1.0 INTRODUCTION

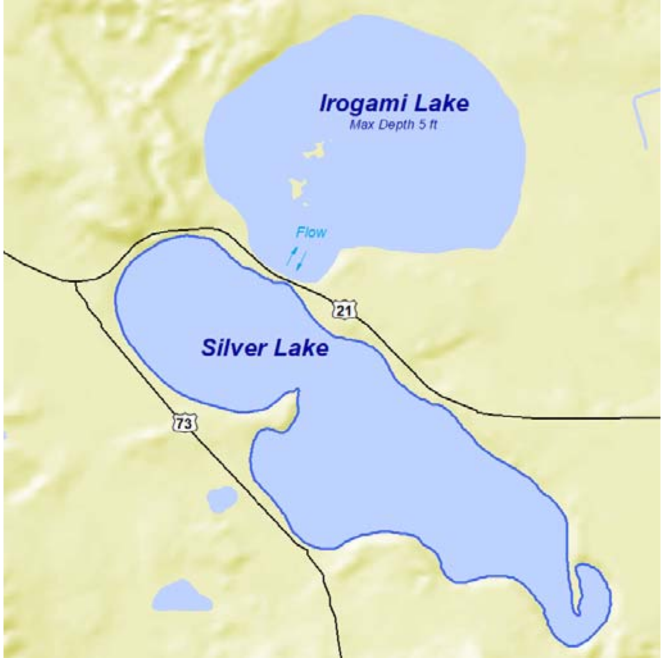


Figure 1.0-1. Silver Lake, Waushara County, Wisconsin.

[Big] Silver Lake, Waushara County, is a seepage lake with a maximum depth of 48 feet and a mean depth of 21 feet. The lake area as determined through a heads-up digitization of the lake from a 2015 aerial photograph is approximately 360.3 acres, whereas the WDNR website lists the lake as 328 acres. This mesotrophic lake has a relatively small watershed when compared to the size of the lake (3.5:1). Only when water levels are near full pool does water exchange occur with Irogami Lake via a culvert under State Hwy 21 (Figure 1.0-1). Four exotic species are known to exist in Silver Lake: banded mystery snail, curly-leaf pondweed (*Potamogeton crispus,* CLP), Eurasian water milfoil (*Myriophyllum* spicatum, EWM), and zebra mussel. Genetic analysis confirms that the invasive milfoil population is comprised of both EWM and hybrid water milfoil (*M. spicatum x sibiricum*, HWM). Subsequent discussion using "HWM" will represent the collective invasive milfoil population of Silver Lake unless specifically referenced otherwise.

The Silver Lake Management District (SLMD) is the local citizen-based organization leading the management of Silver Lake. The group has worked for years to protect and enhance the lake, including an increased effort in recent years to control HWM within the lake. The SLMD realizes that in order to effectively control the HWM population in Silver Lake, aggressive herbicide strategies need to be implemented, which could have increased collateral effects on the native aquatic plant community compared with more-typical use rates employed for pure-strain EWM control projects.

The 2014 Aquatic Plant Management (APM) Plan recommended the SLMD initiate a large-scale (aka whole-lake) herbicide treatment targeting HWM in Silver Lake. Based on discussion with industry professionals and following herbicide challenge testing (SePRO, unpublished), two large-scale herbicide use patterns were discussed within the APM Plan: liquid fluridone and combination treatment of 2,4-D/endothall. Both of these strategies were not commonly used in Wisconsin at that time, so an additional herbicide, granular triclopyr, was also entertained during discussions that occurred in late-winter of 2013-14 between Stantec, SLMD, and WDNR. Ultimately, a large-scale granular triclopyr (Renovate OTF®) treatment occurred in early-June 2014 targeting 180-200 ppb acid equivalent (ae) lake-wide.

