board of Directors
Description:	FLA stakeholders realize the complexities and capabilities of Forest Lake ecosystem with respect to the fishery it can produce. With this, an opportunity for education and habitat enhancement is present in order to help the ecosystem reach its maximum fishery potential. Often, property owners will remove downed trees, stumps, etc. from a shoreland area because these items may impede watercraft navigation shore-fishing or swimming. However, these naturally occurring woody pieces serve as crucial habitat for a variety of aquatic organisms, particularly fish. The Shoreland Condition Section (3.3) and Fisheries Data Integration Section (3.6) discuss the benefits of coarse woody habitat in detail. The WDNR's Healthy Lakes Initiative Grant allows partial cost coverage for coarse woody habitat improvements (referred to as "fish sticks"). This reimbursable grant program is intended for relatively straightforward and simple projects. More advanced projects that require advanced engineering design may seek alternative funding opportunities, potentially through the county. • 75% state share grant with maximum award of $25,000; up to 10% state share for technical assistance

	• Maximum of $1,000 per cluster of 3-5 trees (best practice cap)
	• Implemented according to approved technical requirements (WDNR Fisheries Biologist) and complies with local shoreland zoning ordinances
	• Buffer area (350 ft^2) at base of coarse woody habitat cluster must comply with local shoreland zoning or:
	    o The landowner would need to commit to leaving the area un-mowed
	    o The landowner would need to implement a native planting (also cost share through this grant program available)
	• Coarse woody habitat improvement projects require a general permit from the WDNR
	• Landowner must sign Conservation Commitment pledge to leave project in place and provide continued maintenance for 10 years

Action Steps:	
1.	Recruit facilitator from Planning Committee (potentially same facilitator as previous management actions).
2.	Facilitator contacts WDNR Lakes Coordinator and WDNR Fisheries Biologist to gather information on initiating and conducting coarse woody habitat projects.
3.	The FLA will encourage property owners that have enhanced coarse woody habitat to serve as demonstration sites.

Management Goal 4: Increase the FLA's Capacity to Communicate with Lake Stakeholders and Facilitate Partnerships with Other Management Entities

Management Action:	Use education to promote lake protection and enjoyment through stakeholder education
Timeframe:	Continuation of current efforts
Facilitator:	FLA Board of Directors
Description:	Education represents an effective tool to address many lake issues. The FLA annually distributes a newsletter to its membership, maintains a closed Facebook Group, and has developed a website, which is the official repository of the FLA information. These mediums allow for communication with association members, but increasing the level of communication is important within a management group because it facilitates the spread of important association news, social events, educational topics, and dispels misconceptions. The FLA has a strong commitment to keeping Forest Lake healthy; therefore, it is a requirement of a FLA board member to contact any and all new lake property owners regarding the deed restrictions, the importance of maintaining the lake water quality, and any other pertinent information relating to the lake. The FLA seek to provide any new landowners around the lake with a binder of materials highlighting the important information about protecting Forest Lake. The FLA will continue to make the education of lake-related issues a priority. These may include educational materials, awareness events, and demonstrations for lake users as well as activities which solicit local and state government support. *Example Educational Topics* • History and summary of Forest Lake Lot Restrictions • Tribal spearing • Aquatic invasive species identification • Basic lake ecology • Impacts of drought and low water levels • Sedimentation • Boating safety (promote existing guidelines, Lake Use Information handout) • Swimmers itch • Shoreline habitat restoration and protection • Fireworks use and impacts to the lake • Noise and light pollution • Fishing regulations and overfishing • Minimizing disturbance to spawning fish • Recreational use of the lakes

Management Action:	Continue FLA's involvement with other entities that have responsibilities in managing Forest Lake
Timeframe:	Continuation of current efforts
Facilitator:	FLA Board of Directors
Description:	The waters of Wisconsin belong to everyone and therefore this goal of protecting and enhancing these shared resources is also held by other entities. Some of these entities are governmental while others organizations rely on voluntary participation. It is important that the FLA actively engage with all management entities to enhance the association's understanding of common management goals and to participate in the development of those goals. This also helps all management entities understand the actions that others are taking to reduce the duplication of efforts. Each entity will be specifically addressed in the table on the next page:
Action Steps:	
	See table guidelines on the next pages.

Table 5.0-1 Management Partner List.

Partner	Contact Person	Role	Contact Basis
Forest Lake Preservation Foundation	Bruce Smith Brucesmith1238@gmail.com	Source of funding for AIS control & education regarding topics to preserve and protect Forest Lake and its watershed	
Town of Land O' Lakes	Town Chair (Daniel Balog,715.617-0952, chairman@landolakeswi.gov)	Oversees ordinances, funding, and other items pertaining to town	Involved in lake management activities, monitoring, implementation, funding, volunteer recruitment. May be contacted regarding ordinance questions, and for information on community events.
Great Lakes Indian Fish and Wildlife Commission	General (715.682.6619)	Resource management within Ceded Territory	Collaborate on lake related studies, AIS management, inform of meetings, etc.
Vilas County Lakes & Rivers Association (VCLRA)	President (Tom Ewing, president@vclra.org)	Protects Vilas Co. waters through facilitating discussion and education.	Become aware of training or education opportunities, partner in special projects, or networking on other topics pertaining to Vilas Co. waterways.

Partner	Contact Person	Role	Contact Basis
Vilas County Land and Water Conservation Department	Lake Conservation Specialist (Mariquita (Quita) Sheehan, 715.479.3721,	Oversees conservation efforts for lake grants and projects.	Can provide assistance with shoreland restorations and habitat improvements. Assist in connecting FLA with other lake associations
Vilas County AIS Coordinator	Lake Conservation Specialist (Cathy Higley, 715.479.3738	Oversees AIS monitoring and education activities county-wide.	AIS training and ID, monitoring techniques, CBCW training, report summer activities.
Wisconsin Lakes	General staff (800.542.5253)	Facilitates education, networking and assistance on all matters.	Reps can assist on education
Wisconsin Department of Natural Resources	Fisheries Biologist (Eric Wegleitner, 715.356.5211 Ext: 246) eric.wegleitner@wisconsin.gov	Manages the fish populations and fish habitat enhancement efforts.	Stocking activities, scheduled surveys, survey results, volunteer opportunities for improving fishery.
	Lakes Coordinator (Kevin Gauthier) 715-365-8937 Kevin.GauthierSr@wisconsin.gov	Oversees management plans, grants, all lake activities.	Information on updating a lake management plan, submitting grants & permits, and to seek advice on other lake issues.
	Environmental Grant Specialist (Jill Sunderland, 608.358.9319)	Oversees financial aspects of grants.	Information on grant financials and reimbursement, CBCW grant applications.
	Conservation Warden Eagle River Ranger Station (Tim Price, 715.892.0054)	Oversees regulations handed down by the state.	Suspected violations, including fishing, boating safety, ordinance violations, etc.
	AIS Regional Coordinator Alan Wirt Alan.Wirt@wisconsin.gov	Oversees local AIS monitoring and prevention.	AIS training and ID, AIS monitoring techniques
	Citizen Lake Monitoring Network (Sandy Wickman – 715.365.8951, sandra.wickman@wisconsin.gov)	Provides information, training, and equipment for CLMN volunteers.	Contact of information regarding CLMN program, including training, equipment, and data entry into SWIMS

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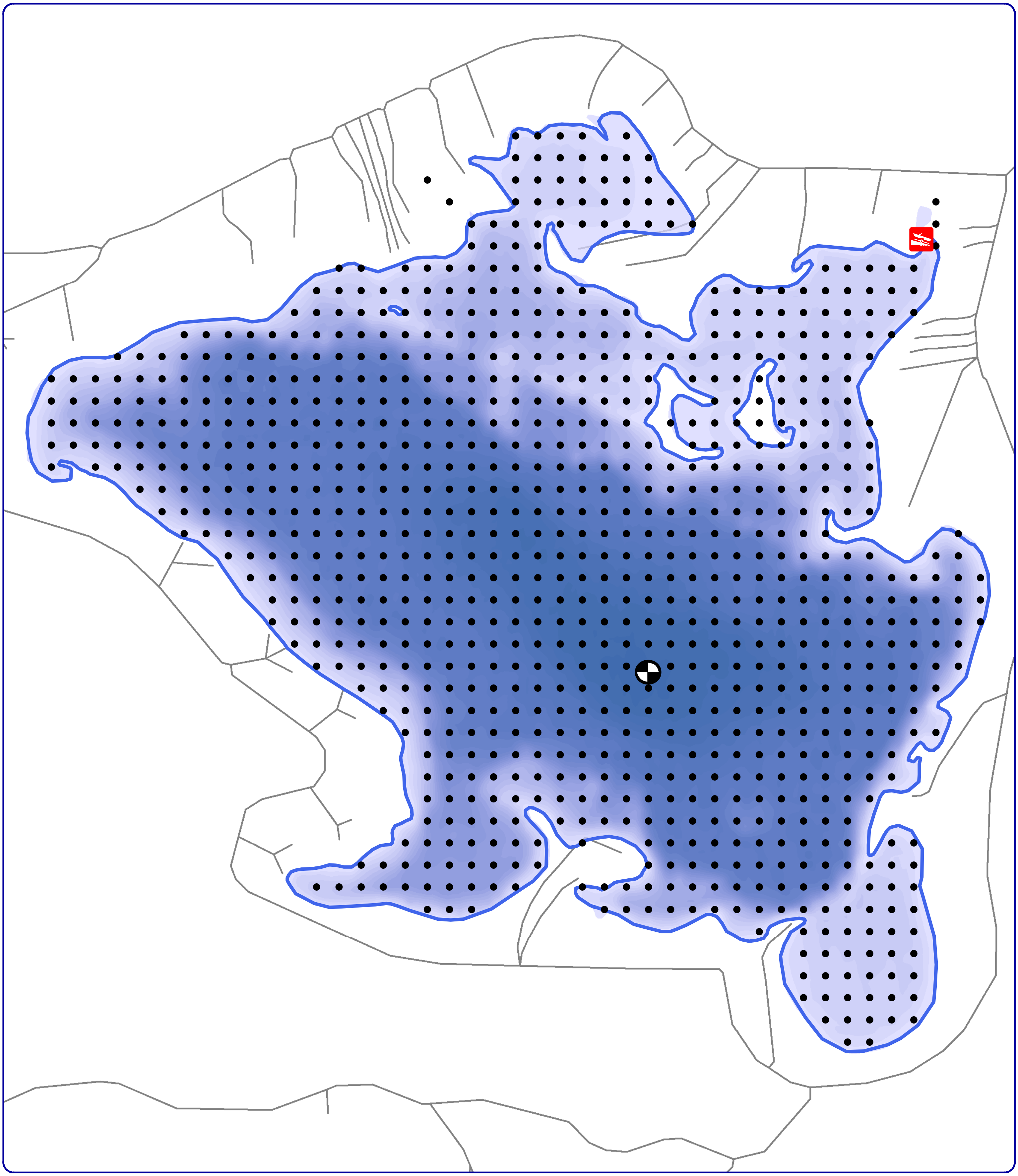
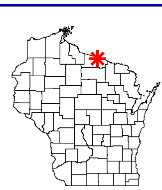


850

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
Map Date: November 13, 2023 JMB
Filename: Forest_Vilas_location.mxd



Project Location in Wisconsin

Legend

Forest Lake ~ 442 acres
Based upon 2013 Orthophotograpy

Public Boat Landing

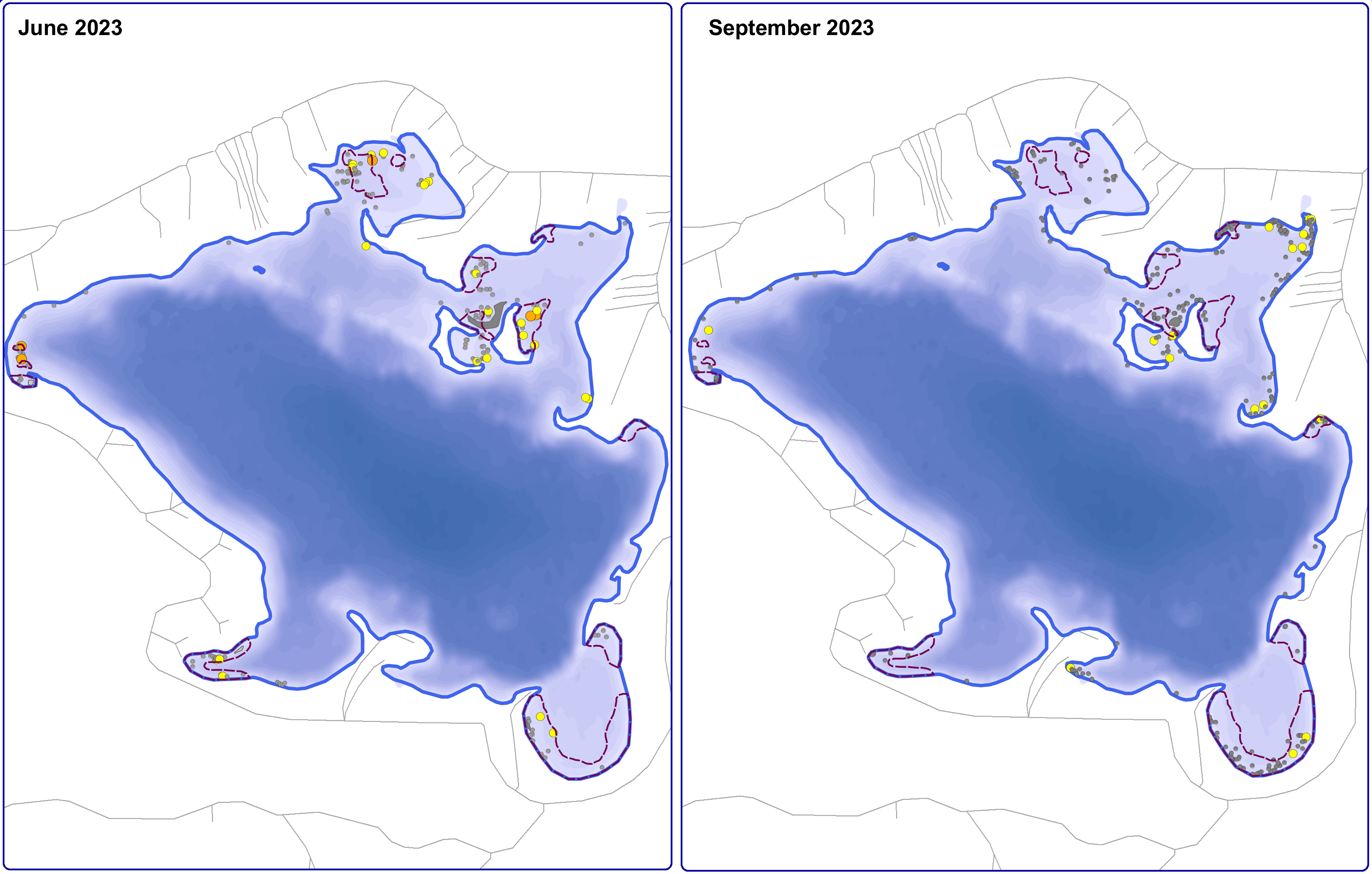
Water Quality
Sampling Location

Map 1

Forest Lake

Vilas County, Wisconsin

Project Location & Lake Boundary



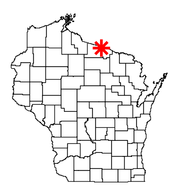
N
W E
S

1,300
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, 2016
Aquatic Plants: Onterra, 2023

Map Date: February 27th, 2024 KLW
Filename: ForestVilas_EWM_Compare2023.mxd



Project Location in Wisconsin

Legend

Eurasian watermilfoil (June & September, 2023)

- Highly Scattered
- Scattered
- Dominant
- Highly Dominant
- Surface Matting
- Single or Few Plants
- Clumps of Plants
- Small Plant Colony
- Public Boat Landing
- 2023 DASH Site

Map 2
Forest Lake
Vilas County, Wisconsin

June & September 2023 Eurasian watermilfoil Survey Results

A

APPENDIX A

Public Participation Materials

Management Planning Meeting I Presentation Materials

Forest Lake Association & Forest Lake Preservation Foundation

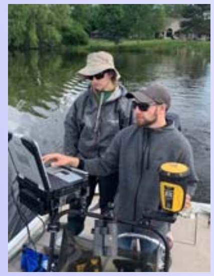
Management Plan Update Planning Meeting

April 22, 2024

Todd Hanke
Onterra LLC
Lake Management Planning

Presentation Outline

- Introduction to Onterra
- Lake Management Planning
- Survey Results
  - Water Quality (Limited)
  - Watershed (Limited)
  - Aquatic Plants
- EWM Management & Monitoring
- Updating the Management Plan



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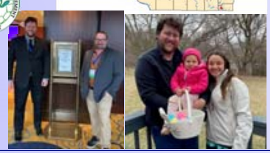
Onterra, LLC

- Founded in 2005, HQ in De Pere, WI
- Staff
  - Three aquatic ecologists
  - One paleoecologist
  - Four full-time field technicians
  - Four summer interns
- Services
  - Science and planning
- Philosophy
  - Promote realistic planning
  - Assist, not direct



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Lake Management Planning

What is a Lake Management Plan?

- Many organizations may have "plans" for managing Forest Lake and its watershed
- The FLA's *Comprehensive Management Plan* for managing Forest Lake was finalized in January 2019
  - Based upon FLA capacity
  - Addresses your concerns
- Long-term & useable plan (~10 years)
- Living plan subject to revision over time

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Management Plan and Grants

- WDNR recommends ***Comprehensive Management Plans*** generally get updated every 10 years
  - Aquatic Plant Management (APM) Plan is one component of a Comprehensive Plan, along with water quality, watershed, shoreland, fisheries, etc.
  - Particularly for grants/permits related to water quality/watershed improvements
- WDNR recommends lakes conducting active plant management update aspects of the plan every 5 years (***APM Plan***)
  - Particularly for grants/permits related to aquatic plant management (AIS control grants, NR107, NR109)
  - Updates management goals and actions to be consistent with changing BMP's, incorporates knowledge gained from past APM activities on the lake
  - Management action in AIS Grant needs to be supported by Plan
- Annual AIS Control Plan
  - Consistent with the framework outlined in APM Plan
  - Includes specific plans, delineated prioritized areas and description of monitoring components

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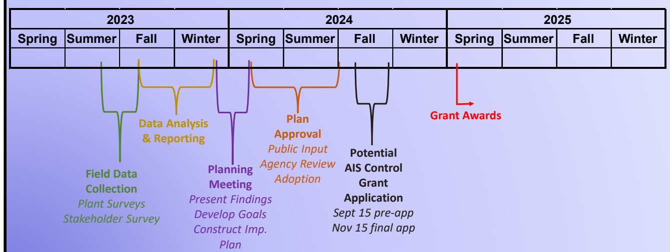
Management Planning Timeline



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Lake Management Planning

Aquatic Plant Management Plan Outline

- 1.0 Introduction
- 2.0 Stakeholder Participation
- 3.0 Results
  - 3.1 Water Quality
  - 3.2 Watershed Assessment
  - 3.3 Aquatic Plants
  - 3.4 Non-native Aquatic Plants
- 4.0 Summary & Conclusions
- 5.0 Implementation Plan
- 6.0 Literature Cited

Plan Mtg

Update Mngmt Goals/Actions Meeting

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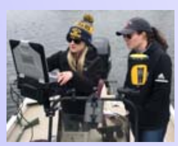
Lake Management Planning

Management Plan Update – Data Collection

- 2023 FLA & Riparian Survey (Summer 2023)
  - 82 Sent, 51 returned = 62%
  - Included questions on WQ, AIS, other
- 2023 Aquatic Plant Surveys
  - Early Season AIS
  - Late-Summer EWM Mapping Survey
  - Utilizes 2022 Whole-lake Point Intercept Data
- 2023 Water Quality Assessment
  - CLMN Ongoing
  - Limited additional sampling – spring & summer
- Watershed Assessment
  - Landcover update, comparison



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Water Quality

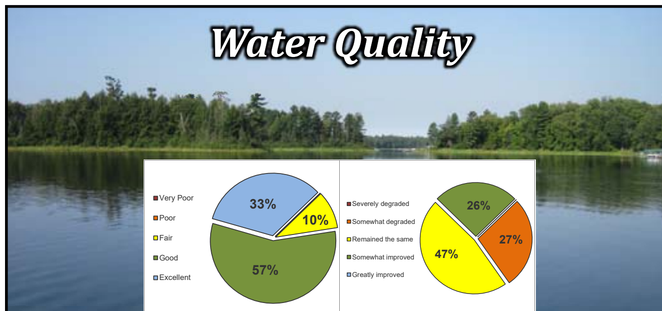


Figure 3.1-11. Stakeholder survey response Question #17. How would you describe the current water quality of Forest Lake?

Figure 3.1-12. Stakeholder survey response Question #18. How has the water quality changed in Forest Lake since you first visited the lake?

Introduction to Lake Water Quality

Phosphorus
Naturally occurring & essential for all life
Regulates phytoplankton biomass in most WI lakes
Most often 'limiting plant nutrient' (shortest supply)
Human development often increases P delivery to lakes

N:P Ratio: 32:1 – Phosphorus

Chlorophyll-*a*
Pigment used in photosynthesis
Used as surrogate for phytoplankton biomass

Secchi Disk Transparency
Measure of water clarity
Measured using a Secchi disk

Total Phosphorus

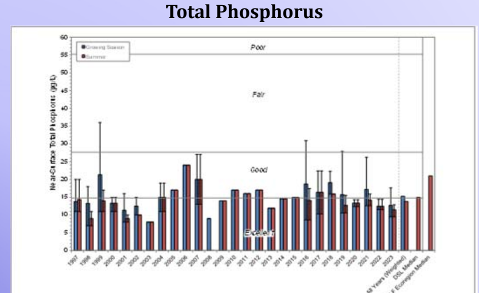
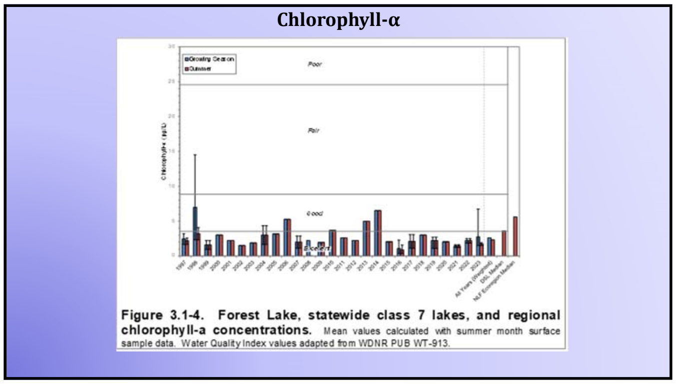
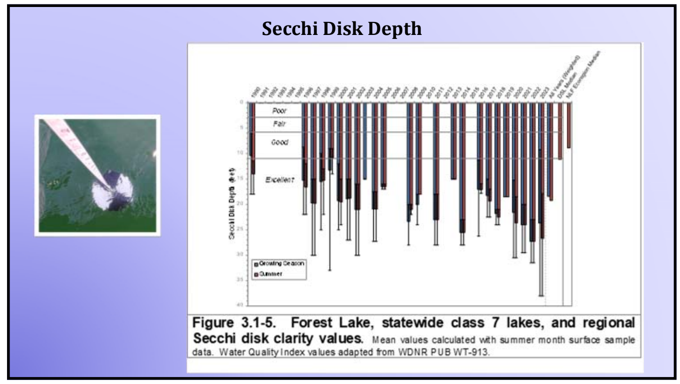
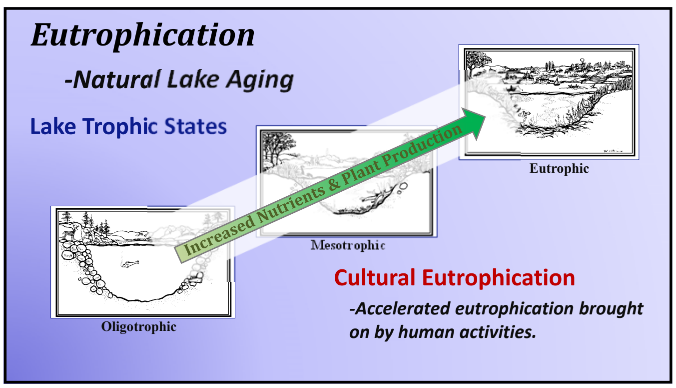
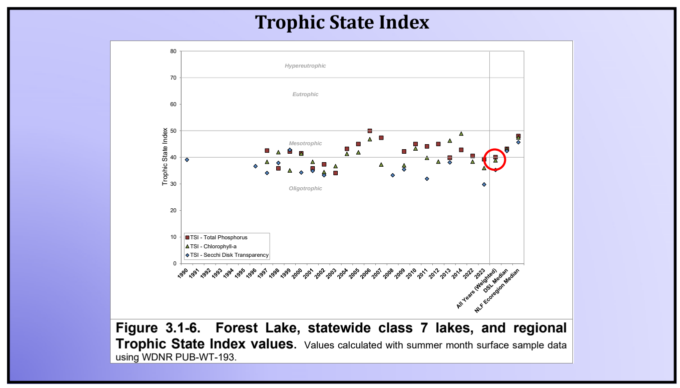


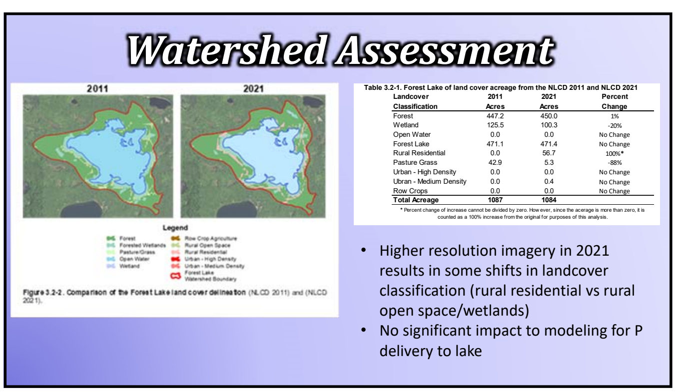
Figure 3.1-3. Forest Lake, statewide class 7 lakes, and regional total phosphorus concentrations. Mean values calculated with summer month surface sample data. Water Quality Index values adapted from WDNR PUB WT-913.


Chlorophyll-α
Poor
Fair
Good
Figure 3.1-4. Forest Lake, statewide class 7 lakes, and regional chlorophyll-a concentrations. Mean values calculated with summer month surface sample data. Water Quality Index values adapted from WDNR PUB WT-913.


Secchi Disk Depth
Poor
Fair
Good
Excellent
Figure 3.1-5. Forest Lake, statewide class 7 lakes, and regional Secchi disk clarity values. Mean values calculated with summer month surface sample data. Water Quality Index values adapted from WDNR PUB WT-913.


Eutrophication
-Natural Lake Aging
Lake Trophic States
Increased Nutrients & Plant Production
Oligotrophic
Mesotrophic
Eutrophic
Cultural Eutrophication
-Accelerated eutrophication brought on by human activities.

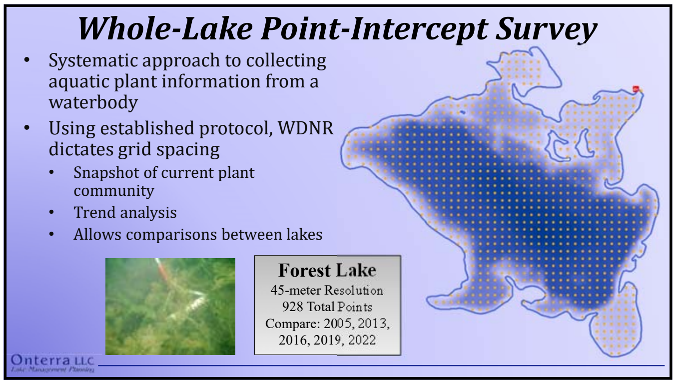

Trophic State Index
Figure 3.1-6. Forest Lake, statewide class 7 lakes, and regional Trophic State Index values. Values calculated with summer month surface sample data using WDNR PUB-WT-193.


Watershed Assessment
2011
2021
Legend
Figure 3.2-2. Comparison of the Forest Lake land cover delineation (NLCD 2011) and (NLCD 2021).
Table 3.2-1. Forest Lake of land cover acreage from the NLCD 2011 and NLCD 2021

Landcover Classification	2011 Acres	2021 Acres	Percent Change
Forest	447.2	450.0	1%
Wetland	125.5	100.3	-20%
Open Water	0.0	0.0	No Change
Forest Lake	471.1	471.4	No Change
Rural Residential	0.0	56.7	100%*
Pasture Grass	42.9	5.3	-88%
Urban - High Density	0.0	0.0	No Change
Ubran - Medium Density	0.0	0.4	No Change
Row Crops	0.0	0.0	No Change
Total Acreage	1087	1084	

* Percent change of increase cannot be divided by zero. However, since the acerage is more than zero, it is counted as a 100% increase from the original for purposes of this analysis.
• Higher resolution imagery in 2021 results in some shifts in landcover classification (rural residential vs rural open space/wetlands)
• No significant impact to modeling for P delivery to lake


Aquatic Plants


Whole-Lake Point-Intercept Survey
Systematic approach to collecting aquatic plant information from a waterbody
Using established protocol, WDNR dictates grid spacing
Snapshot of current plant community
Trend analysis
Allows comparisons between lakes
Forest Lake
45-meter Resolution
928 Total Points
Compare: 2005, 2013, 2016, 2019, 2022
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Aquatic Plant Species List
-50+ unique native species
- 1 Non-Native Species – Eurasian watermilfoil
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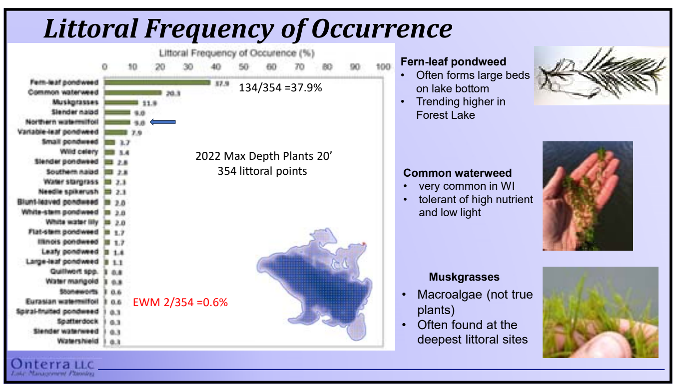
Littoral Frequency of Occurrence
Littoral Frequency of Occurence (%)
134/354 =37.9%
2022 Max Depth Plants 20’
354 littoral points
EWM 2/354 =0.6%
Fern-leaf pondweed
Often forms large beds on lake bottom
Trending higher in Forest Lake
Common waterweed
very common in WI
tolerant of high nutrient and low light
Muskgrasses
Macroalgae (not true plants)
Often found at the deepest littoral sites
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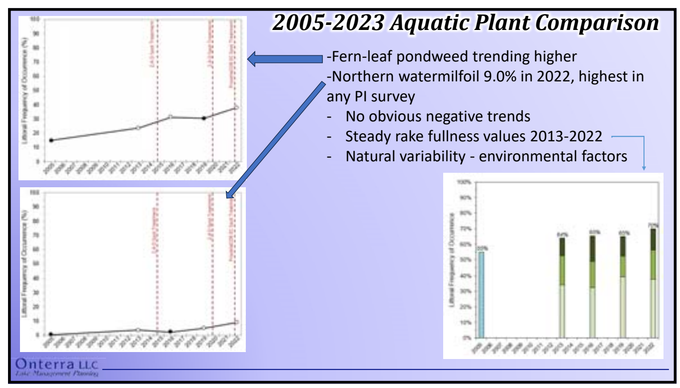
2005-2023 Aquatic Plant Comparison
-Fern-leaf pondweed trending higher
-Northern watermilfoil 9.0% in 2022, highest in any PI survey
- No obvious negative trends
- Steady rake fullness values 2013-2022
- Natural variability - environmental factors
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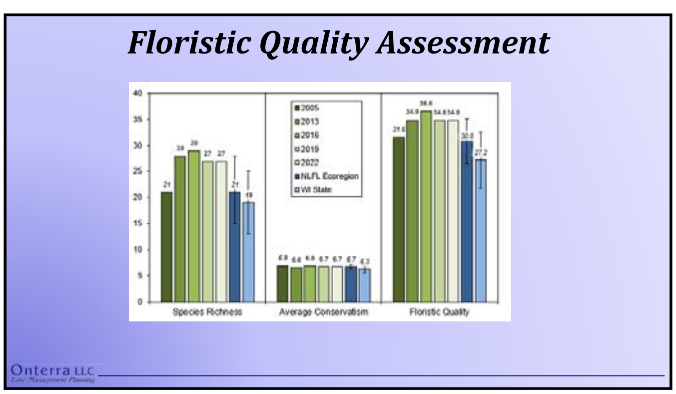
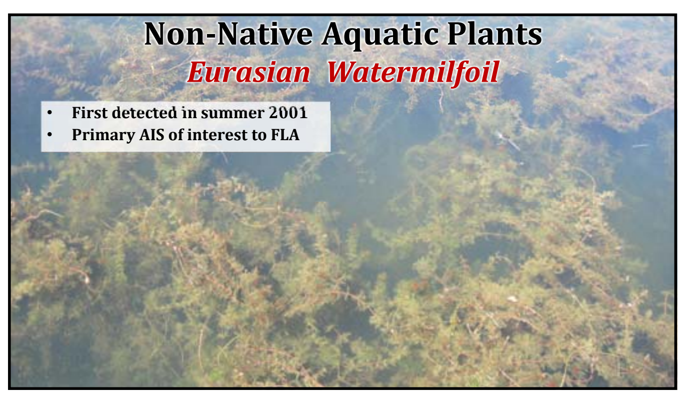

Floristic Quality Assessment
2005
2013
2016
2019
2022
NLFL Ecoregion
WI State
Species Richness
Average Conservatism
Floristic Quality
Onterra LLC


Non-Native Aquatic Plants
Eurasian Watermilfoil
First detected in summer 2001
Primary AIS of interest to FLA

EWM Propagation

- **Produces seed, but low viability**
- **Spread primarily through fragments, a vegetative clone**
- **Ability to manage spread from fragments is overstated**

Auto-fragment
- Purposefully produced
- High energy storage
- Higher viability

Allo-fragment
- Mechanical breakage
- Low energy storage
- Lower viability

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Types of Aquatic Plant Surveys

Quantitative
- Point-Intercept Survey
  - Numeric & systematic
  - Applied at various scales

Qualitative
- EWM Mapping Surveys
  - Fine-scale location accuracy
  - Subjective designations

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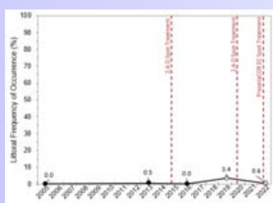
EWM in Point Intercept Surveys

2023 PI Survey: 2/354 Littoral Sampling Sites = 0.6%



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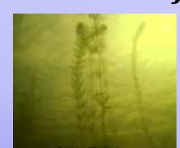
Professional AIS Mapping



Point-Based Mapping
- Single or Few Plants
- Clumps of Plants
- Small Plant Colony

Polygon-Based Mapping
- Highly Scattered
- Scattered
- Dominant
- Highly Dominant
- Surface Matting

True Colonies (Dominant, Highly Dominant, Surface Matting)

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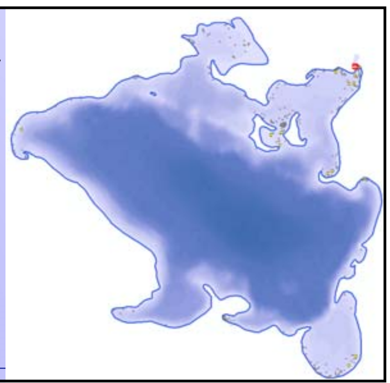
September 14, 2023 EWM Mapping Survey

- Single or Few Plants
- Clumps of Plants
- Small Plant Colony

- Highly Scattered
- Scattered
- Dominant
- Highly Dominant
- Surface Matting



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EWM Population Progression

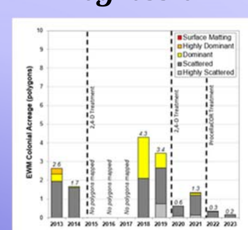
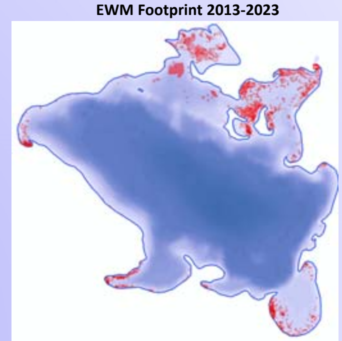


Figure 3.4-4. Acres of colonized EWM (polygons) from 2013-2023 in Forest Lake.

EWM Footprint 2013-2023



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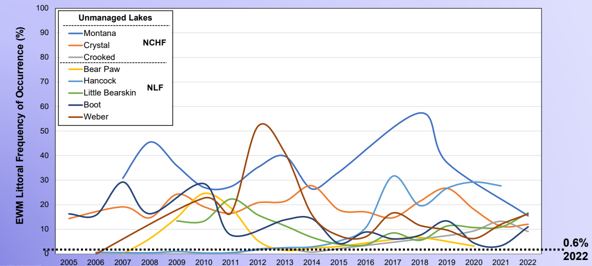
WDNR EWM Long-Term Monitoring Trends

Unmanaged



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EWM Management Perspectives

1. **No Coordinated Active Management (Let Nature Take its Course)**
   - Group does not organize or fund management efforts
   - Monitor population
2. **Reduce AIS Population on a lake-wide level (Population Management – "Control")**
   - Will not *eradicate* EWM
   - Early populations may be targeted with manual removal efforts, established populations may need to entertain herbicide treatment (risk assessment)
   - Set triggers (thresholds) of implementation and tolerance
3. **Minimize navigation and recreation impediment (Nuisance Control)**
   - Often accomplished through mechanical harvesting or herbicide treatment, limited applicability for hand harvesting
   - Prioritize areas based on human use & EWM density

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Manual Removal – *Hand Harvesting & DASH*

- **Goal – to manage the EWM population or nuisance control**

EWM population	nuisance control
Initial populations Low density & isolated occurrences Follow-up after treatments	In riparian footprint Navigation lanes or small areas

- **Removal of entire root material required for mortality**
- **Scale limitations, not for large or dense areas**
- **Diver-Assisted Suction Harvest (DASH) can increase efficiency**
- **Limitations**
  - –Density of EWM & native plants
  - –Clarity of water
  - –Sediment type
  - –Obstructions


Photo credit Anvil Lake Association

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Mechanical Harvesting

- **Goal – to restore aspects of use and aesthetics**
- **Cuts and removes plant biomass; does not cause mortality**
- **Suitable for large and dense EWM**
- **Applied as clear-cutting or confined to lanes**
- **Concern for spread of EWM is overstated**
- **Risk of bi-catch**
  - –Native plants
  - –Fish & amphibians
  - –Insects, small animals


Photo Credit: Aquatic Plant Management, LLC

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Herbicide Treatment

- **Goal – multi-year EWM population control**
- **Meet concentration & exposure times (CETs) for mortality**
  - –Spot vs whole-lake/basin treatments
  - –Small (< 5 acres) spot treatments are often ineffective
  - –Protected areas more effective
- **Introduces greater need for risk assessment discussion**
  - –Impacts to native plants, particularly native watermilfoils and select dicots
  - –Potential impacts to early life stages of select fish species (i.e. walleye)
  - –Unknown impacts


Photo Credit: Schmidt's Aquatic, LLC

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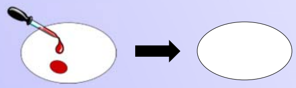
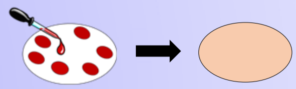
Ecological Definitions of Herbicide Treatment

Spot Treatment: Herbicide applied at a scale where dissipation will not result in significant lake wide concentrations; impacts are anticipated to be localized to in/around application area.



Whole-Lake (basin-wide) Treatment: Herbicide applied at a scale where dissipation will result in significant lake wide concentrations; impacts are anticipated to be within a defined Area of Potential Impact (AOPI).



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Lake Management Planning

Recent EWM Herbicide Treatment History

Year	AIS Management	Acres Treated	Chemical
2013	DASH/HH		
2014	DASH/HH		
2015	Herbicide Treatment	8.1	2,4-D
2016	None		
2017	None		
2018	DASH/HH		
2019	DASH/HH		
2020	Herbicide Treatment & DASH/HH	14.34	2,4-D
2021	DASH/HH		
2022	Herbicide Treatment and DASH/HH	2.8	florpyrauxifen-benzyl
2023	DASH/HH		

- 2020 & 2022 Treatments were designed as basin-wide strategies

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Recent Hand Harvesting Management History

Professional Hand Harvesting and/or Diver Assisted Suction Harvest			
Year	AIS Management	Harvest Yield (cubic ft)	Days
2018	DASH/HH	129.25	3
2019	DASH/HH	49.5	3
2020	DASH/HH	31	6
2021	DASH/HH	218	10
2022	DASH/HH	228.5	7
2023	DASH/HH	283.5	17
Totals		939.75	46

- IPM Strategy, follow up after herbicide treatment
- Suppress population in other areas
- Population management
- Annual monitoring, re-prioritization of efforts

AQUATIC PLANT MANAGEMENT

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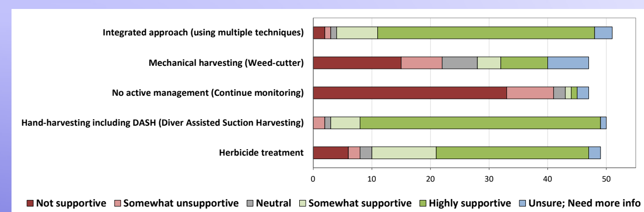
Lake Management Planning

Stakeholder Perceptions of EWM Management

23. The Forest Lake Association is currently assessing future techniques for continuing to manage the EWM population. What is your level of support for the future use of the following EWM management techniques in Forest Lake? Please select one response for each control technique.



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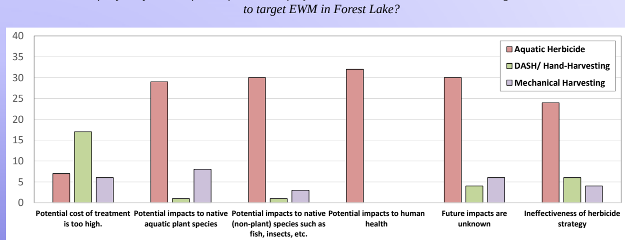
Lake Management Planning

Stakeholder Perceptions of EWM Management

24. What concerns, if any, do you have for the future use of aquatic herbicides, DASH/hand harvesting, or Mechanical Harvesting to target EWM in Forest Lake?



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Lake Management Planning

Updating Lake Management Plan

FLA 2019 Comp Management Plan

Management Goal #1: ***Control Existing & Prevent Further Aquatic Invasive Species Infestations within Forest Lake***

Action: Continue with CBCW inspection at public access

Action: Coordinate annual volunteer monitoring for EWM

Action: Conduct EWM population control using hand harvesting (includes DASH) and/or herbicide spot treatment (herbicide use threshold in place)

Action: Conduct periodic quantitative vegetation monitoring (Point-Intercept 3-5 yrs) Community Mapping Survey 7-10 yrs

Action: Initiate rapid response plan following detection of new AIS

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FLA 2019 Comp Management Plan

Management Goal #2: ***Maintain Current Water Quality Conditions***

Action: Monitor WQ through WDNR CLMN Program

Management Goal #3: Improve Lake & Fishery Resource

Action: Educate stakeholders on importance of shoreland condition and shoreland restoration on Forest Lake

Action: Coordinate w/WDNR and private landowners to expand coarse woody habitat in Forest Lake

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FLA 2019 Comp Management Plan

Management Goal #4: ***Increase the FLA's capacity to communicate with lake stakeholders and facilitate partnerships with other management entities***

Action: Use education to promote lake protection and enjoyment through stakeholder education

Action: Continue FLA's involvement with other management entities that have responsibilities in managing Forest Lake

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Thank You

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Lake Management Planning

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B

APPENDIX B

Stakeholder Survey Response Charts and Comments

Forest Lake - Anonymous Stakeholder Survey

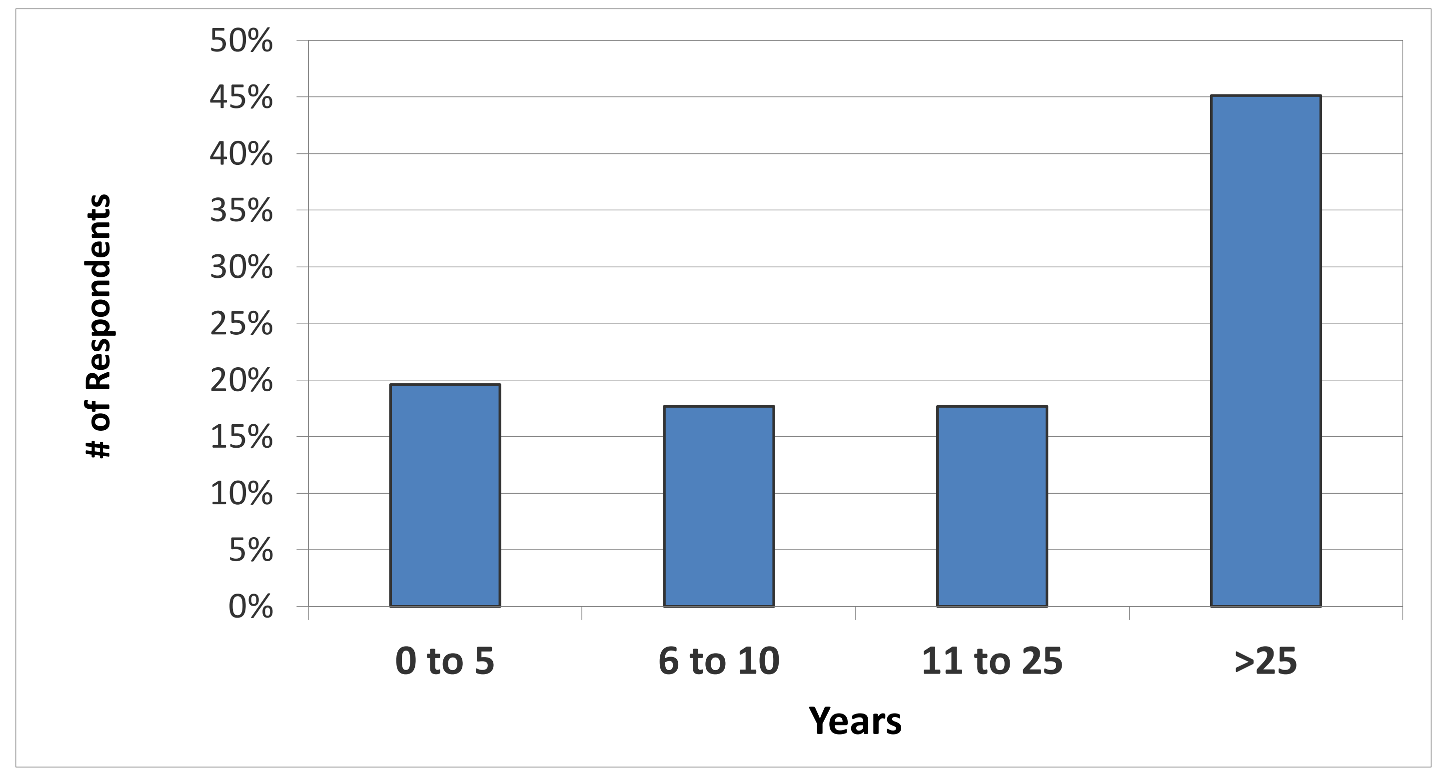
Surveys Distributed: *82*
Surveys Returned: *51*
Response Rate: *62%*