Triclopyr concentrations fell short of achieving target levels, with the following hypotheses formulated by Onterra: uneven lake-wide mixing, expansion of mixing zone (i.e. epilimnion) following weather events, inaccurate bathymetric data which calculations were based off, and herbicide granules releasing below epilimnion. The point-intercept data indicate that HWM was reduced lake-wide from 33.5% in 2013 (*year before treatment*) to 7.8% in 2014 (*year of treatment*); a 76.7% decline. SLMD members suspect that if the point-intercept survey would have occurred a month or two later in 2014, the HWM frequency of occurrence would have been higher as HWM was in the process of rebounding during the late-August survey. It is clear from 20.0% frequency recorded in the 2015 point-intercept survey (*year after treatment*) that the 2014 treatment resulted in only seasonal HWM control, likely greatly injuring HWM during the *year of treatment* but the population was in the process of recovering during 2015.

1.1 2016 Fluridone Treatment Summary

Silver Lake riparian property owners have voiced increased frustration over the 2014 treatment results and the overall historic lack of success controlling HWM within the lake. In response, the SLMD contracted with Onterra, LLC during May 2015 to provide technical direction as they pursued their goal to implement a large-scale herbicide treatment strategy during spring of 2016. Onterra developed a preliminary three-year control and monitoring strategy in which a large-scale herbicide treatment would occur in year two of the project.

Three herbicide use patterns were investigated for applicability on Silver Lake in 2016: combination of liquid 2,4-D/endothall, liquid fluridone, and pelletized fluridone. Ultimately, the SLMD decided to move forward with a pelletized fluridone to target HWM in Silver Lake in 2016. Fluridone is a systematic herbicide that disrupts photosynthetic pathways (carotenoid synthesis inhibitor). The herbicide degrades via photolysis (some microbial degradation may also occur) and requires long exposure times (>90 days) to cause mortality to HWM. Herbicide concentrations within the lake are kept at target levels by periodically adding additional herbicide ("bump treatment") over the course of the summer based upon herbicide concentration monitoring results.

While liquid fluridone treatments result in a high initial concentration that tapers off over time as the herbicide degrades, pelletized fluridone treatments gradually reach peak concentrations over time (extended release) and result in a lower, sustained lake-wide herbicide concentration. This use-pattern of fluridone appears to demonstrate increased selectivity towards native plants in some field trials.

For Silver Lake, SePRO recommended a 4 ppb initial treatment, with an understanding that the measured concentrations within the lake would be approximately 2-3 ppb because of the extended release rate, herbicide degradation, and plant uptake. Once measured herbicide concentrations from the lake fall below 2 ppb, additional bump treatments would occur to keep the concentration between 2-3 ppb. The water levels at the time of the treatment planning were too low for water exchange with Irogami Lake to be a factor in herbicide dissipation.

The treatment included application of 941.5 lbs of pelletized fluridone (Sonar One ®, SePRO) over 86.4 acres of the littoral zone known to contain HWM (based on the 2015 HWM Peak-Biomass Mapping Survey. The initial herbicide treatment was conducted by on May 26, 2016 using a Vortex gas powered spreader system. Based upon reviewing the measured herbicide concentration during the summer as well as technical advice from SePRO, 2 ppb bump treatments of pelletized fluridone (Sonar One®) were conducted on July 21 and September 1 by Clean Lakes. The final dosing of these treatments was based on a mixing zone down to 21 feet and includes application of 655.9 lbs of pelletized fluridone over the same 86.4 acres where the initial application occurred.

Figure 1 shows the results of the fluridone monitoring that occurred in association with the 2016 large-scale treatment on Silver Lake. It was anticipated that herbicide degradation would be minimal over the winter as fluridone is primarily broken down by sunlight. Average surface herbicide concentrations following the ice-out event on Silver Lake were basically unchanged to before ice-on levels. A July 7, 2017 herbicide concentration sample (407 days after initial application) confirmed that fluridone was still present within the lake, but only slightly above the detection levels (Figure 1.1-1).