Forest Lake Property

1. How many years have you owned or rented your property on or near Forest Lake?

Answer Options	Response Count
	51
answered question	**51**
skipped question	**0**

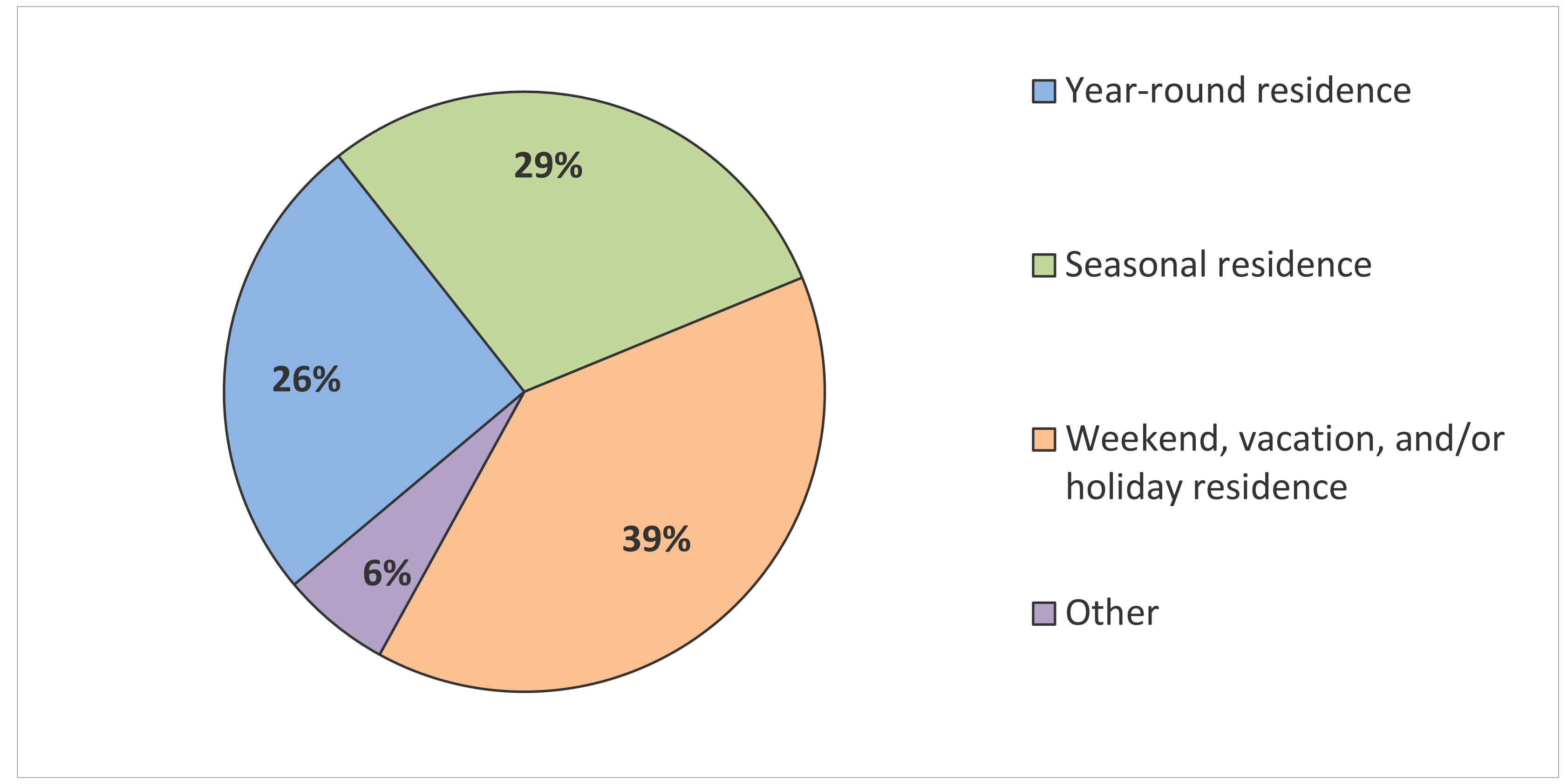
Category (# of years)	Responses	% Response
0 to 5		**20%**
6 to 10		**18%**
11 to 25		**18%**
>25		**45%**



2. How is your property on or near Forest Lake used?

Answer Options	Response Percent	Response Count
Year-round residence	25%	13
Seasonal residence	29%	15
Weekend, vacation, and/or holiday residence	39%	20
Undeveloped	0%	0
Other	6%	3
	answered question	**51**
	skipped question	**0**

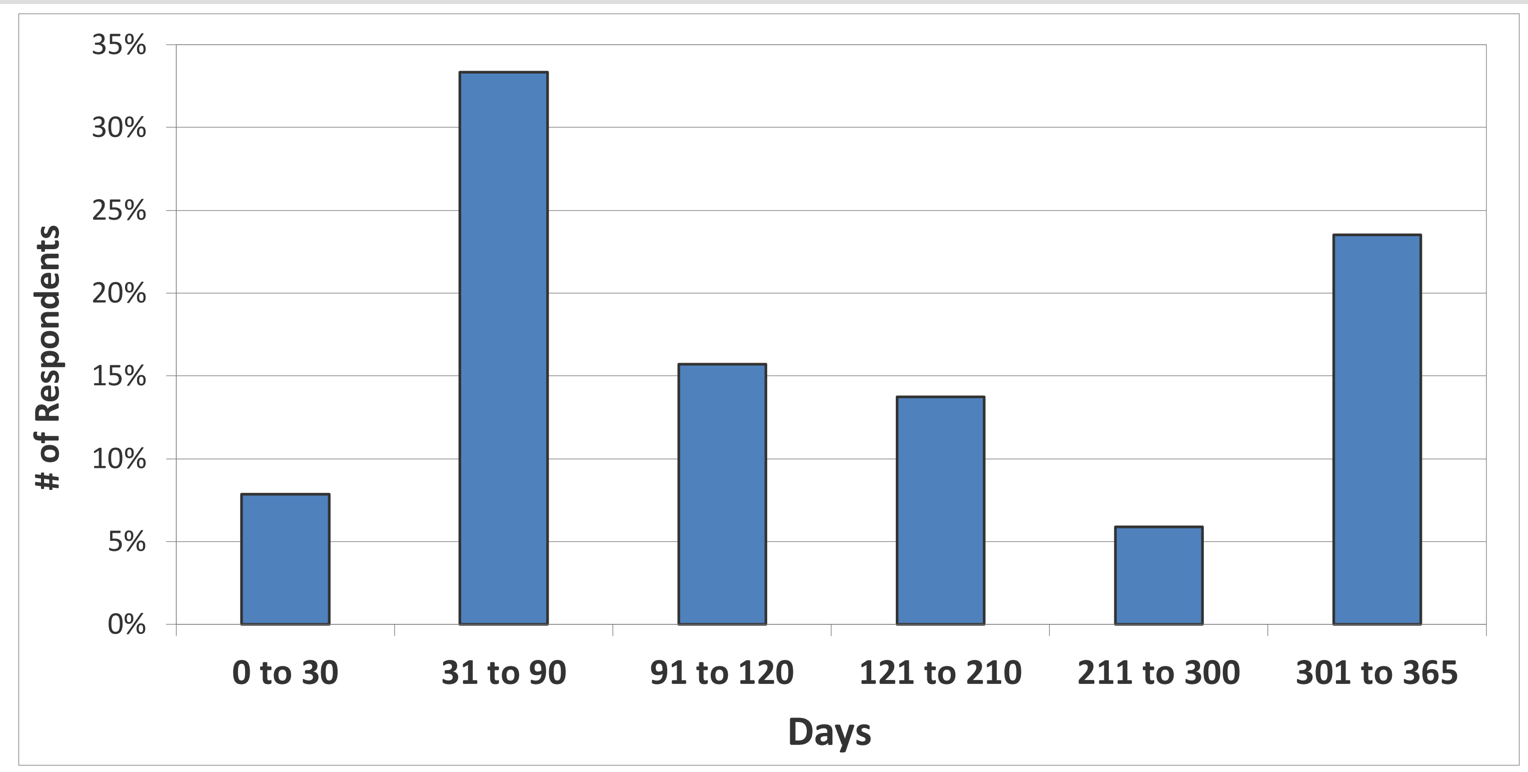
Number	"Other" responses
1	Year round, not continuous
2	Once each month for approximately a week, plus or minus.
3	1-3 weeks 6 times as year



3. Considering the past three years, how many days each year is your property used by you or others?

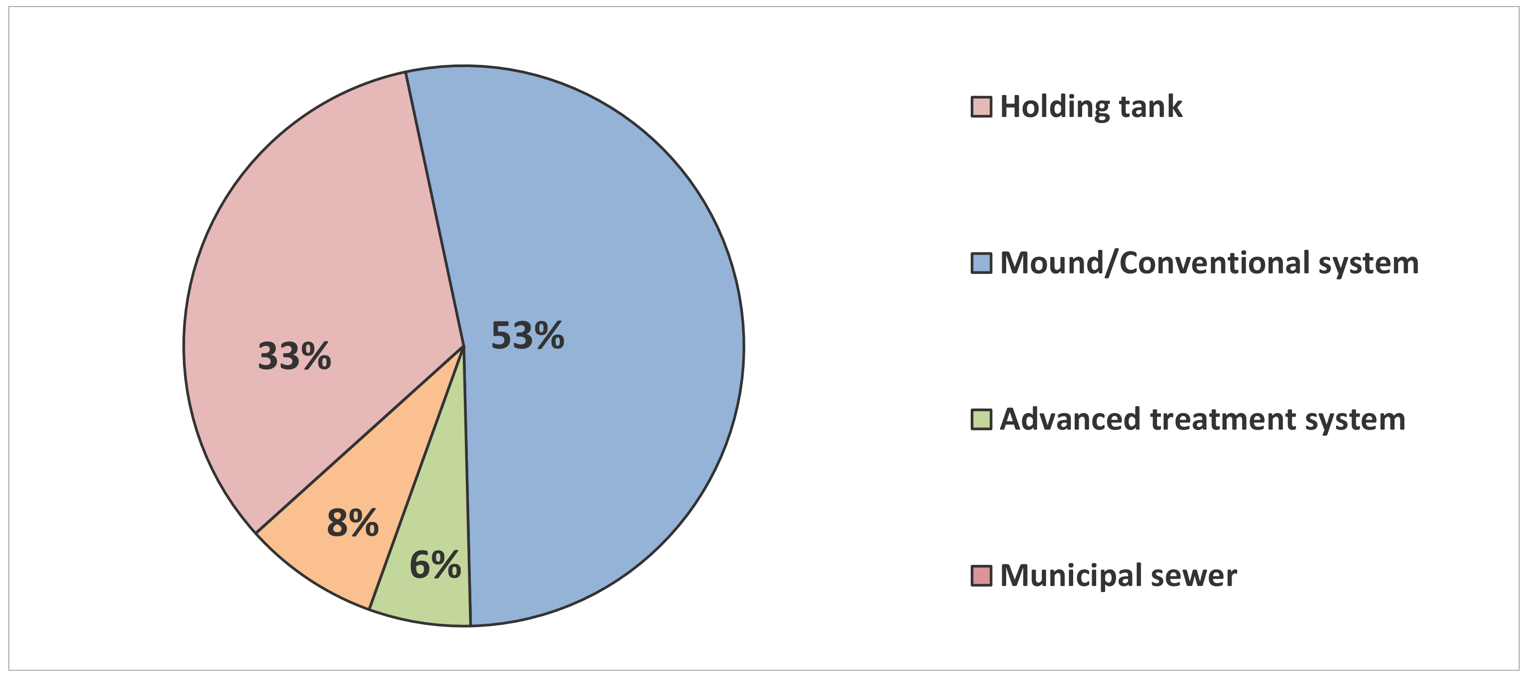
	Response Count
answered question	**51**
skipped question	**0**

Category (# of days)	Responses	%
0 to 30		**8%**
31 to 90		**33%**
91 to 120		**16%**
121 to 210		**14%**
211 to 300		**6%**
301 to 365		**24%**



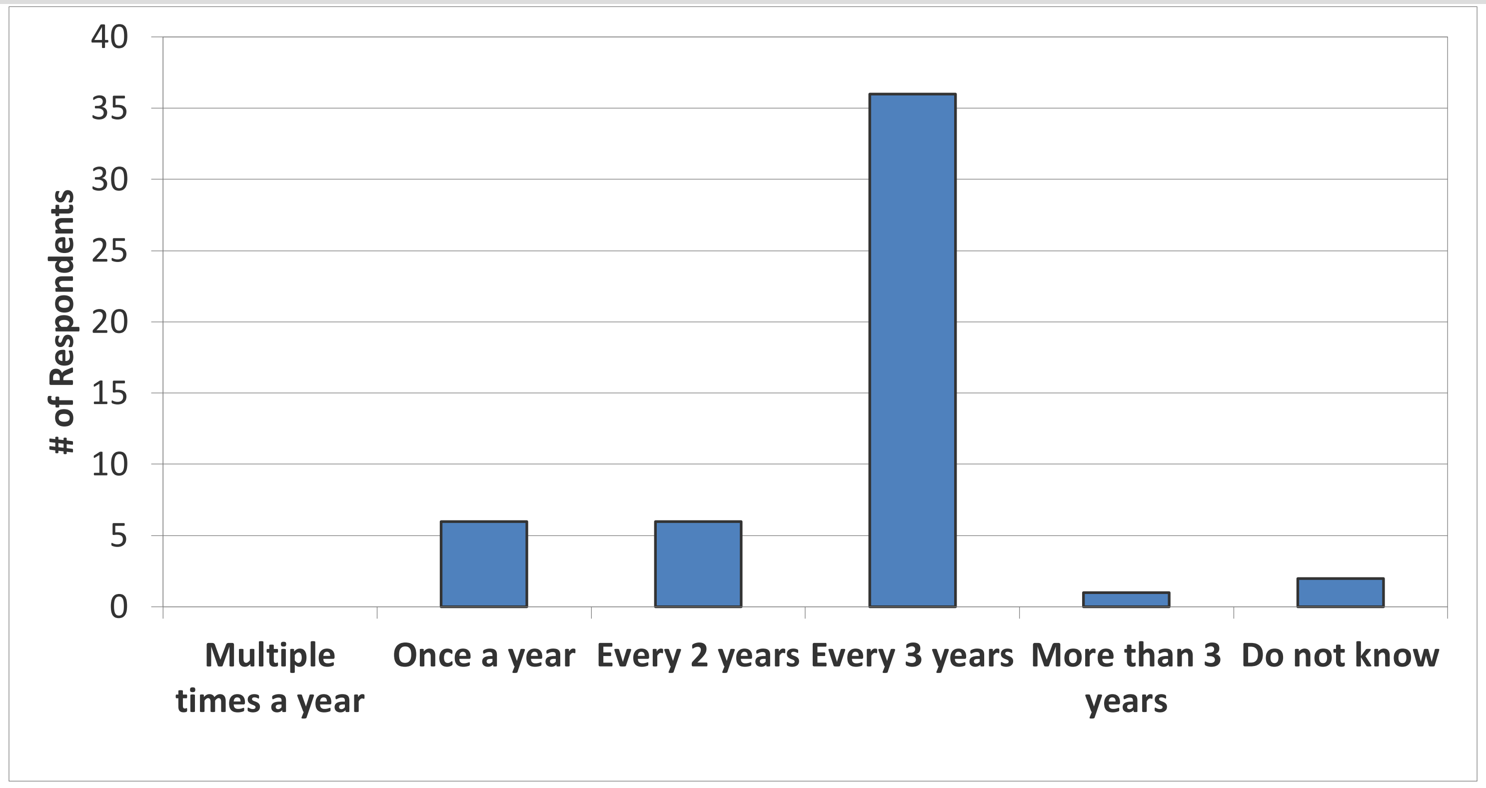
4. What type of septic system does your property have?

Answer Options	Response Percent	Response Count
Holding tank	33%	17
Mound/Conventional system	53%	27
Advanced treatment system	6%	3
Municipal sewer	0%	0
Do not know	8%	4
No septic system	0%	0
	answered question	**51**
	skipped question	**0**



5. How often is the septic system on your property pumped?

Answer Options	Response Percent	Response Count
Multiple times a year	0%	0
Once a year	12%	6
Every 2 years	12%	6
Every 3 years	71%	36
More than 3 years	2%	1
Do not know	4%	2
	answered question	**51**
	skipped question	**0**

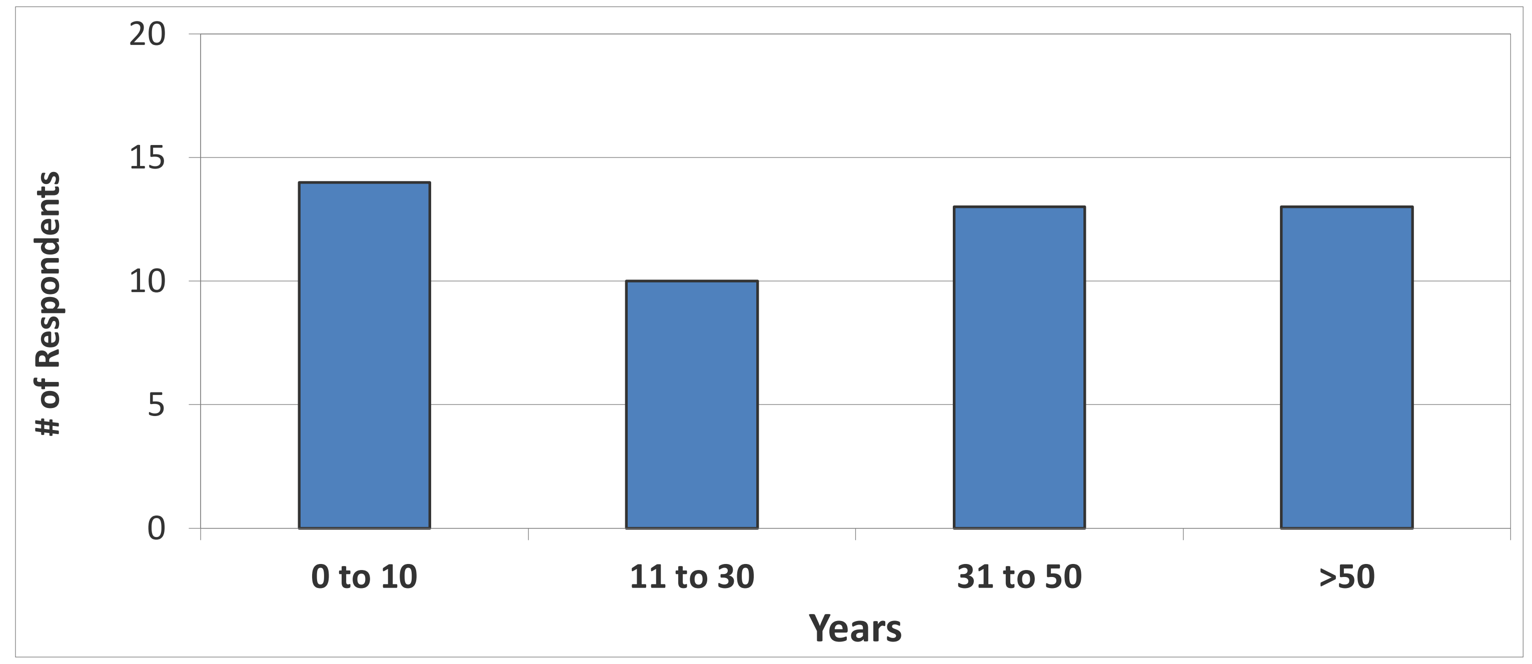


Recreational Activity on Forest Lake

6. How many years ago did you first visit Forest Lake?

Answer Options	Response Count
answered question	50
skipped question	1

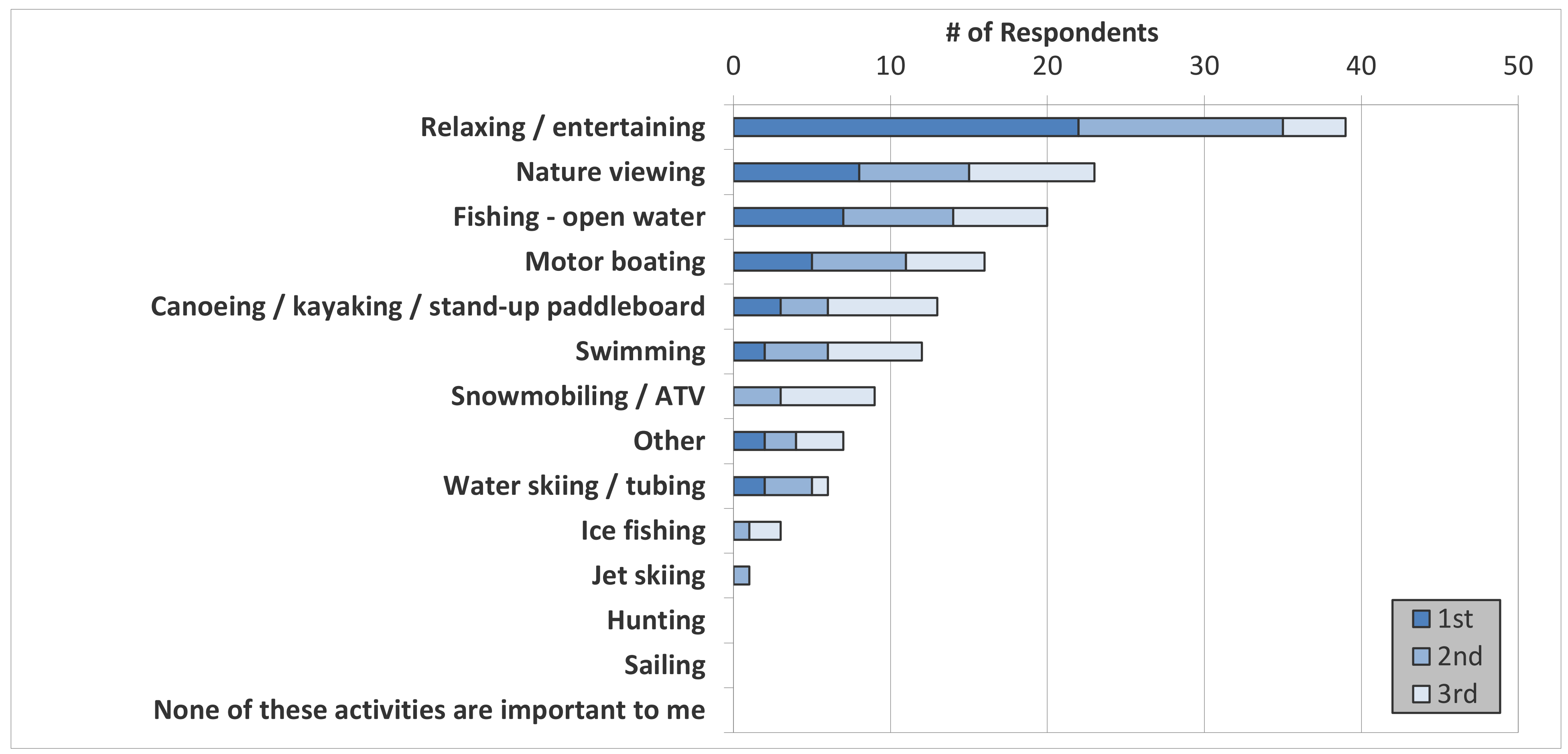
Category (# of years)	Response Percent	Response Count
0 to 10	28%	14
11 to 30	20%	10
31 to 50	26%	13
>50	26%	13



7. Please rank up to three activities that are important reasons for owning your property on or near Forest Lake, with 1 being the most important.

Answer Options	1st	2nd	3rd	Rating Average	Response Count
Relaxing / entertaining	22	13	4	1.54	39
Nature viewing	8	7	8	2	23
Fishing - open water	7	7	6	1.95	20
Motor boating	5	6	5	2	16
Canoeing / kayaking / stand-up paddleboard	3	3	7	2.31	13
Swimming	2	4	6	2.33	12
Snowmobiling / ATV	0	3	6	2.67	9
Other	2	2	3	2.14	7
Water skiing / tubing	2	3	1	1.83	6
Ice fishing	0	1	2	2.67	3
Jet skiing	0	1	0	2	1
Hunting	0	0	0	0	0
Sailing	0	0	0	0	0
None of these activities are important to me	0	0	0	0	0
				answered question	**51**
				skipped question	**0**

Number	"Other" responses
1	To Live here
2	Passed down from relatives
3	Pristine, quiet beauty.
4	Exercising
5	All of the above water activities.
6	Water sports
7	Family memories
8	gardening and family memories

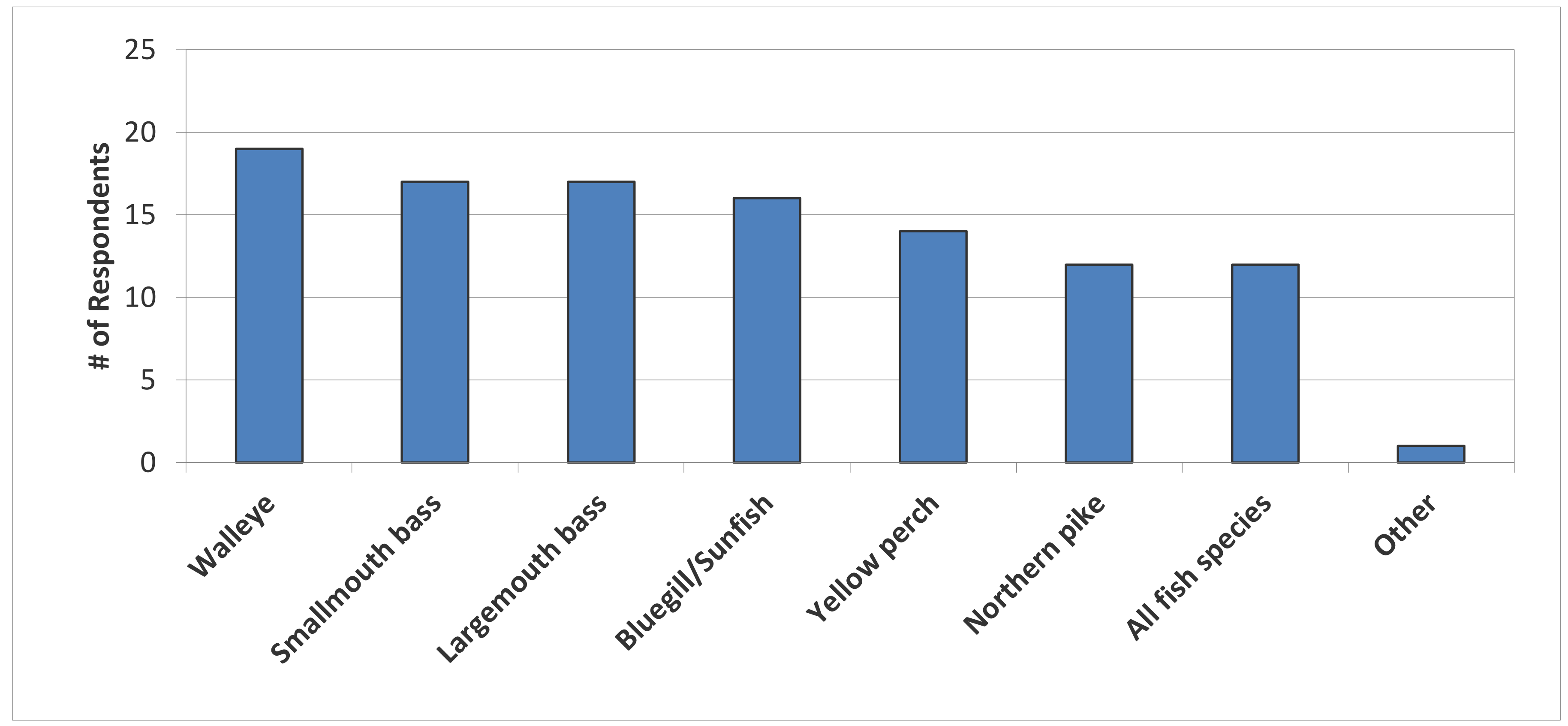


8. Have you personally fished on Forest Lake in the past three years?

Answer Options	Response Percent	Response Count
Yes	73%	37
No	27%	14
	answered question	**51**
	skipped question	**0**

9. What species of fish do you try to catch on Forest Lake?

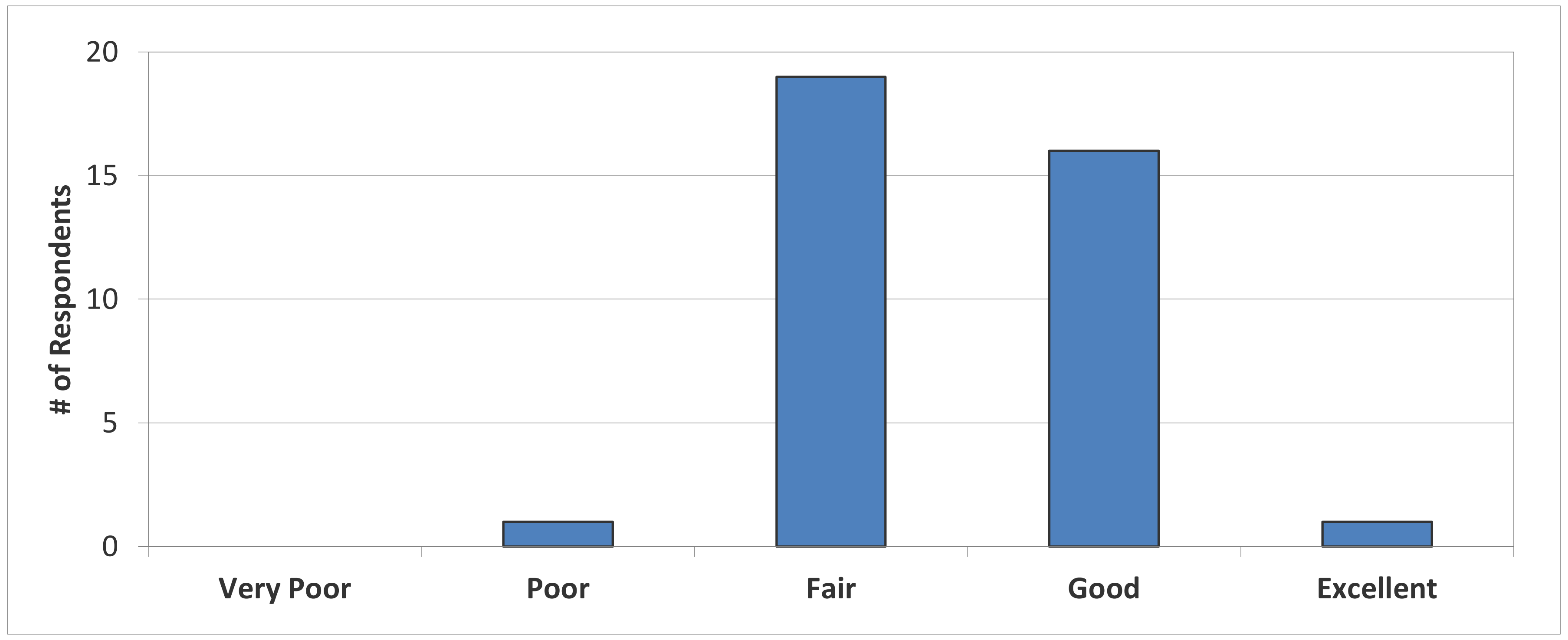
Answer Options	Response Percent	Response Count
Walleye	51%	19
Smallmouth bass	46%	17
Largemouth bass	46%	17
Bluegill/Sunfish	43%	16
Yellow perch	38%	14
Northern pike	32%	12
All fish species	32%	12
Other	3%	1
	answered question	**37**
	skipped question	**14**



Number	"Other" responses
1	Rock bass

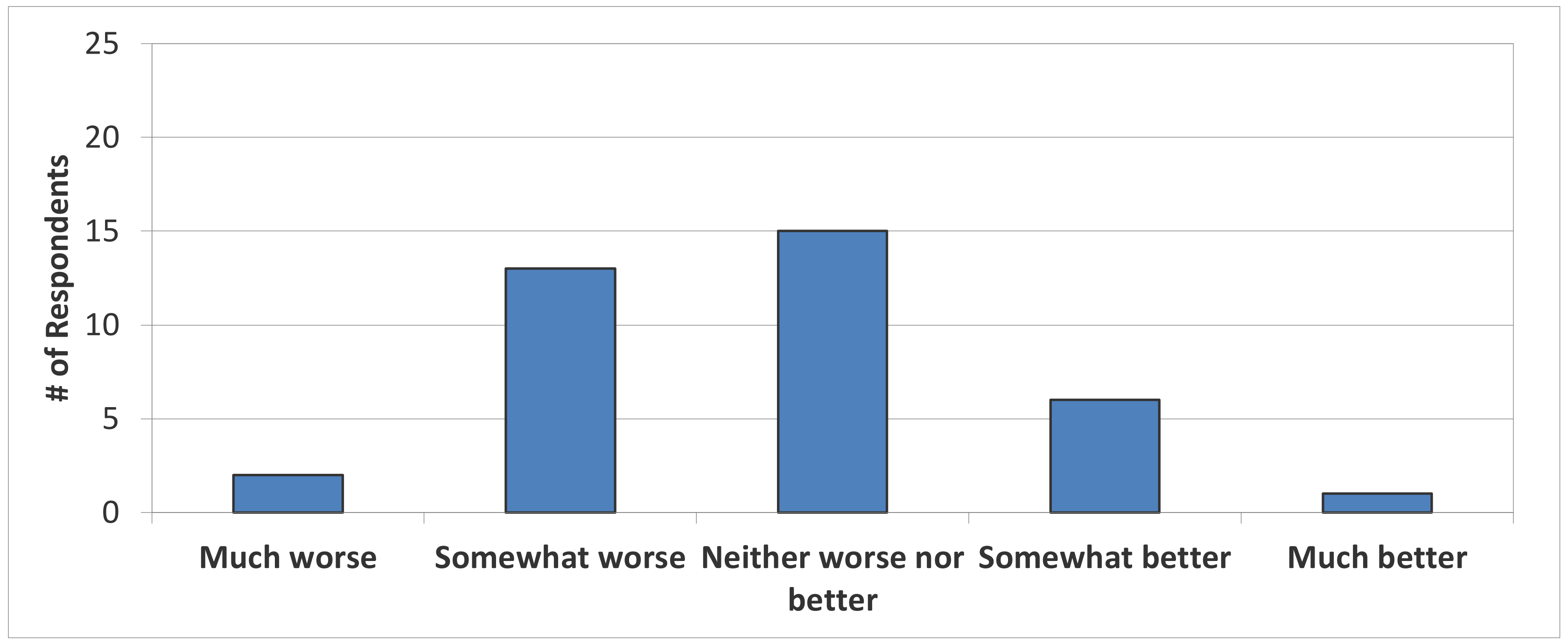
10. How would you describe the current quality of fishing on Forest Lake?

Answer Options	Very Poor	Poor	Fair	Good	Excellent	Response Count
	0	1	19	16	1	37
					answered question	**37**
					skipped question	**14**



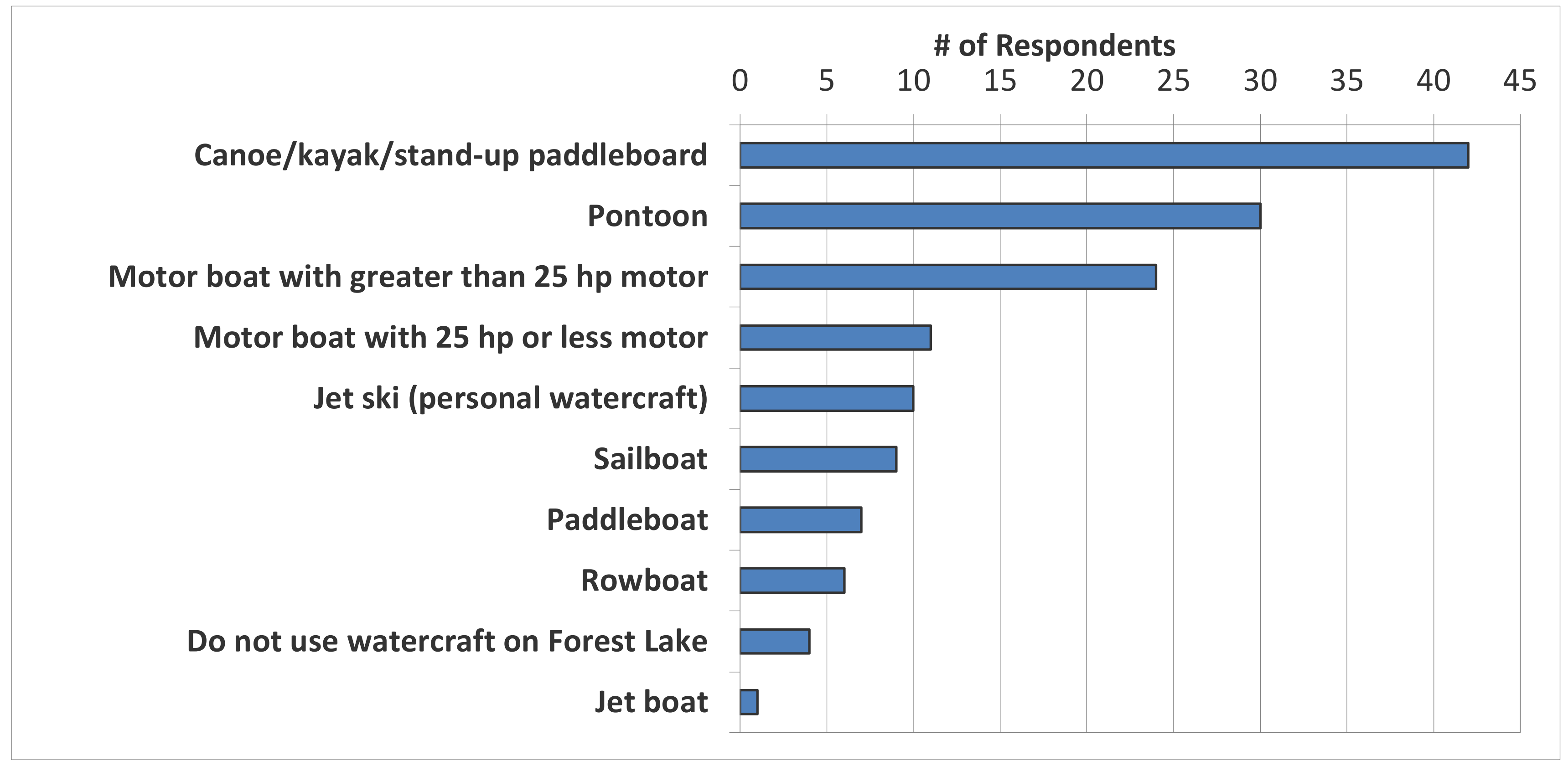
11. How has the quality of fishing changed on Forest Lake since you have started fishing the lake?

Answer Options	Much worse	Somewhat worse	Neither worse nor better	Somewhat better	Much better	Response Count
	2	13	15	6	1	37
					answered question	**37**
					skipped question	**14**



12. What types of watercraft do you currently use on Forest Lake?

Answer Options	Response Percent	Response Count
Canoe/kayak/stand-up paddleboard	82%	42
Pontoon	59%	30
Motor boat with greater than 25 hp motor	47%	24
Motor boat with 25 hp or less motor	22%	11
Jet ski (personal watercraft)	20%	10
Sailboat	18%	9
Paddleboat	14%	7
Rowboat	12%	6
Do not use watercraft on Forest Lake	8%	4
Jet boat	2%	1
	answered question	**37**
	skipped question	**14**



13. Do you use your watercraft on waters other than Forest Lake?

Answer Options	Response Percent	Response Count
Yes	31%	15
No	69%	34
	answered question	**49**
	skipped question	**2**

14. What is your typical cleaning routine after using your watercraft on waters other than Forest Lake?

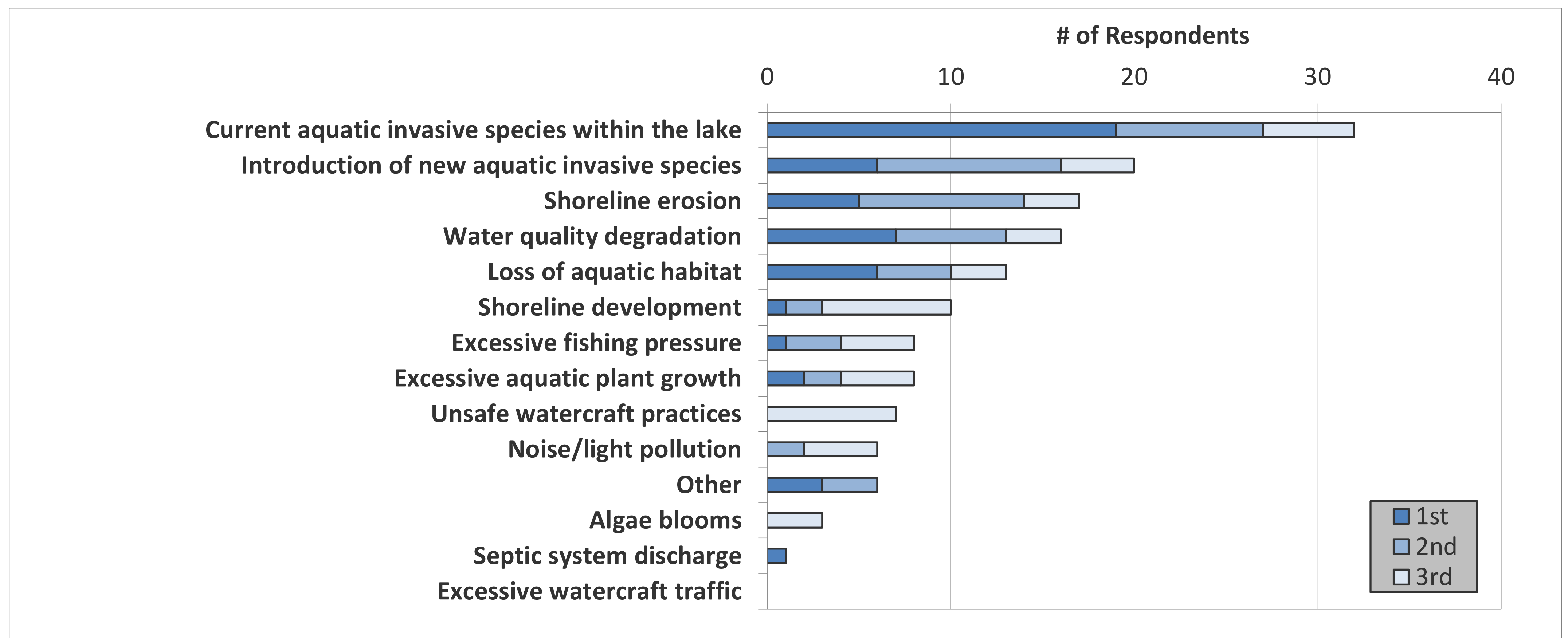
Answer Options	Response Percent	Response Count
Remove aquatic hitch-hikers (ex. - plant material, clams, mussels)	88%	14
Drain bilge	75%	12
Rinse boat	13%	2
Power wash boat	25%	4
Apply bleach	0%	0
Air dry boat for 5 or more days	50%	8
Do not clean boat	0%	0
Other	19%	3
	answered question	**16**
	skipped question	**35**

Number	"Other" responses
1	It is only our kayaks that we take to other lakes.
2	Have not actually used on other lake - yet
3	PUT ON BOAT LIFT

15. From the list below, please rank your top three concerns regarding Forest Lake, with 1 being your greatest concern.

Answer Options	1st	2nd	3rd	Response Count
Current aquatic invasive species within the lake	19	8	5	32
Introduction of new aquatic invasive species	6	10	4	20
Shoreline erosion	5	9	3	17
Water quality degradation	7	6	3	16
Loss of aquatic habitat	6	4	3	13
Shoreline development	1	2	7	10
Excessive fishing pressure	1	3	4	8
Excessive aquatic plant growth	2	2	4	8
Unsafe watercraft practices	0	0	7	7
Noise/light pollution	0	2	4	6
Other	3	3	0	6
Algae blooms	0	0	3	3
Septic system discharge	1	0	0	1
Excessive watercraft traffic	0	0	0	0
			answered question	51
			skipped question	0

Number	"Other" responses
1	More positive change over the years than negative!
2	swimmers itch
3	Lack of knowledge and/or inconsistent enforcement of deed restrictions.
4	Improper aerator use creating open water in front of others' shoreline, and/or causing unsafe conditions.
5	Maybe not loss of aquatic habitat - does lake have the right aquatic plants expanding?
6	Fish management - low walleye counts
	Water run off from developed properties with fertilized lawns and chemicals that come with run off. Also, disregard and/or lack of understanding of shoreline preservation ordinances.



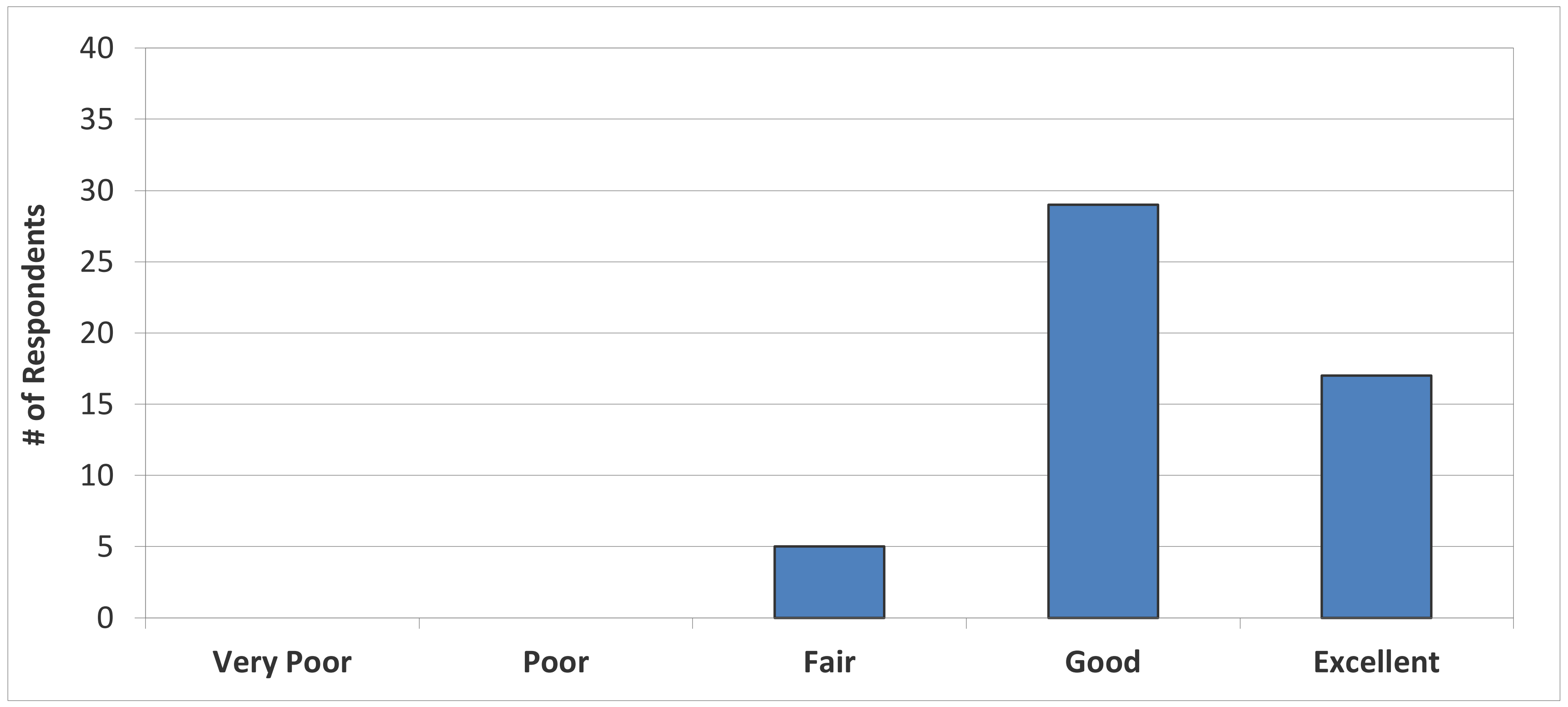
16. Which of the following would you say is the single most important aspect when considering water quality?

Answer Choices	Response Percent	Response Count
Water clarity (clearness of water)	55%	28
Water color	0%	0
Aquatic plant growth	20%	10
Algae blooms	4%	2
Smell/odors	0%	0
Water level	6%	3
Fish kills	2%	1
Other	14%	7
	answered question	**51**
	skipped question	**1**

Number	"Other" responses
1	Other
2	Phosphorus levels
3	Water composition
4	Toxins
5	Health of the fish and other aquatic species in the lake and animals dependant on the lake for subsistence.
6	I'm not sure.
7	AIS
8	Not educated enough to know - think water clarity is top; which helps aquatic plant growth

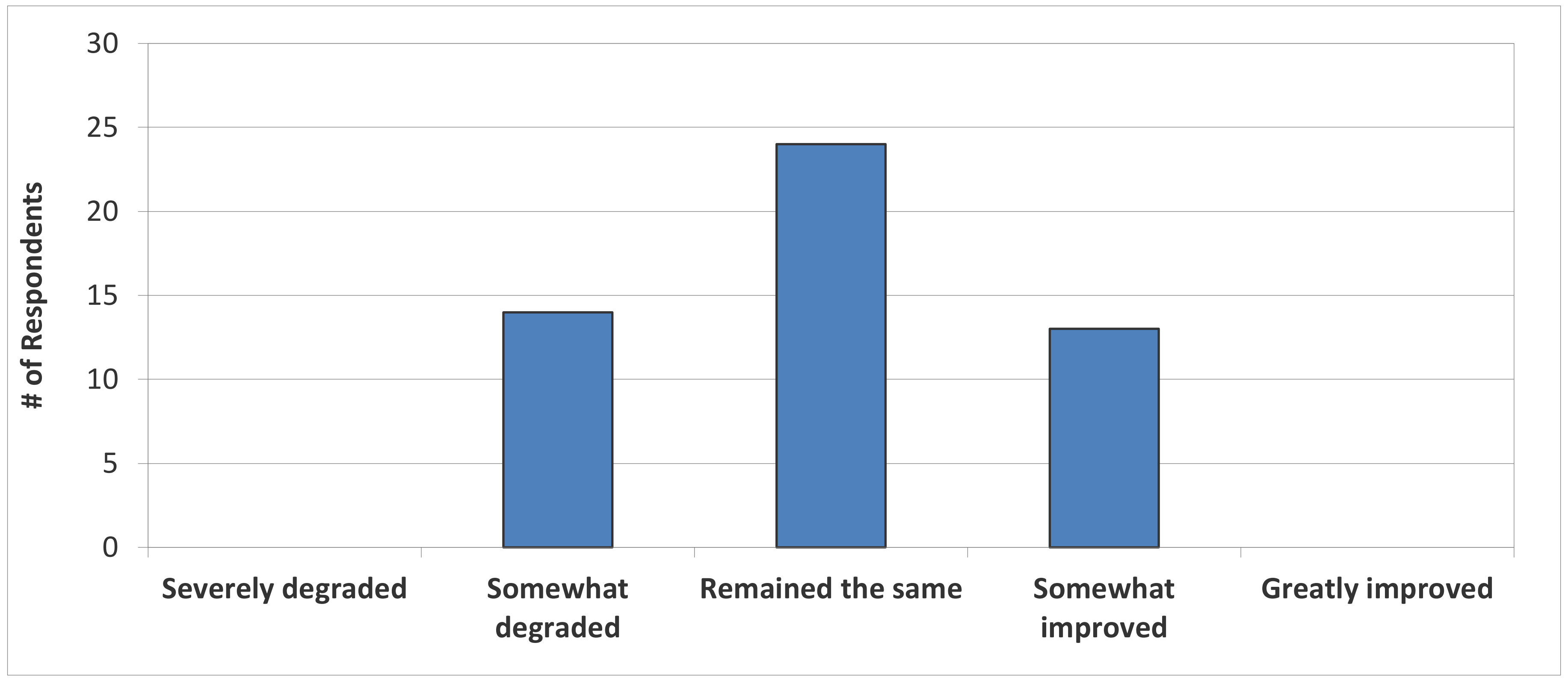
17. How would you describe the overall current water quality of Forest Lake?

Answer Options	Very Poor	Poor	Fair	Good	Excellent	Response Count
	0	0	5	29	17	51
					answered question	51
					skipped question	0

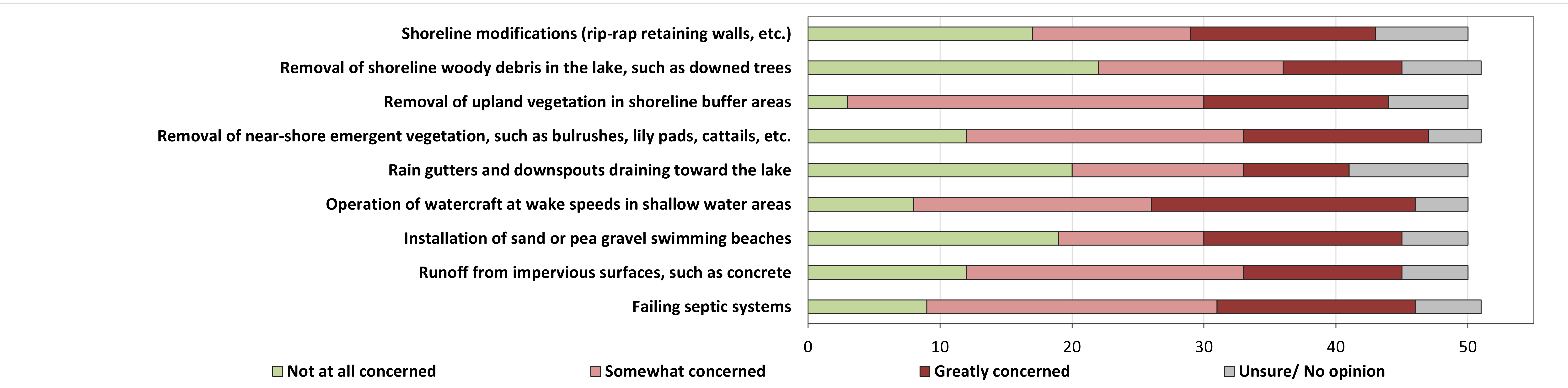


18. How do you think the overall water quality has changed in Forest Lake since you first visited the lake?

Answer Options	Severely degraded	Somewhat degraded	Remained the same	Somewhat improved	Greatly improved	Response Count
	0	14	24	13	0	51
					answered question	**51**
					skipped question	**0**



19. How concerned are you about the effect of the following practices upon the water quality in Forest Lake?					
Answer Options	**Not at all concerned**	**Somewhat concerned**	**Greatly concerned**	**Unsure/ No opinion**	**Response Count**
Failing septic systems	9	22	15	5	51
Runoff from impervious surfaces, such as concrete	12	21	12	5	50
Installation of sand or pea gravel swimming beaches	19	11	15	5	50
Operation of watercraft at wake speeds in shallow water areas	8	18	20	4	50
Rain gutters and downspouts draining toward the lake	20	13	8	9	50
Removal of near-shore emergent vegetation, such as bulrushes, lily pads, cattails, etc.	12	21	14	4	51
Removal of upland vegetation in shoreline buffer areas	3	27	14	6	50
Removal of shoreline woody debris in the lake, such as downed trees	22	14	9	6	51
Shoreline modifications (rip-rap retaining walls, etc.)	17	12	14	7	50
				answered question	**51**
				skipped question	**0**



20. Have you, anyone from your household, or a guest experienced swimmer’s itch as a result of participating in water activities in Forest Lake?

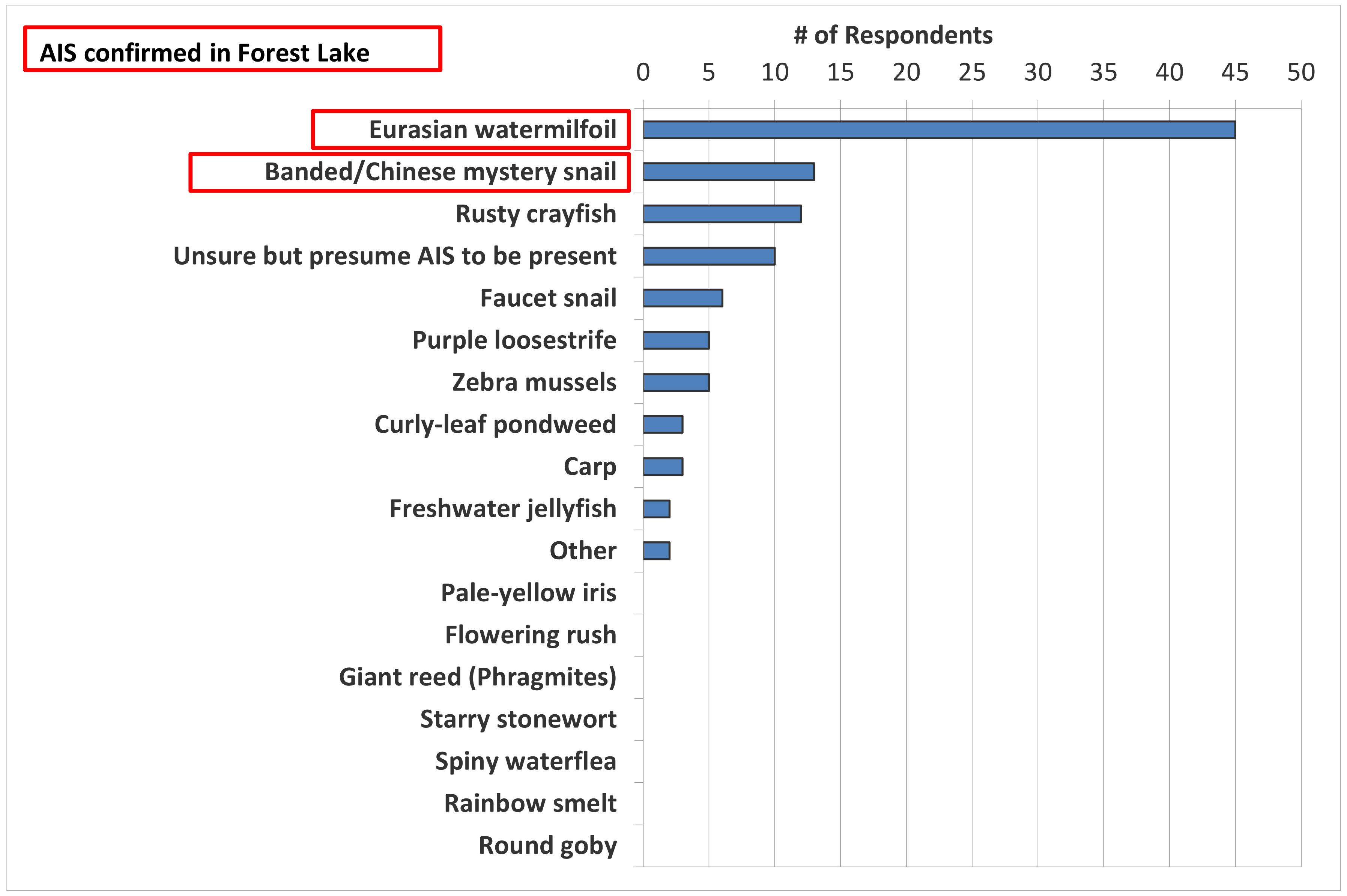
Answer Options	Responses
Yes	26
I think so but can't say for certain	2
No	23
Answered	**51**
Skipped	**0**

21. Have you ever heard of aquatic invasive species (AIS)?

Answer Options	Response Percent	Response Count
Yes	98%	50
No	2%	1
	answered question	**51**
	skipped question	**0**

22. Which aquatic invasive species do you believe are present in or immediately around Forest Lake?

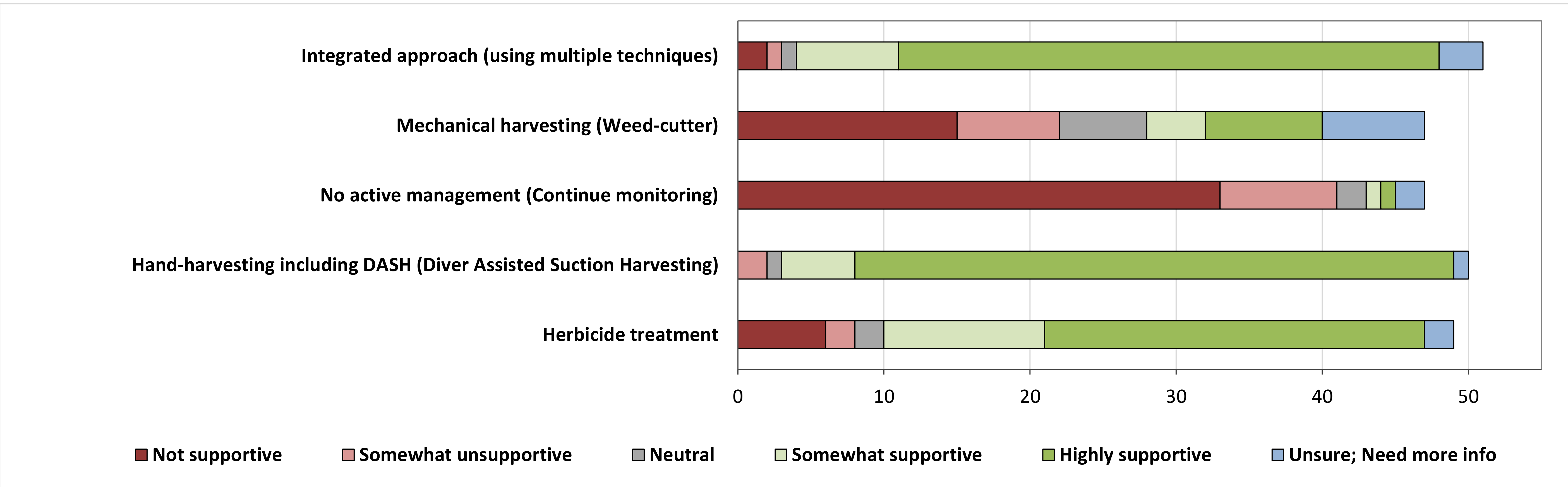
Answer Options	Response Percent	Response Count
Eurasian watermilfoil	90%	45
Banded/Chinese mystery snail	26%	13
Rusty crayfish	24%	12
Unsure but presume AIS to be present	20%	10
Faucet snail	12%	6
Purple loosestrife	10%	5
Zebra mussels	10%	5
Curly-leaf pondweed	6%	3
Carp	6%	3
Freshwater jellyfish	4%	2
Other	4%	2
Pale-yellow iris	0%	0
Flowering rush	0%	0
Giant reed (Phragmites)	0%	0
Starry stonewort	0%	0
Spiny waterflea	0%	0
Rainbow smelt	0%	0
Round goby	0%	0
	answered question	50
	skipped question	1



Number	"Other" responses
1	Snails but not sure of species
2	Snail, not sure which kind

23. The Forest Lake Association is currently assessing future techniques for continuing to manage the EWM population. What is your level of support for the future use of the following EWM management techniques in Forest Lake? Please select one response for each control technique.

Answer Options	Not supportive	Somewhat unsupportive	Neutral	Somewhat supportive	Highly supportive	Unsure; Need more info	Response Count
Herbicide treatment	6	2	2	11	26	2	49
Hand-harvesting including DASH (Diver Assisted Suction Harvesting)	0	2	1	5	41	1	50
No active management (Continue monitoring)	33	8	2	1	1	2	47
Mechanical harvesting (Weed-cutter)	15	7	6	4	8	7	47
Integrated approach (using multiple techniques)	2	1	1	7	37	3	51
						answered question	**51**
						skipped question	**0**



24. What concerns, if any, do you have for the future use of aquatic herbicides, DASH/hand harvesting, or Mechanical Harvesting to target EWM in Forest Lake?

Answer Options	Aquatic Herbicide	DASH/ Hand-Harvesting	Mechanical Harvesting	Response Count
Potential cost of treatment is too high.	7	17	6	21
Potential impacts to native aquatic plant species	29	1	8	34
Potential impacts to native (non-plant) species such as fish, insects, etc.	30	1	3	31
Potential impacts to human health	32	0	0	32
Future impacts are unknown	30	4	6	33
Ineffectiveness of herbicide strategy	24	6	4	29
Another reason				5
			answered question	**40**
			skipped question	**11**

Number	"Other" responses
1	The milfoil becomes resistant to the herbicides over time. Plus the plants die, releasing even more nutrients into the lake which will cause the lake to become more eutrophic over time
2	Release of viable sections of euasion milfoil when hand harvesting.
3	Don't think Mechanical harvesting is needed now or in the future when other methods are maintained.
4	Supportive of herbicide but do not know enough to say if highly supportive
5	Always a concern on all, but we have been blessed to have Tom Macek who I would trust to make decisions 100%

Forest Lake Association (FLA)

25. Before receiving this mailing, have you ever heard of the Forest Lake Association?

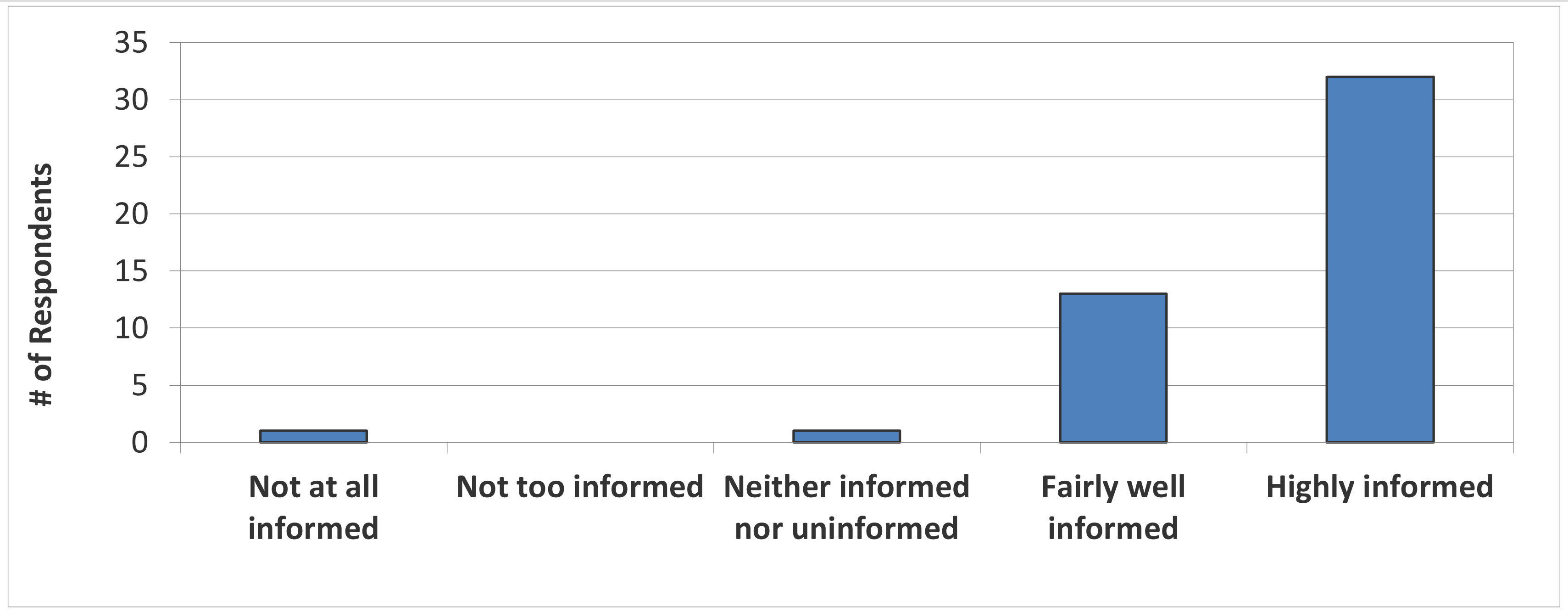
Answer Options	Response Percent	Response Count
Yes	100%	51
No	0%	0
	answered question	**51**
	skipped question	**0**

26. What is your membership status with the Forest Lake Association?

Answer Options	Response Percent	Response Count
Current member	92%	47
Former member	2%	1
Never been a member	6%	3
	answered question	**51**
	skipped question	**0**

27. How informed has the Forest Lake Association kept you regarding issues with the management of Forest Lake?

Answer Options	Not at all informed	Not too informed	Neither informed nor uninformed	Fairly well informed	Highly informed	Response Count
	1	0	1	13	32	47
					answered question	**47**
					skipped question	**4**



28. Before receiving this mailing, had you ever heard of the Forest Lake Preservation Foundation?

Answer Options	Response Percent	Response Count
Yes	96%	49
No	4%	2
answered question		**51**
skipped question		**1**

29. Have you ever donated to the Forest Lake Preservation Foundation?

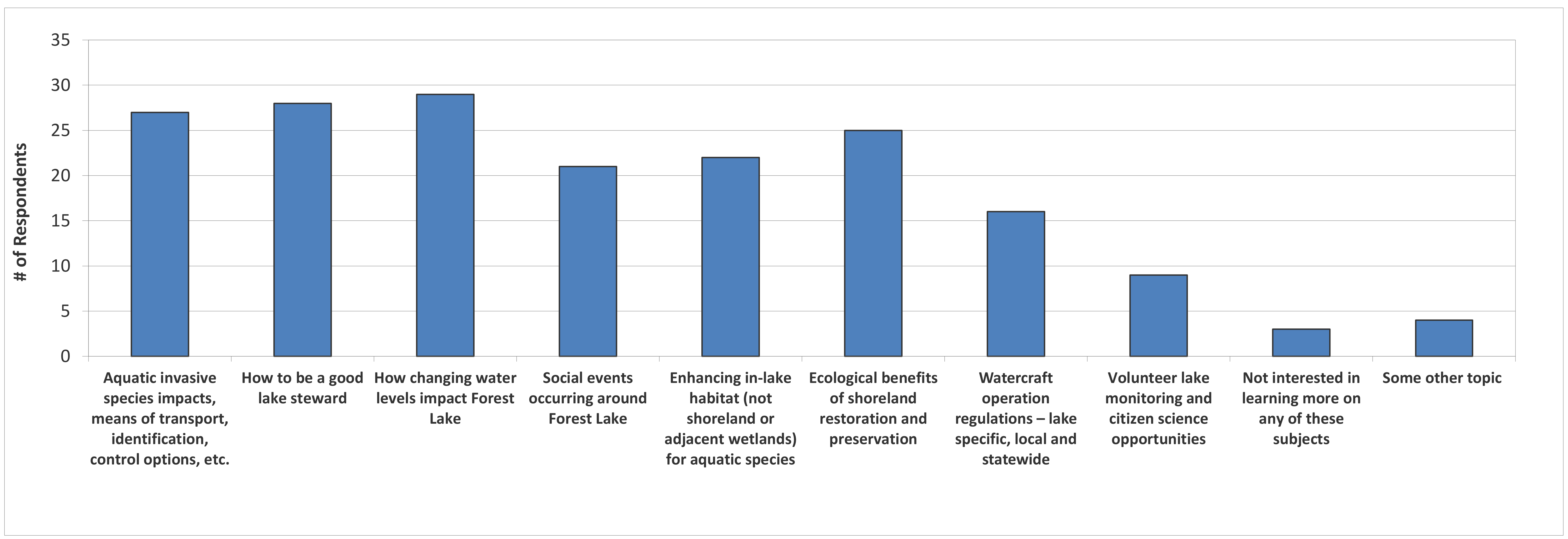
Answer Options	Response Percent	Response Count
Yes	88%	45
No	12%	6
	answered question	**51**
	skipped question	**1**

30. How informed has the Forest Lake Preservation Foundation kept you regarding issues with the management of Forest Lake? Please select one response.

Answer Options	Not at all informed	Not too informed	Neither informed nor uninformed	Fairly well informed	Highly informed	Total
	2	3	2	18	25	50
					answered question	**50**
					skipped question	**2**

31. Stakeholder education is an important component of every lake management planning effort. Which of these subjects would you like to learn more about?		
Answer Options	**Response Percent**	**Response Count**
Aquatic invasive species impacts, means of transport, identification, control options, etc.	57%	27
How to be a good lake steward	60%	28
How changing water levels impact Forest Lake	62%	29
Social events occurring around Forest Lake	45%	21
Enhancing in-lake habitat (not shoreland or adjacent wetlands) for aquatic species	47%	22
Ecological benefits of shoreland restoration and preservation	53%	25
Watercraft operation regulations – lake specific, local and statewide	34%	16
Volunteer lake monitoring and citizen science opportunities	19%	9
Not interested in learning more on any of these subjects	6%	3
Some other topic	9%	4
	answered question	**47**
	skipped question	**4**

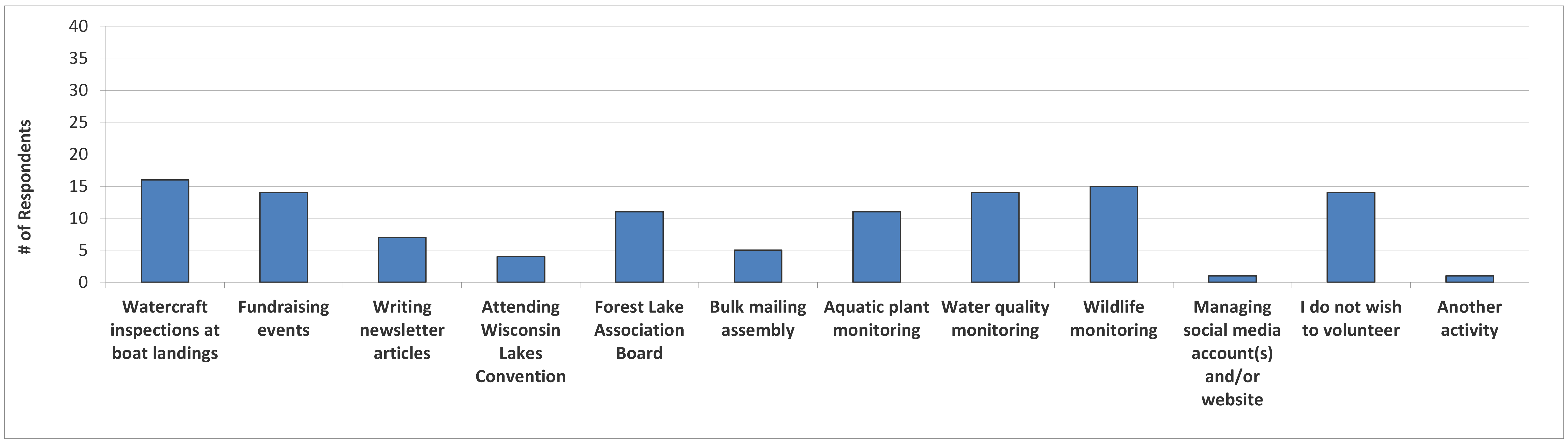
Number	"Some other topic" responses
1	I am very well versed in all these topics, but believe newer lake property owners would benefit from them.
2	Wake board boats banned
3	Mgmt and balance of fish species in lake
4	Would really like to have people understand better what they can do better on y their property to prevent the negative affects of water run off to the lake. And the importance of keeping the first 35 feet from waters edge as natural as possible as this is what Vilas County has said can be the most important thing to do to prevent unnatural chemicals getting into the lake.



32. The effective management of Forest Lake will require the cooperative efforts of numerous volunteers. Please select the activities you would be willing to participate in if the FLA requires additional assistance.

Answer Options	Response Percent	Response Count
Watercraft inspections at boat landings	33%	16
Fundraising events	29%	14
Writing newsletter articles	14%	7
Attending Wisconsin Lakes Convention	8%	4
Forest Lake Association Board	22%	11
Bulk mailing assembly	10%	5
Aquatic plant monitoring	22%	11
Water quality monitoring	29%	14
Wildlife monitoring	31%	15
Managing social media account(s) and/or website	2%	1
I do not wish to volunteer	29%	14
Another activity	2%	1
	answered question	**49**
	skipped question	**2**

Number	"Another activity" responses
1	We will be leaving Forest Lake within the next two years.



33. If the Association and Foundation were running out of funds (knowing that Hand harvesting and D.A.S.H. harvesting are much more costly than herbicides) would this make you more likely to support the Association and/or Foundation in order to minimize herbicide use?

Answer Options	Response Percent	Response Count
Yes	68%	34
No	32%	16
	answered question	**50**
	skipped question	**2**

34. Given the potential positive and negative consequences of doing so, which of the following reflects your beliefs about what should be done?

Answer Options	Response Percent	Response Count
Leave the boat landing the way it is	75%	38
Ask the Town to work with the DNR to improve access to the boat landing	25%	13
	answered question	**51**
	skipped question	**1**

35. Please feel free to provide written comments concerning Forest Lake, its current and/or historic condition and its management.

Answer Options	Response Count
	17
answered question	**17**
skipped question	**35**

Number	Response Text
1	We have noticed many more fish this year and wondered if it was because of the boat launch access. We vote to leave the launch as is. We will be happy to donate to the foundation but we do not want our donation used towards herbicides so we would like an option for earmarking funds towards the non herbicide methods only. One important factor to consider is land on the other side of forest lake road. The woods on the other side help to greatly protect the lake. Any development on that side can also lead to run off into the lake as well and so keeping that part of the ecosystem as undeveloped as possible is also really important. Thanks to everyone working on these issues!
2	When its 6:30 am at the boat landing and there are 4-6 trucks parked with empty trailers, the "clean boats" program's a bit of a well intended sham. How about intentionally connecting with the 20-40 year olds who love being here and will probably inherit their family property? Their prospectives on the future of Forest Lake might be very interesting. And an occasional letter from the heart like Bruce and Janine's daughter wrote would be lovely.
3	Would like board to investigate complaints/ comments that come to them rather than making decisions when they may not have all of the information
4	Some people are doing a wonderful job contributing to lake management, but we need more people to contribute. I think that the Lake Preservation Board should be doing more than asking for big contributions. Fund raising efforts are pretty much nonexistent other than just asking for donations. There are so many fun ways to raise money that can be inclusive of all people. We need to create more of a Forest Lake community that is not based on the size of your wallet. Involvement in the Lake Preservation Board should not be a bought seat, but based on other skills to make a continued, successful board.
5	For the question about running out of money, the association should be using the methods that deliver the most cost effective results. Safety of herbicide seems like a preliminary question. If it is believed safe, then its use should be determined based on its cost and effectiveness.
6	Having almost 70 years experience visiting and owning property on Forest Lake coupled with my grand fathers experience before me, most changes in the lake are cyclical, i.e., high water /low water; good fishing/not so good fishing; periods of drought/rainy periods etc. and I would urge caution in over reacting to these "natural" cycles. Invasive species are different and should be addressed with chemical use only as a last resort. Prevention of the introduction of additional invasive species should be paramount and I urge the Board to work with the foundation to find a way to fund the permanent "dawn to dusk" monitoring of the boat landing. Overall, despite significant development, including larger boats with larger motors and annoying personal watercraft, the lake has, overall, survived quite nicely. I attribute that to the care and attention paid to the lake by the past residents. With the lake undergoing a significant change in ownership, I strongly encourage the Board to do what it can to instill in this "new generation" of lake property owners, who may not have a historical connection to the lake, that same sense of responsibility to care for Forest Lake for future generations.
7	We greatly appreciate the stewardship of Forest Lake by the Association and Foundation. We do find the Association board to be visibly, publicly inconsistent in the way it enforces CC&Rs. causing a "two-tier" sense of favoritism. Example: "Not more than one RV, camper etc. per property. Except for Painless's friends at the 4th of July? Not that I want to deny him that...but perhaps you could be explicit about exceptions? Would be great to address that kind of thing more equitably.
8	Wake boat operators are severely eroding the shoreline, ignoring no-wake and distance regulations. Would also like to see a total ban of old (pre-1990?) 2 stroke outboards to reduce the carbon/oil deposits from the exhaust.
9	Appreciate all that is being done to preserve/improve Forest Lake!
10	Impressive, active management!
11	The boats are too big for this lake
12	I appreciate all who have served in the past as well as those serving now for their stewardship of the lake.
13	Thank you to all who care so deeply about this resource!!!

14	Wish all homeowners and visitors were more aware and followed Courtesy Code rules/suggestions.
15	I love Forest Lake and really appreciate my neighbor, Silas Beattie who helps me continue to live here!
16	Thank you to all the volunteers who make this great lake the best there is with a bright future for generations to come
17	First of all, certain people, such as Bill Heinig and Tom Macek have been class acts for our lake… Bill from years ago and Tom as water quality issues has escalated over the past 25 years. They and a handful of others have always tried to do the right thing on addressing issues and considering other people's perspectives. On the more concerning side, two things. First, over the past ten to fifteen years, it seems like people are quicker to pass judgement on "things/issues" before the facts…along with other perspectives… are first gathered. I am sure this has escalated as well because of the classic pitfalls of social media. Either way, it's just been disheartening. Second, the escalated new and renovation building development starting in the late 80's and early 90's has not put enough emphasis on the negative impact of the additional water run-off into the lake that this varied development activity has created. Now thirty to forty years later, we have seen the negative impact of this. Some have paid more attention to it and others not so much, but in general, the lack of attention has had a negative impact And now as property ownership is changing more rapidly as generations pass, newcomers… in general… within existing ownership and new ownership do not appear as sensitive to stewardship of the land and lake for those here today and future generations as those that came before them. There seems to be more of a "monkey see monkey do" mentality , where if "one can do it so can I". I hate to do this, but, for example, the three properties in the bay by the landing that have changed hands in the past few years that slowly have trees "disappearing"from their frontage, and/or more rip rap put along the shoreline with no additional vegetation added to slow the run-off down. Or the new home rebuilt so close to the lake on the southwest side that has the long asphalt driveway that accelerates run -off to the lake with no additional vegetation planted in the 35 foot zone next to the lake. Wish more people would have done what the Bablitch family did when they renovated their classic white place on the north side of the lake years ago. They replanted their entire frontage with more vegetation! Now that's a true example of lake and land stewardship! I hesitated to call these situations out, but finally decided to speak my piece as keeping quiet and closing an eye to it has not been effective. I am not saying these are bad people, but they… as well as I…need to take more of an approach of asking themselves/ourselves the following . First, am I educated enough on how what I am doing to my property is impacting the land and water of the lake, and then reacting appropriately on what they /themselves/ myself can do better…. No matter what others have done or are doing. Enough said. And again, thanks to so many that are always trying to do the right thing and putting the time and effort forth to do so.

C

APPENDIX C

Point-Intercept Survey – Aquatic Plant Littoral Frequency Matrix

Forest Lake Point-Intercept
Survey Matrix

Scientific Name	Common Name	LFOO (%)				
		2005	2013	2016	2019	2022
Potamogeton robbinsii	Fern-leaf pondweed	15.0	23.6	31.2	30.5	37.9
Elodea canadensis & E. Nuttallii	Common & Slender waterweed	15.7	26.8	15.8	18.6	20.3
Elodea canadensis	Common waterweed	15.4	26.8	15.8	17.2	20.3
Najas flexilis	Slender naiad	16.9	9.5	10.9	9.3	9.0
Potamogeton pusillus & P. berchtoldii	Small & Slender pondweed	8.8	15.3	5.5	4.0	6.2
Chara spp.	Muskgrasses	1.9	1.5	12.2	5.1	11.9
Potamogeton pusillus	Small pondweed	8.8	15.3	5.5	4.0	3.7
Myriophyllum sibiricum	Northern watermilfoil	0.5	3.8	2.3	5.1	9.0
Potamogeton gramineus	Variable-leaf pondweed	1.9	2.3	2.9	4.2	7.9
Potamogeton zosteriformis	Flat-stem pondweed	1.0	9.0	7.5	4.5	1.7
Eleocharis acicularis	Needle spikerush	1.0	4.0	6.0	3.7	2.3
Heteranthera dubia	Water stargrass	0.0	1.0	2.6	7.3	2.3
Bidens beckii	Water marigold	2.6	3.3	3.6	3.4	0.8
Potamogeton berchtoldii	Slender pondweed	0.0	0.0	5.5	4.0	2.8
Vallisneria americana	Wild celery	0.0	0.3	1.3	2.8	3.4
Potamogeton amplifolius	Large-leaf pondweed	0.2	0.5	2.1	2.5	1.1
Nymphaea odorata	White water lily	0.7	0.5	0.8	0.8	2.0
Potamogeton obtusifolius	Blunt-leaved pondweed	0.0	0.0	0.0	2.5	2.0
Potamogeton illinoensis	Illinois pondweed	0.0	1.3	0.0	1.4	1.7
Najas guadalupensis	Southern naiad	0.0	0.0	0.0	0.0	2.8
Potamogeton praelongus	White-stem pondweed	0.0	0.0	0.8	0.3	2.0
Myriophyllum spicatum	Eurasian watermilfoil	0.0	0.5	0.0	3.4	0.6
Elodea nuttallii	Slender waterweed	1.7	0.0	0.0	2.0	0.3
Potamogeton spirillus	Spiral-fruited pondweed	0.2	0.8	0.5	1.1	0.3
Potamogeton foliosus	Leafy pondweed	0.0	0.0	0.0	0.0	1.4
Nitella spp.	Stoneworts	0.0	0.3	0.0	1.4	0.6
Isoetes spp.	Quillwort spp.	0.2	0.8	0.0	0.0	0.8
Utricularia vulgaris	Common bladderwort	0.0	0.0	0.0	0.3	0.8
Schoenoplectus tabernaemontani	Softstem bulrush	0.0	0.5	1.3	0.0	0.0
Potamogeton praelongus X P. richardsonii	White-stem X clasping-leaf pondweed	0.0	0.0	1.3	0.0	0.0
Potamogeton richardsonii X P. gramineus	Clasping-leaf X variable-leaf pondweed	1.2	0.0	0.0	0.0	0.0
Sparganium angustifolium	Narrow-leaf bur-reed	0.2	0.0	0.3	0.6	0.0
Potamogeton alpinus	Alpine pondweed	0.0	1.0	0.0	0.0	0.0
Eleocharis palustris	Creeping spikerush	0.0	0.5	0.5	0.0	0.0
Fissidens spp. & Fontinalis spp.	Aquatic Moss	0.0	0.0	0.3	0.0	0.3
Brasenia schreberi	Watershield	0.0	0.0	0.3	0.0	0.3
Potamogeton richardsonii	Clasping-leaf pondweed	0.0	0.0	0.0	0.6	0.0
Persicaria amphibia	Water smartweed	0.0	0.0	0.5	0.0	0.0
Nuphar variegata	Spatterdock	0.0	0.0	0.0	0.0	0.3
Typha spp.	Cattail spp.	0.0	0.3	0.0	0.0	0.0
Sparganium androcladum	Shining bur-reed	0.0	0.0	0.3	0.0	0.0
Sparganium americanum	American bur-reed	0.0	0.0	0.3	0.0	0.0
Sagittaria sp. (rosette)	Arrowhead sp. (rosette)	0.0	0.0	0.3	0.0	0.0
Sagittaria rigida	Stiff arrowhead	0.0	0.3	0.0	0.0	0.0
Sagittaria graminea	Grass-leaved arrowhead	0.0	0.3	0.0	0.0	0.0
Potamogeton natans	Floating-leaf pondweed	0.0	0.0	0.0	0.3	0.0
Potamogeton amplifolius X P. illinoensis	Large-leaf X illinois pondweed	0.0	0.3	0.0	0.0	0.0
Dulichium arundinaceum	Three-way sedge	0.0	0.0	0.3	0.0	0.0

D

APPENDIX D

Strategic Analysis of Aquatic Plant Management in Wisconsin (June 2019). Extracted Supplemental Chapters:

- 3.3 Herbicide Treatment
- 3.4 Physical Removal
- 3.5 Biological Control

In 2016-2019, the WDNR conducted a Strategy Analysis of Aquatic Plant Management in Wisconsin, which will serve as a reference document to mold future 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.

Carfentrazone-ethyl

Registration and Formulations

Carfentrazone-ethyl is a contact herbicide that was registered with the EPA in 1998. The active ingredient is ethyl 2-chloro-3-[2 -chloro-4-fluoro-5-[4 -(difluoromethyl)-4,5-diydro-3-methyl-5-oxo-1H-1,2,4-trizol-1-yl)phenyl]propanoate. A liquid formulation of carfentrazone-ethyl is commercially sold for aquatic use.

Mode of Action and Degradation

Carfentrazone-ethyl controls plants through the process of membrane disruption which is initiated by the inhibition of the enzyme protoporphyrinogen oxidase, which interferes with the chlorophyll biosynthetic pathway. The herbicide is absorbed through the foliage of plants, with injury symptoms viable within a few hours after application, and necrosis and death observed in subsequent weeks.

Carfentrazone-ethyl breaks down rapidly in the environment, while its degradates are persistent in aquatic and terrestrial environments. The herbicide primarily degrades via chemical hydrolysis to carfentrazone-chloropropionic acid, which is then further degraded to carfentrazone -cinnamic, -propionic, -benzoic and 3-(hydroxymethyl)-carfentrazone-benzoic acids. Studies have shown that degradation of carfentrazone-ethyl applied to water (pH = 7-9) has a half-life range of 3.4-131 hours, with longer half-lives (>830 hours) documented in waters with lower pH (pH = 5). Extremes in environmental conditions such as temperature and pH may affect the activity of the herbicide, with herbicide symptoms being accelerated under warm conditions.

While low levels of chemical residue may occur in surface and groundwater, risk concerns to non-target organisms are not expected. If applied into water, carfentrazone-ethyl is expected to adsorb to suspended solids and sediment.

Toxicology

There is no restriction on the use of treated water for recreation (e.g., fishing and swimming). Carfentrazone-ethyl should not be applied directly to water within ¼ mile of an active potable water intake. If applied around or within potable water intakes, intakes must be turned off prior to application and remain turned off for a minimum of 24 hours following application; the intake may be turned on prior to 24 hours only if the carfentrazone-ethyl and major degradate level is determined by laboratory analysis to be below 200 ppb. Do not use water treated with carfentrazone-ethyl for irrigation in commercial nurseries or greenhouses. In scenarios where the herbicide is applied to 20% or more of the surface area, treated water should not be used for irrigation of crops until 14 days after treatment, or until the carfentrazone-ethyl and major degradate level is determined by analysis to be below 5 ppb.

In scenarios where the herbicide is applied as a spot treatment to less than 20% of the waterbody surface area, treated water may be used for irrigation by commercial turf farms and on residential turf and ornamentals without restriction. If more than 20% of the waterbody surface area is treated, water should not be used for irrigation of turf or ornamentals until 14 days after treatment, or until the carfentrazone-ethyl and major degradate level is determined by analysis to be below 5 ppb.

Carfentrazone-ethyl is listed as very toxic to certain species of algae and listed as moderately toxic to fish and aquatic animals. Treatment of dense plants beds may result in dissolved oxygen declines from plant decomposition which may lead to fish suffocation or death. To minimize impacts, applications of this herbicide should treat up to a maximum of half of the waterbody at a time and wait a minimum of 14 days before retreatment or treatment of the remaining half of the waterbody. Carfentrazone-ethyl is considered to be practically non-toxic to birds on an acute and sub-acute basis.

Carfentrazone-ethyl is harmful if swallowed and can be absorbed through the skin or inhaled. Those who mix or apply the herbicide need to protect their skin and eyes from contact with the herbicide to minimize irritation and avoid breathing the spray mist. Carfentrazone-ethyl is not carcinogenic, neurotoxic, or mutagenic and is not a developmental or reproductive toxicant.

Species Susceptibility

Carfentrazone-ethyl is used for the control of floating and emergent aquatic plants such as duckweeds (*Lemna* spp.), watermeals (*Wolffia* spp.), water lettuce (*Pistia stratiotes*), water hyacinth (*Eichhornia crassipes*), and salvinia (*Salvinia* spp.). Carfentrazone-ethyl can also be used to control submersed plants such as Eurasian watermilfoil (*Myriophyllum spicatum*).

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.

<u>Species Susceptibility</u>

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.

<u>Species Susceptibility</u>

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. Manual and Mechanical Cutting

<u>Manual and Mechanical Cutting</u>

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). A synthesis of numerous historical mechanical harvesting studies is compiled by Breck et al. 1979.

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).

<u>Ecological Impacts of Manual and Mechanical Cutting</u>

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 all 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).

S.3.4.2. 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 Hand-Pulling and DASH

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. Because DASH is a relatively new management approach, less information is available about potential impacts than for some more established techniques like large-scale mechanical harvesting.

S.3.4.3. Benthic Barriers

Benthic barriers can be used to kill existing plants or prevent their growth from the outset. They are sometimes referred to as benthic mats, or screens, and involve placing some sort of covering over a plant bed, which provides a physical obstruction to plant growth and reduces light availability. They may be best used for dense, confined infestations or along shore or for providing boat lanes (Engel 1983; Payne et al. 1993; Bailey and Calhoun 2008). Reductions in abundance of live aquatic plants beneath the barrier may be seen within weeks (Payne et al. 1993; Carter et al. 1994). The target plant species, light availability, and sediment accumulation have been shown to influence the efficacy of benthic barriers for aquatic plant control. Effects on the target plants may be more rapid in finer sediments because anoxic conditions are reached more quickly due to higher sediment organic content and oxidization by bacteria (Carter et al. 1994). Benthic barriers may be more expensive but less time intensive than some of the physical removal approaches described above (Carter et al. 1994; Bailey and Calhoun 2008). Engel (1983) suggests that benthic barriers may be useful in situations where plants are growing too deep for other physical removal approaches or effective herbicide application. They may also improve plant control when used in combination with herbicide treatments to hold most of the herbicide to a given treatment area (Helsel et al. 1996).

There is some necessary upkeep associated with the use of benthic barriers. Some barriers can be difficult to re-use because of algae and plants that can grow on top of the barrier. Periodically removing sediment that accumulates on the barrier can help offset this (Engel 1983; Carter et al. 1994; Laitala et al. 2012). Some materials are made to be removed after the growing season, which may make cleaning and re-use easier (Engel 1983). Additionally, gases often accumulate beneath benthic barriers as a result of plant decay, which can cause them to rise off the bottom of the waterbody, requiring further maintenance (Engel 1983; Ussery et al. 1997; Bailey and Calhoun 2008). Eurasian watermilfoil (*Myriophyllum spicatum*) and other plant species have been shown to recolonize the managed area quickly following barrier removal (Eichler et al. 1995; Boylen et al. 1996), so this approach may require hand-pulling or other integrated approaches once the barrier is removed (Carter et al. 1994; Eichler et al. 1995; Bailey and Calhoun 2008). Some studies have observed low abundance of plants maintained for 1-2 months after barriers were removed (Engel 1983). Others found that combining 2,4-D treatments with benthic barriers could reduce Eurasian watermilfoil to a degree that helped native plants recolonize the target site (Helsel et al. 1996).

The material used to create benthic barriers can vary and include biodegradable jute matting, fiberglass screens, and woven polypropylene fibers (Mayer 1978; Perkins et al. 1980; Lewis et al. 1983; Hoffman et al. 2013). Some plants such as Eurasian watermilfoil and common waterweed (Elodea canadensis; Eichler et al. 1995) are able to growth through the mesh in woven barriers but this material can be effective in reducing growth on certain target plant species (Payne et al. 1993; Caffrey et al. 2010; Hoffman et al. 2013). Hofstra and Clayton (2012) suggested that less dense materials barriers may provide selective control of some species while allowing more tolerant species, such as some charophytes (*Chara* spp. and *Nitella* spp.), to grow through. More dense materials may prevent growth of a wider range of aquatic plants (Hofstra and Clayton 2012). Most materials must be well anchored to the bottom of the waterbody, which can be accomplished early in the growing season or by placing the barriers on ice before thawing of the waterbody (Engel 1983). Gas accumulation can occur in using both fibrous mesh and screen-type barriers (Engel 1983).

Eurasian watermilfoil and common waterweed have been found to be somewhat resistant to control by benthic barriers (Perkins et al. 1980; Engel 1983) while affected species include hydrilla (*Hydrilla verticillata*), curly-leaf pondweed (*Potamogeton crispus*), and coontails (*Ceratophyllum* spp.; Engel 1983; Payne et al. 1993; Carter et al. 1994). One study found that an 8-week barrier placement removed Eurasian watermilfoil while allowing native plant regrowth after the barrier was retrieved; while shorter durations were less effective in reducing Eurasian watermilfoil abundance and longer durations negatively impacted native plant regrowth (Laitala et al. 2012).

<u>Ecological Impacts of Benthic Barriers</u>

Macroinvertebrates will be negatively affected by benthic barriers while they are in place (Engel 1983) but have been shown to rebound to pre-management conditions shortly after removal of the barrier (Payne et al. 1993; Ussery et al. 1997). Benthic barriers may also affect spawning of some warm water fish species through direct disruption of spawning habitat (NYSFOLA 2009). Additionally, increased ammonium and decreased dissolved oxygen contents are often observed beneath benthic barriers (Carter et al. 1994; Ussery et al. 1997). These water chemistry considerations may partially explain decreases in macroinvertebrate populations (Engel 1983; Payne et al. 1993) and ammonium content is likely to increase with sediment organic content (Eakin 1992). Toxic methane gas has also been found to accumulate beneath benthic barriers (Gunnison and Barko 1992).

There may be some positive ecological aspects of benthic barriers. Barriers may reduce turbidity and nutrient release from sediments (Engel 1983). They may also provide channels that improve ease of fish foraging when other aquatic plant cover is present near the managed area. Fish may feed on the benthic organisms colonizing any sediment accumulating on top of the barrier (Payne et al. 1993). Payne et al. (1993) also suggest that, despite negative impacts in the managed area, the overall impact of benthic barriers is negligible since they typically are only utilized in small areas of the littoral zone. However, further research is needed on the effects of benthic barriers on fish and wildlife populations and their ability to rebound following barrier removal (Eichler et al. 1995).

S.3.4.4. Dredging

Dredging is a method that involves the removal of top layers of sediment and associated rooted plants, sediment-dwelling organisms, and sediment-bound nutrients. This approach is "non-selective" (USACE 2012), meaning that it offers limited control over what material is removed. In addition to being employed as an APM technique, dredging is often used to manage water flow, provide navigation channels, and reduce the chance of flooding (USACE 2012). Due to the expense of this method, APM via dredging is often an auxiliary effect of dredging performed for other purposes (Gettys et al. 2014). However, reduced sediment nutrient load and decreased light penetration due to greater depth post-dredging may result in multi-season reductions in plant biomass and density (Gettys et al. 2014).

Several studies discuss the utility of dredging for APM. Dredging may be effective in controlling species that propagate by rhizomes, by removing the rhizomes from the sediment before they have a chance to grow (Dall Armellina et al. 1996b). Additionally, invasive phragmites has been controlled in areas where dredging increases water depth to ≥ 5-6 feet; though movement of the equipment used in dredging activities has been implicated in expanding the range of invasive phragmites (Gettys et al. 2014). In streams, dredging resulted in a significant reduction in plant biomass (≥ 90%). However, recovery of plant populations reflected the timing of management actions relative to flowering: removal prior to flowering allowed for plant population recovery within the same growing season, while removal after flowering meant populations did not rebound until the next spring (Kaenel and Uehlinger 1999). Sediment testing for chemical residue levels high enough to be considered hazardous waste (from historically used sodium arsenite, copper, chromium, and other inorganic compounds) should be conducted before dredging, to avoid stirring of toxic material into the water column. The department routinely requires sediment analysis before dredging begins and destination approval of spoils to prevent impacts from sediment leachate outside of the disposal area. Planning and testing can be an extensive component to a dredging project.

<u>Ecological effects of Dredging</u>

Repeated dredging may result in plant communities consisting of populations of fast-growing species that are capable of rebounding quickly (Sand-Jensen et al. 2000). In experimental studies, faster growing invasive plant species with a higher tolerance for disturbance were able to better recover from simulated dredging than slower growing native plant species, suggesting that post-dredging plant communities may be comprised of undesirable invasives (Stiers et al. 2011).

Macroinvertebrate biomass has been shown to decrease up to 65% following dredging, particularly among species which use plants as habitat. Species that live deeper in sediments, or those that are highly mobile, were less affected. As macroinvertebrates are valuable components of aquatic ecosystems, it is recommended that plant removal activities consider impacts on macroinvertebrates (Kaenel and Uehlinger 1999). Dredging can also result in declines to native mussel populations (Aldridge 2000).

Impacts to fish and water quality parameters have also been observed. Dredging to remove aquatic plants significantly increased both dissolved oxygen levels and the number of fish species found inhabiting farm ponds (Mitsuo et al. 2014). This increase in fish abundance may have been due to extremely high pre-dredging density of aquatic plants, which can negatively influence fish foraging success. In another study, aquatic plant removal decreased the amplitude of daily oxygen fluctuations in streams. However, post-dredging changes in metabolism were short-lived, suggesting that algae may have taken over primary productivity (Kaenel et al. 2000). Finally, several studies have also documented or suggested a reduction in sediment phosphorous levels after dredging, which may in turn reduce nutrient availability for aquatic plant growth (Van der Does et al. 1992; Kleeberg and Kohl 1999; Meijer et al. 1999; Søndergaard et al. 2001; Zuccarini et al. 2011). However, consideration must be given to factors affecting whether goals are obtainable via dredging (e.g., internal or external phosphorus inputs, water retention time, sediment characteristics, etc.).

S.3.4.5. Drawdown

Water-level drawdown is another approach for aquatic plant control as well as aquatic plant restoration. Exposure of aquatic plant vegetation, seeds, and other reproductive structures may reduce plant abundance by freezing, drying, or consolidation of sediments. This management technique is not effective for control of all aquatic plant species. Due to potential ecological impacts, it is necessary to consider other factors such as: waterfowl habitat, fisheries enhancement, release of nutrients and solids downstream, and refill and sediment consolidation potential. Often drawdowns for aquatic plant control and/or restoration can be coordinated to time with dam repair or repair of shoreline structures. A review by Cooke (1980), suggests drawdown can provide at least short-term aquatic plant control (1-2 years) when the target species is vulnerable to drawdown and where sediment can be dewatered under rigorous heat or cold for 1-2 months. Costs can be relatively low when a structure for manipulating water level is in place (otherwise high capacity pumps must be used). Conversely, costs can be high to reimburse an owner for lost power generation if the water control structure produces hydro-electric power. The aesthetic and recreational value of a waterbody may be reduced during a drawdown, as large areas of sediment are exposed prior to revegetation. Bathymetry is also important to consider, as small decreases in water level may lead to drop-offs if a basin does not have a gradual slope (Cooke 1980). The downcutting of the stream to form a new channel can also release high amounts of solids and organic matter that can impair water quality downstream. For example, in July 2005, the Waupaca Millpond, Waupaca Co. had to conduct an emergency drawdown that resulted in the river downcutting a new channel. High suspended solid concentrations and BOD resulted in decreased water clarity, sedimentation and depressed dissolved oxygen levels. A similar case occurred in 2015 with the Amherst Mill Pond, Portage Co. during a drawdown at a rate of six inches per day (Scott Provost [WDNR], *personal communication*).

Because extreme heat or cold provide optimal conditions for aquatic plant control, drawdowns are typically conducted in the summer or winter. Because of Wisconsin's cold winters, winter drawdown is likely to have several advantages when used for aquatic plant management, including avoiding many conflicts with recreational use, potential for cyanobacterial blooms, and terrestrial and emergent plant growth in sediments exposed by reduced water levels (ter Heerdt and Drost 1994; Bakker and Hilt 2016).

A synthesis of the abiotic and biotic responses to annual and novel winter water level drawdowns in littoral zones of lakes and reservoirs is summarized by Carmignani and Roy 2017. Climatic conditions also determine the capacity of a waterbody to support drawdown (Coops et al. 2003). Resources managers pursuing drawdown must carefully calculate the waterbody's water budget and the potential for increased cyanobacterial blooms in the future may reduce the number of suitable waterbodies (Callieri et al. 2014). Additionally, mild winters and groundwater seepage in some waterbodies may prevent dewatering, leading to reduced aquatic plant control (Cooke 1980). Complete freezing of sediment is more likely to control aquatic plants. Sediment exposure during warmer temperatures (>5° C) can also result in the additional benefit of oxidizing and compacting organic sediments (Scott Provost and Ted Johnson [DNR], *personal communication*). When drawdowns are conducted to improve migratory bird habitat, summer drawdowns prove to be more beneficial for species of shorebirds, as mudflats and shallow water are exposed to promote the production of and accessibility to invertebrates during late summer months that coincide with southward migration (Herwig and Gelvin-Innvaer 2015). Drawdowns conducted during mid-late summer can result in conditions that are favorable for cattails (*Typha* spp.) germination and expansion. However, cattails can be controlled if certain stressors are implemented in conjunction with a drawdown, such as cutting, burning or herbicide treatment during the peak of the growing season. The ideal situation is to cut cattail during a drawdown and flood over cut leaves when water is raised. However, this option is not always feasible due to soil conditions and equipment limitations.

Ecological Impacts of Water-level Drawdown

Artificial manipulation of water level is a major disturbance which can affect many ecological aspects of a waterbody. Because drawdown provides species-selective aquatic plant control, it can alter aquatic plant community composition and relative abundance and distribution of species (Boschilia et al. 2012; Keddy 2000). Sometimes this is the intent of the drawdown, which creates plant community characteristics that are desired for wildlife or fish habitat. Consecutive annual drawdowns may prevent the re-establishment of native aquatic plants or lead to reduced control of aquatic plant abundance as drawdown-tolerant species begin to dominate the community (Nichols 1975). Sediment exposure can also lead to colonization of emergent vegetation in the drawdown zone. In one study, four years of consecutive marsh drawdown led to dominance of invasive phragmites (*Phragmites australis subsp. australis*; ter Heerdt and Drost 1994). However, when drawdowns are conducted properly, it can provide a favorable response to native emergent plants for providing food and cover for migrating waterfowl in the fall. Population increases in emergent plant species such as bulrush (*Schoenoplectus* spp.), bur-reeds (*Sparganium* spp.), and wild rice (*Zizania palustris*) is often a goal of drawdowns, which provides a great food source for fish and wildlife, and provides important spawning and nesting habitat. Full or partial drawdowns that are conducted after wild rice production in the fall tend to favor early successional emergent germination such as wild rice and bulrush the following spring. Spring drawdowns are also possible for producing wild rice but must be done during a tight window following ice-out and slowly raised prior to the wild rice floating leaf stage.

Drawdown can also have various effects on ecosystem fauna. Drawdowns can influence the mortality, movement and behavior of native freshwater mussels (Newton et al. 2014). Although mussels can move with lowering water levels, they can be stranded and die if they are unable to move fast enough or get trapped behind logs or other obstacles (WDNR et al. 2006). Some mussels will burrow down into the mud or sand to find water but can desiccate if the water levels continue to lower (Watters et al. 2001). Maintaining a slow drawdown rate can allow mussels to respond and stranded individuals can be relocated to deeper water during the drawdown period to reduce mussel death (WDNR et al. 2006). Macroinvertebrate communities may experience reduced species diversity and abundance from changes to their environment due to drawdown and loss of habitat provided by aquatic plants (Wilcox and Meeker 1992; McEwen and Butler 2008). These effects may be reduced by considering benthic invertebrate phenology in determining optimal timing for drawdown release. Adequate moisture is required to support the emergence of many macroinvertebrate species and complete drawdown may also result in hardening of sediments which can trap some species (Coops et al. 2003). Reduced macroinvertebrate availability can have negative effects on waterfowl and game fish species which rely on macroinvertebrate food sources (Wilcox and Meeker 1992). Depending on the time of year, drawdown may also lead to decreased reproductive success of some waterfowl through nest loss, including common loon (*Gavia immer*) and red-necked grebe (*Podiceps grisegena*; Reiser 1998). However, drawdown may lead to increased production of annual plants and seed production, thereby increasing food availability for brooding and migrating waterfowl. Semi-aquatic mammals such as muskrats and beavers may also be adversely affected by water level drawdown (Smith and Peterson 1988, 1991). DNR Wildlife Management staff follow guidance to ensure drawdowns are timed with the seasons or temperature to minimize negative impacts to wildlife. Negative impacts to reptiles are possible during the spring if water is raised following a drawdown, as nests may be flooded. In the fall, negative impacts to reptiles and amphibians are possible if water is lowered when species are attempting to settle into sediments for hibernation. The impact may be reduced dissolved oxygen if they are below the water or freezing if the water is dropped below the point of hibernation (Herwig and Smith 2016a, 2016b). Surveying and relocation of stranded organisms may help to mitigate some of these impacts. In Wisconsin there are general provisions for conducting drawdowns for APM that are designed to mitigate or even eliminate potential negative impacts.

Water chemistry can also be affected by water level fluctuation. Beard (1973) describes a substantial algal bloom occurring the summer following a winter drawdown which provided successful aquatic plant control. Other studies reported reduced dissolved oxygen, severe cyanobacterial blooms with summer drawdown, or increased nutrient concentrations and reduced water clarity during summer drawdown for urban water supply (Cooke 1980; Geraldes and Boavida 2005; Bakker and Hilt 2016). Water clarity and trophic state may be improved when drawdown level is similar to a waterbody's natural water level regime (Christensen and Maki 2015).

Species Susceptibility to Water-level Drawdown

Not all plant species are susceptible to management by water level drawdown and some dry- or cold-tolerant species may benefit from it (Cooke 1980). Generally, plants and charophytes which reproduce primarily by seed benefit from drawdowns while those that reproduce vegetatively tend to be more negatively affected. Marsh vegetation can be dependent on water level fluctuation (Keddy and Reznicek 1986). Cooke (1980) provides a summary table of drawdown responses for 63 aquatic plant species. Watershield (Brasenia schreberi), fern pondweed (*Potamogeton robbinsii*), pond-lilies (*Nuphar* spp.) and watermilfoils (*Myriophyllum* spp.) tend to be controlled by drawdown. Increases in abundance associated with drawdown have often been seen for duckweed (*Lemna minor*), rice cutgrass (*Leersia oryzoides*) and slender naiad (*Najas flexilis*; Cooke 1980). One study showed drawdown reduced Eurasian watermilfoil (*Myriophyllum spicatum*) at shallow depths while another cautioned that Eurasian watermilfoil vegetative fragments may be able to grow even after complete desiccation (Siver et al. 1986; Evans et al. 2011). Similarly, a tank-simulated drawdown experiment suggested short-term summer drawdown may be effective in controlling monoecious hydrilla (*Hydrilla verticillata*; Poovey and Kay 1998). However, other studies have shown hydrilla fragments to be resistant to drying following drawdown (Doyle and Smart 2001; Silveira et al. 2009). A study on Brazilian waterweed (*Egeria densa*) showed that stems were no longer viable after 22 days of exposure due to drawdown (Dugdale et al. 2012).

Two examples of recent drawdowns in Wisconsin that were evaluated for their efficacy in controlling invasive aquatic plants occurred in Lac Sault Dore and Musser Lake, both in Price County, which were conducted in 2010 and 2013, respectively. Dam maintenance was the initial reason for these drawdowns, with the anticipated control of nuisance causing aquatic invasive species as a secondary benefit. Aquatic plant surveys showed that the drawdown in Lac Sault Dore resulted in a 99% relative reduction in the littoral cover of Eurasian watermilfoil when comparing pre- vs. post-drawdown frequencies. Native plant cover expanded following the drawdown and Eurasian watermilfoil cover has continued to remain low (82% relative reduction compared to pre-drawdown) as of 2017 (Onterra 2013). Lake-wide cover of curly-leaf pondweed in Musser Lake decreased following drawdown (63% relative reduction compared to pre-drawdown), and turion viability was also reduced. Reductions in native plant populations were observed, though population recovery could be seen in the second year following the drawdown (Onterra 2016). These examples of water-level drawdowns in Wisconsin show that they can be valuable approaches for aquatic invasive species control in some waterbodies. Water level reduction must be conducted such that a sufficient proportion of the area occupied by the target species is exposed. Numerous other single season winter drawdowns monitored in central Wisconsin by department staff show similar results (Scott Provost [DNR], *personal communication*). Careful timing and proper duration is needed to maximize control of target species and growth of favorable species.

S.3.5.Biological Control

Biological control refers to any method involving the use of one organism to control another. This method can be applied to both invasive and native plant populations, since all organisms experience growth limitation through various mechanisms (e.g., competition, parasitism, disease, predation) in their native communities. As such, when control of aquatic plants is desired it is possible that a growth limiting organism, such as a predator, exists and is suitable for this purpose.

Care must be taken to ensure that the chosen biological control method will effectively limit the target population and will not cause unintended negative effects on the ecosystem. The world is full of examples of biological control attempts gone wrong: for example, Asian lady beetles (*Harmonia axyridis*) have been introduced to control agricultural aphid pests. While the beetles have been successful in controlling aphid populations in some areas, they can also outcompete native lady beetles and be a nuisance to humans by amassing on buildings (Koch 2003). Additionally, a method of control that works in some Wisconsin lakes may not work in other parts of the state where differing water chemistry and/or biological communities may affect the success of the organism. The department recognizes the variation in control efficacy and well as potential unintentional effects of some organisms and is very cautious in allowing their use for control of aquatic plants.

Purple loosestrife beetles

The use of herbivorous insects to reduce populations of aquatic plants is another method of biocontrol. Several beetle species native to Eurasia (*Galerucella calmariensis*, *G. pusilla*, *Hylobius transversovittatus*, and *Nanophyes marmoratus*) have been well-studied and intentionally released in North America for their ability to suppress populations of the invasive wetland plant, purple loosestrife (*Lythrum salicaria*). These beetles only feed on loosestrife plants and therefore are not a threat to other wetland plant species (Kok et al. 1992; Blossey et al. 1994a, 1994b; Blossey and Schroeder 1995). The department implements a purple loosestrife biocontrol program, in which citizens rear and release beetles on purple loosestrife stands to reduce the plants' ability to overtake wetlands, lakeshores, and other riparian areas.

Beetle biocontrol can provide successful long-term control of purple loosestrife. The beetles feed on purple loosestrife foliage which in turn can reduce seed production (Katovich et al. 2001). This approach typically does not eradicate purple loosestrife but stresses loosestrife populations such that other plants are able to compete and coexist with them (Katovich et al. 1999). Depending on the composition of the plant community invaded by purple loosestrife and the presence of other non-native invasive species, further restoration efforts may be needed following biocontrol efforts to support the regrowth of beneficial native plants (McAvoy et al. 2016).

Several factors have been identified that may influence the efficacy of beetle biocontrol of purple loosestrife. Purple loosestrife beetles have for the most part been shown to be capable of successfully surviving and establishing in a variety of locations (Hight et al. 1995; McAvoy et al. 2002; Landis et al. 2003). The different species have different preferred temperatures for feeding and reproduction (McAvoy and Kok 1999; McAvoy and Kok 2004). In addition, one study suggests that the number of beetles introduced does not necessarily correlate with greater beetle colonization (Yeates et al. 2012). Disturbance, such as flooding and predation by other animals on the beetles, can also reduce desired effects on loosestrife populations (Nechols et al. 1996; Dech and Nosko 2002; Denoth and Myers 2005). Finally, one study suggests that the use of triclopyr amine for purple loosestrife control may be compatible with beetle biocontrol, although there may be negative effects on beetle egg-batch size or indirect effects if the beetle's food source is too greatly depleted (Lindgren et al. 1998). Some mosquito larvicides may harm purple loosestrife beetles (Lowe and Hershberger 2004).

Milfoil weevils

Similar to the use of beetles for biological control of purple loosestrife, the use of milfoil weevils (*Euhrychiopsis lecontei*) has been investigated in North America to control populations of non-native Eurasian and hybrid watermilfoils (*Myriophyllum spicatum* x *sibiricum*). This weevil species is native to North America and is often naturally present in waterbodies that contain native watermilfoils, such as northern watermilfoil (*M. sibiricum*). The weevils have the potential to damage Eurasian watermilfoil (*M. spicatum*) by feeding on stems and leaves and/or burrowing into stems. Weevils may reduce milfoil plant biomass, inhibit growth, and compromise buoyancy (Creed and Sheldon 1993; Creed and Sheldon 1995; Havel et al. 2017a). Damage caused to the milfoil tissue may then indirectly increase susceptibility to pathogens (Sheldon and Creed 1995).

In experiments, weevils have been shown to negatively impact Eurasian watermilfoil populations to varying degrees. Experiments by Creed and Sheldon (1994) found that plant weight was negatively affected when weevils were at densities of 1 and 2 larvae/tank, and Eurasian watermilfoil in untreated control tanks added more root biomass than those in tanks with weevils, suggesting that weevil larvae may interfere with the plant's ability to move nutrients. Similarly, experiments by Newman et al. (1996) found that weevils at densities of 6, 12, and 24 adults/tank caused significant decreases in Eurasian watermilfoil stem and root biomass, and that higher weevil densities generally produced more damage.

In natural communities, effects of weevils have been mixed, likely because waterbody characteristics may play a role in determining weevil effects on Eurasian watermilfoil populations in natural lakes. In a 56 ha (138 acre) pond in Vermont, weevil density was negatively associated with Eurasian watermilfoil biomass and distribution; Eurasian watermilfoil beds were reduced from 2.5 (6.2 acres) to 1 ha (2.5 acres) in one year, and biomass decreased by 4 to 30 times (Creed and Sheldon 1995). A survey of Wisconsin waterbodies conducted by Jester et al. (2000) revealed that most lakes containing Eurasian watermilfoil also contained weevils. Weevil abundance varied from functionally non-detectable to 2.5 weevils/stem and was positively associated with the presence of large, shallow Eurasian watermilfoil beds (compared to deep, completely submerged beds). There was no relationship between natural weevil abundance and Eurasian watermilfoil density between lakes. However, when the authors augmented natural weevil populations in plots in an attempt to achieve target densities of 1, 2, or 4/stem, they found that augmentation was associated with significant decreases in Eurasian watermilfoil biomass, stem density and length, and tips/stem (Jester et al. 2000). However, another more recent study conducted in several northern Wisconsin lakes found no effect of weevil stocking on Eurasian watermilfoil or native plant biomass (Havel et al. 2017a).

There are several factors to consider when determining whether weevils are an appropriate method of biocontrol. First, previous research has suggested that densities of at least 1.5 weevils per stem are required for control (Newman and Biesboer 2000). Adequate densities may not be achievable due to factors including natural population fluctuations, the amount of available milfoil biomass within a waterbody, the presence of insectivorous predators, such as bluegills (*Lepomis macrochirus*), and the availability of nearshore overwintering habitat (Thorstenson et al. 2013; Havel et al. 2017a). In addition, weevils fed and reproduce on native milfoil species and biocontrol efforts could potentially impact these species, although experiments conducted by Sheldon and Creed (2003) found that native milfoil weevil density was lower and weevils caused less damage than when they were found on Eurasian watermilfoil. Adult weevils spend their winters on land, so available habitat for adults must be present for a waterbody to sustain weevil populations (Reeves and Lorch 2011; Newman et al. 2001). Additionally, one study found that lakes with no Eurasian watermilfoil (despite the presence of other milfoil species) and lakes that had a recent history of herbicide treatment had lower weevil densities than similar, untreated lakes or lakes with Eurasian watermilfoil (Havel et al. 2017b).

Grass carp – not allowed in Wisconsin

The use of grass carp (*Ctenopharyngodon idella*) to control aquatic plants is not allowed in Wisconsin; they are a prohibited invasive species under ch. NR 40, Wis. Admin. Code, which makes it illegal to possess, transport, transfer, or introduce grass carp in Wisconsin.

Sterile (also known as triploid) grass carp have been used to control populations of aquatic plants with varying success (Pípalová 2002; Hanlon et al. 2000). Whether this method is effective depends on several factors. For instance, each individual fish must be tested to ensure sterility before stocking, which can be a time- and resource-consuming process. Since the sterile fish do not reproduce, it can be difficult to achieve the desired density in a given waterbody. In addition, grass carp, like many fish species, have dietary preferences for different plant species which must be considered (Pine and Anderson 1991). Further information summarizing the effects of stocking triploid grass carp can be found in Pípalová (2006), Dibble and Kovalenko (2009), and Bain (1993).

F

APPENDIX F

Forest Lake 2023 EWM Removal Report - Aquatic Plant Management, LLC

EST. 2013

AQUATIC PLANT MANAGEMENT

Forest Lake EWM Removal Report 2023

PO Box 1134 Minocqua, WI 54548

Forest Lake EWM Removal Summary 2023

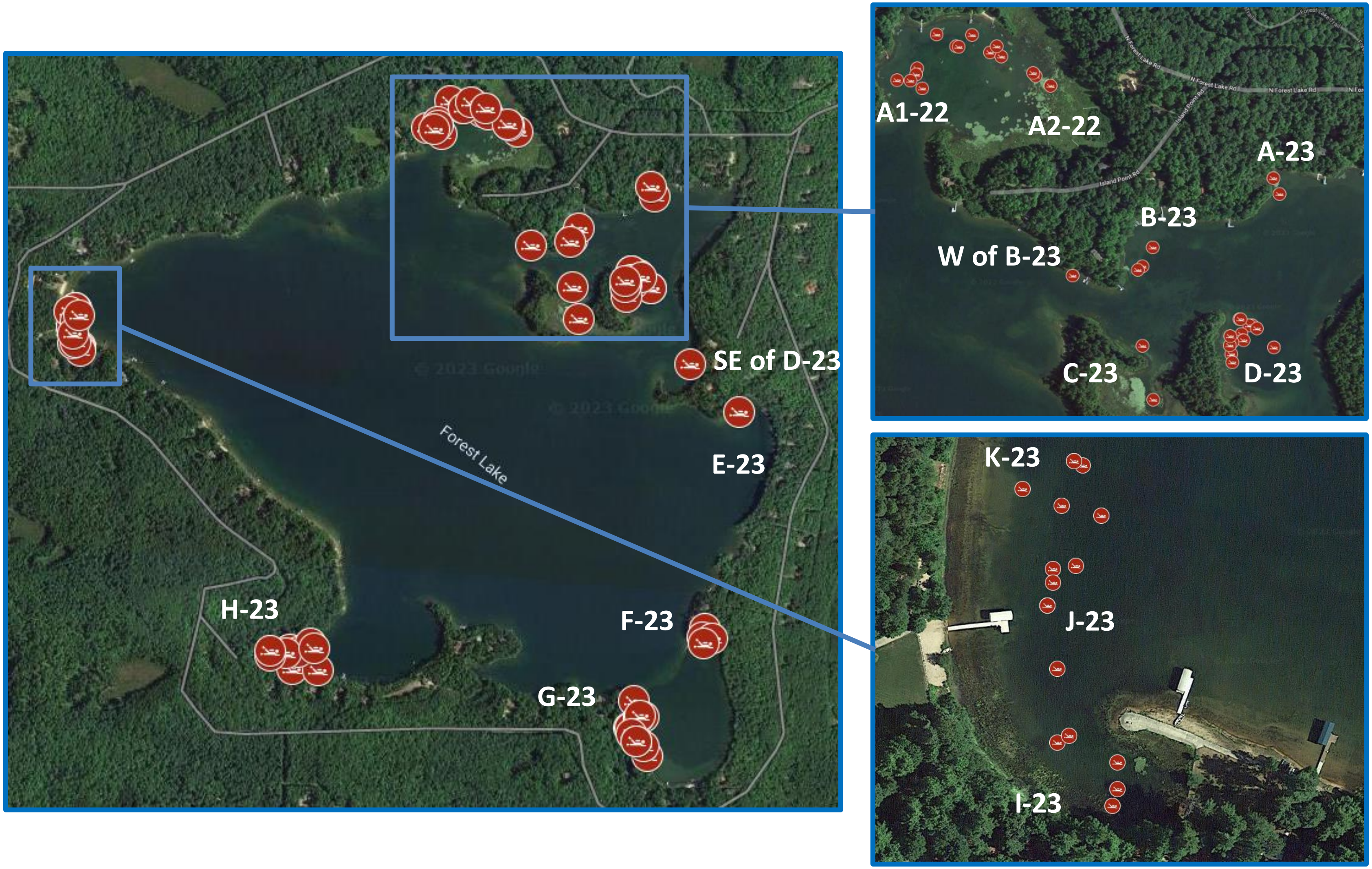
Dive Background: In July, August, and September, Aquatic Plant Management LLC (APM) conducted five (5) days of Diver Assisted Suction Harvesting (DASH) and thirteen (13) days of Hand Harvesting for Eurasian Watermilfoil (EWM) on Forest Lake in Vilas County, WI. The team focused their efforts at 15 sites as prioritized by the Forest Lake Association and Onterra LLC. In total APM was able to remove **283.5 cubic feet of EWM** from Forest Lake.

Service	Dive Locations	Avg. Water Depth	# of Dives	Underwater Dive Time	AIS Removed (cubic feet)
DASH	10	7.2	27	31.3	64.0
HH	15	6.4	57	79.4	219.5
Grand Total		**6.7**	**84**	**110.7**	**283.5**

Date	Weather Conditions	Water Temp (F)	Underwater Dive Time (hrs)	AIS Removed (cubic ft)
7/10/2023	Cloudy	73	8.2	13.5
7/11/2023	Sunny	73	7.3	37.0
7/12/2023	Partly Cloudy	72	6.9	5.5
7/13/2023	Cloudy	73	7.0	5.5
7/14/2023	Cloudy	72	1.8	2.5
7/21/2023	Sunny	72	6.1	13.5
7/24/2023	Sunny	73	6.7	18.5
7/25/2023	Periods of rain	69	5.4	6.0
8/8/2023	Partly Cloudy	70	6.5	38.0
8/10/2023	Partly Cloudy	70	6.7	11.0
8/18/2023	Sunny	72	5.8	38.0
8/21/2023	Cloudy	72	5.7	13.0
8/22/2023	Cloudy	70	4.4	5.0
8/24/2023	Sunny	69	12.6	25.5
8/28/2023	Sunny	70	7.3	16.0
8/29/2023	Partly Cloudy	70	5.4	11.0
9/1/2023	Sunny	71	6.9	24.0
Grand Total			**110.7**	**283.5**

Dive Highlights and Recommendations: The DASH crew came for the first week of 7/10 – 7/14 and spent the majority of their time at site D-22 in the northeastern section of the lake. The hand harvesting teams followed up of the next several weeks targeting lower density sites throughout the lake, along with return visits to some DASH sites. Overall, Forest Lake should continue to take an Integrated Pest Management (IPM) approach and evaluate different strategies to manage the EWM population on the lake. Continued monitoring and management efforts are important to prevent the spread of EWM throughout Forest Lake.

Map of Forest Lake Dive Sites



Detailed Diving Activities - July

Date	Dive Location	Latitude	Longitude	Underwater Dive Time (hrs)	AIS Removed (cubic ft)	AIS Density	Avg Water Depth (ft)	Native Species	Native By-Catch	Substrate Type
7/10/2023	D-23	46.15240	-89.37044	1.33	5.0	Single or Few	9.0	Northern Milfoil	0.5	Organic/Sand
7/10/2023	D-23	46.15240	-89.37051	0.83	2.0	Small Plant Colony	7.5	Northern Milfoil	0.0	Organic
7/10/2023	D-23	46.15249	-89.37071	0.50	1.0	Single or Few	6.0	Northern Milfoil	0.0	Organic/Sand
7/10/2023	D-23	46.15249	-89.37069	1.58	2.0	Single or Few	8.0	Northern Milfoil	0.5	Organic/Sand
7/10/2023	G-23	46.14205	-89.37061	0.42	1.0	Single or Few	4.5	None	0.0	Organic/Sand
7/10/2023	G-23	46.14141	-89.37075	3.50	2.5	Scattered	4.0	Northern Milfoil	1.0	Organic
7/11/2023	G-23	46.14141	-89.37079	0.58	1.5	Single or Few	3.5	Northern Milfoil	0.0	Organic/Sand
7/11/2023	A1-22	46.15644	-89.37663	2.75	15.0	Clumps	8.0	Northern Milfoil	0.0	Organic/Sand
7/11/2023	K-23	46.15157	-89.39013	2.00	11.0	Small Plant Colony	14.0	Northern Milfoil	1.0	Organic/Sand
7/11/2023	I-23	46.15068	-89.38998	2.00	9.5	Scattered	5.0	Northern Milfoil	1.0	Organic/Sand
7/12/2023	I-23	46.15060	-89.38998	1.25	1.0	Single or Few	4.0	Northern Milfoil	0.0	Organic/Sand
7/12/2023	H-23	46.14328	-89.38272	1.42	0.5	Single or Few	7.0	Northern Milfoil	0.0	Gravel
7/12/2023	B-23	46.15325	-89.37274	0.67	1.0	Clumps	6.0	Northern Milfoil	0.0	Organic
7/12/2023	F-23	46.14381	-89.36812	0.33	0.5	Single or Few	5.5	Northern Milfoil	0.0	Organic
7/12/2023	D-23	46.15198	-89.37088	2.50	2.0	Single or Few	14.5	Grasses	0.5	Organic
7/12/2023	A1-22	46.15595	-89.37787	0.75	0.5	Clumps	5.0	Northern Milfoil	0.0	Organic
7/13/2023	D-23	46.15208	-89.36999	0.92	0.5	Single or Few	8.0	Northern Milfoil	0.0	Organic
7/13/2023	H-23	46.14328	-89.38272	0.25	0.0	Single or Few	6.0	Northern Milfoil	0.0	Organic
7/13/2023	A1-22	46.15661	-89.37704	1.67	1.5	Single or Few	5.5	Northern Milfoil	0.0	Organic/Sand
7/13/2023	A2-22	46.15635	-89.37592	0.58	0.5	Single or Few	6.0	Northern Milfoil	0.0	Gravel
7/13/2023	J-23	46.15126	-89.39026	1.67	2.0	Clumps	11.0	Northern Milfoil	0.5	Organic
7/13/2023	D-23	46.15208	-89.36999	1.92	1.0	Single or Few	9.0	Northern Milfoil	0.0	Organic
7/14/2023	G-23	46.14167	-89.37025	0.42	1.0	Small Plant Colony	7.0	None	0.0	Organic
7/14/2023	G-23	46.14072	-89.37012	0.58	0.5	Small Plant Colony	8.0	None	0.0	Sand
7/14/2023	D-23	46.15227	-89.37067	0.33	0.0	Single or Few	8.5	Northern Milfoil	0.0	Organic/Sand
7/14/2023	H-23	46.14283	-89.38256	0.25	0.5	Clumps	7.0	Northern Milfoil	0.0	Organic
7/14/2023	A1-22	46.15583	-89.37734	0.25	0.5	Clumps	6.5	Northern Milfoil	0.0	Organic/Sand
7/21/2023	D-23	46.15218	-89.37062	1.17	5.5	Scattered	8.0	Pondweeds	0.5	Organic
7/21/2023	D-23	46.15235	-89.37035	0.75	1.0	Scattered	9.0	Pondweeds	0.5	Organic
7/21/2023	D-23	46.15186	-89.37086	0.92	1.0	Scattered	8.0	Pondweeds	0.5	Organic
7/21/2023	G-23	46.14162	-89.37043	1.33	3.0	Scattered	4.0	Pondweeds	0.5	Organic/Sand
7/21/2023	G-23	46.14110	-89.37051	1.92	3.0	Scattered	4.0	Pondweeds	0.5	Organic/Sand
7/24/2023	G-23	46.14095	-89.37025	2.00	3.5	Single or Few	5.0	None	0.0	Organic/Sand
7/24/2023	F-23	46.14359	-89.36819	0.50	1.0	Single or Few	6.0	Northern Milfoil	0.0	Organic/Sand
7/24/2023	A2-22	46.15629	-89.37569	1.08	4.0	Clumps	6.0	Northern Milfoil	0.0	Organic/Sand
7/24/2023	A1-22	46.15612	-89.37745	1.50	3.5	Single or Few	7.0	Northern Milfoil	0.0	Organic/Sand
7/24/2023	A2-22	46.15587	-89.37466	0.83	6.0	Clumps	4.5	Northern Milfoil	0.5	Organic
7/24/2023	K-23	46.15145	-89.39022	0.75	0.5	Single or Few	7.0	Northern Milfoil	0.0	Organic/Sand
7/25/2023	E-23	46.14906	-89.36693	1.25	1.0	Single or Few	10.0	Grasses	0.0	Organic/Sand
7/25/2023	H-23	46.14280	-89.38169	1.08	1.0	Single or Few	10.0	Grasses	0.0	Organic/Sand
7/25/2023	H-23	46.14330	-89.38268	0.58	1.0	Single or Few	10.0	Grasses	0.0	Organic/Sand
7/25/2023	A-23	46.15430	-89.36987	1.00	1.0	Single or Few	10.0	Grasses	0.0	Organic/Sand
7/25/2023	B-23	46.15353	-89.37252	0.75	1.0	Single or Few	10.0	Grasses	0.0	Organic/Sand
7/25/2023	J-23	46.15122	-89.39026	0.75	1.0	Single or Few	10.0	Grasses	0.0	Organic/Sand
Total	**44**			**49.41**	**102.0**					

Detailed Diving Activities – Aug & Sep

Date	Dive Location	Latitude	Longitude	Underwater Dive Time (hrs)	AIS Removed (cubic ft)	AIS Density	Avg Water Depth (ft)	Native Species	Native By-Catch	Substrate Type
8/8/2023	J-23	46.15115	-89.39028	1.75	12.0	Scattered	5.5	Elodea	0.5	Organic/Sand
8/8/2023	C-23	46.15210	-89.37273	1.58	16.5	Dominant	7.5	Elodea	0.5	Organic/Sand
8/8/2023	J-23	46.15115	-89.39028	0.75	3.5	Scattered	5.5	Elodea	0.5	Organic/Sand
8/8/2023	J-23	46.15115	-89.39028	0.67	2.5	Highly Scattered	9.0	Elodea	0.0	Organic/Sand
8/8/2023	I-23	46.15055	-89.39000	1.33	2.0	Scattered	4.0	Grasses	0.0	Organic/Sand
8/8/2023	H-23	46.14343	-89.38193	0.42	1.5	Scattered	8.0	Grasses	0.0	Organic/Sand
8/10/2023	A-23	46.15453	-89.37000	1.50	0.5	Single or Few	3.5	Grasses	0.0	Organic
8/10/2023	K-23	46.15158	-89.39017	1.17	5.0	Small Plant Colony	8.0	None	0.0	Organic/Sand
8/10/2023	H-23	46.14325	-89.38238	1.33	1.0	Single or Few	7.5	None	0.0	Organic/Sand
8/10/2023	K-23	46.15158	-89.39017	1.17	4.0	Scattered	8.5	None	0.0	Organic/Sand
8/10/2023	G-23	46.14169	-89.37040	1.50	0.5	Single or Few	5.0	None	0.0	Sand
8/18/2023	C-23	46.15132	-89.37252	1.00	9.0	Scattered	5.5	Elodea	1.0	Organic
8/18/2023	A1-22	46.15643	-89.37658	1.58	13.5	Small Plant Colony	5.5	Grasses	2.0	Organic
8/18/2023	A1-22	46.15603	-89.37747	1.58	9.0	Scattered	5.5	Grasses	2.0	Organic
8/18/2023	A1-22	46.15660	-89.37630	1.17	4.5	Scattered	5.5	Grasses	1.0	Organic
8/18/2023	G-23	46.14125	-89.37068	0.50	2.0	Scattered	3.0	None	0.0	Organic
8/21/2023	G-23	46.14108	-89.37050	1.33	4.0	Scattered	3.0	Grasses	0.5	Organic
8/21/2023	G-23	46.14108	-89.37050	1.17	2.0	Scattered	3.0	Grasses	0.5	Organic
8/21/2023	A1-22	46.15594	-89.37759	0.92	1.0	Scattered	5.5	Northern Milfoil	0.5	Organic
8/21/2023	K-23	46.15150	-89.39039	1.00	3.0	Small Plant Colony	7.5	Northern Milfoil	0.5	Organic
8/21/2023	I-23	46.15074	-89.39024	1.25	3.0	Scattered	4.0	Northern Milfoil	0.5	Organic
8/22/2023	W of B-23	46.15312	-89.37420	0.50	0.5	Single or Few	3.5	None	0.0	Organic/Gravel
8/22/2023	I-23	46.15076	-89.39019	1.58	1.5	Scattered	4.5	Pondweeds	0.5	Organic/Gravel
8/22/2023	J-23	46.15127	-89.39016	1.00	1.5	Scattered	7.5	Pondweeds	0.5	Organic
8/22/2023	H-23	46.14322	-89.38281	1.33	1.5	Scattered	5.5	Northern Milfoil	0.5	Organic
8/24/2023	A2-22	46.15601	-89.37497	2.83	3.5	Clumps	6.0	Northern Milfoil	3.0	Organic
8/24/2023	H-23	46.14315	-89.38334	2.75	4.5	Clumps	7.5	Elodea	0.0	Organic/Sand
8/24/2023	D-23	46.15215	-89.37086	0.42	1.0	Clumps	7.0	None	0.0	Organic
8/24/2023	A2-22	46.15605	-89.37502	3.50	7.0	Clumps	6.0	Northern Milfoil	0.5	Organic/Sand
8/24/2023	H-23	46.14326	-89.38335	2.75	6.0	Single or Few	9.0	None	0.0	Organic/Sand
8/24/2023	D-23	46.15211	-89.37091	0.33	3.5	Single or Few	6.0	None	0.0	Organic/Sand
8/28/2023	H-23	46.14334	-89.38180	1.92	4.0	Scattered	5.5	Northern Milfoil	0.5	Organic/Sand
8/28/2023	B-23	46.15320	-89.37283	1.58	4.0	Scattered	4.0	Northern Milfoil	0.5	Organic/Sand
8/28/2023	D-23	46.15224	-89.37090	1.83	4.0	Scattered	7.0	Grasses	1.0	Organic/Sand
8/28/2023	F-23	46.14344	-89.36814	1.92	4.0	Scattered	6.5	Northern Milfoil	0.5	Organic/Sand
8/29/2023	A2-22	46.15644	-89.37579	1.83	3.0	Scattered	5.0	Grasses	0.5	Organic/Sand
8/29/2023	J-23	46.15096	-89.39024	1.08	1.0	Scattered	6.0	Grasses	0.5	Organic/Sand
8/29/2023	K-23	46.15142	-89.39005	0.83	4.0	Scattered	7.5	Grasses	0.5	Organic/Sand
8/29/2023	F-23	46.14358	-89.36785	1.67	3.0	Scattered	6.0	Grasses	0.5	Organic/Sand
9/1/2023	SE of D-23	46.15024	-89.36863	6.92	24.0	Clumps	8.0	Grasses	1.5	Organic/Sand
Total	**40**			**61.24**	**181.5**					