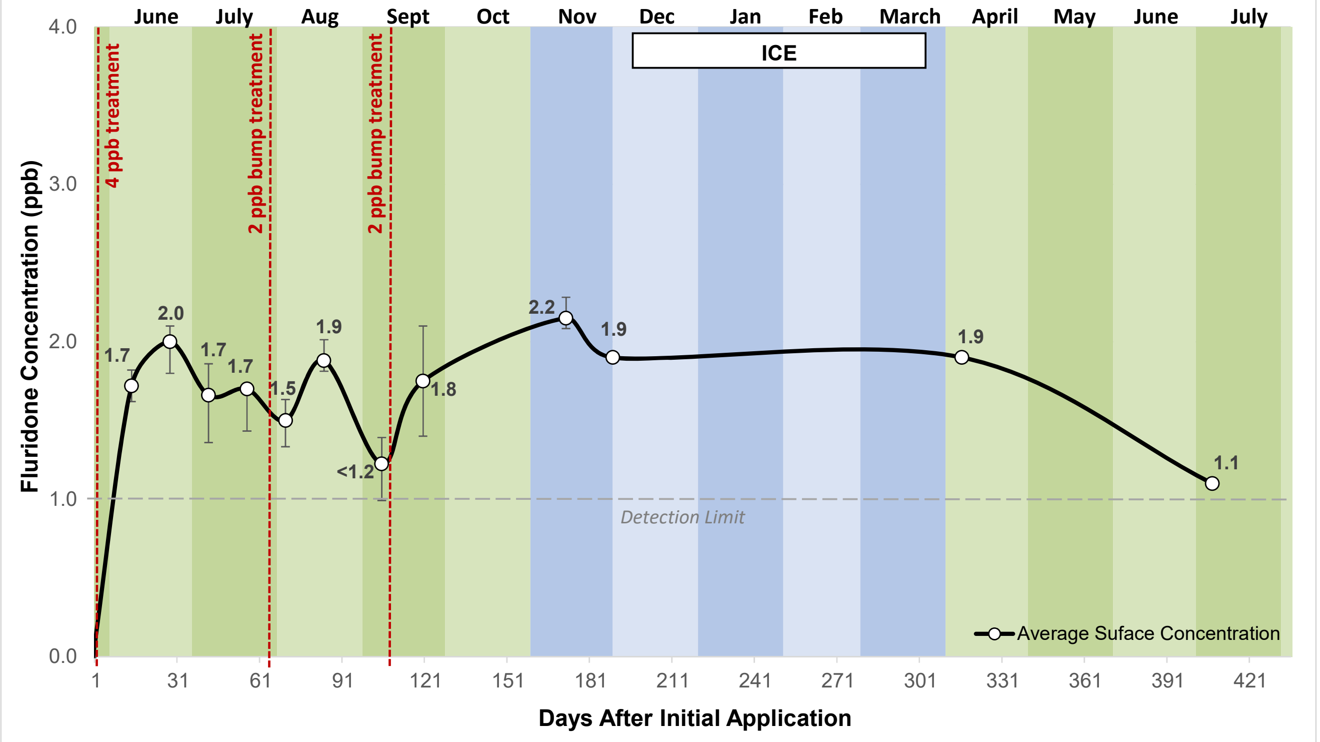


Figure 1.1-1. Silver Lake Herbicide Concentration Monitoring Results from five monitoring locations.

Many lake groups initiate a large-scale herbicide strategy with the intention of implementing smaller-scale control measures (e.g. herbicide spot treatments, hand-removal) when HWM begins rebounding. This is referred to as Integrated Pest Management (IPM) and the approach has shown promise on many lakes. However, the HWM population rebounds on many lakes in a lake-wide fashion that may not lend well to implementing IPM.

Following discussions with the SLMD, a professional hand-harvesting strategy was devised for 2017 such that divers would remove the HWM located during the June 2017 ESAIS survey and then conduct scuba reconnaissance surveys at various locations around the lake where dense colonies of HWM had been mapped during the 2015 growing season. Six subsequent days of scuba reconnaissance allowed the professional hand-harvesting firm to visit all 36-designated search areas, spending a total of 38.95 hours underwater. Divers encountered varying amounts of milfoil, which was presumably all EWM/HWM but difficult identification at this growth stage is noted. The hand-harvesting firm indicated the plants that they removed were typically small single-stalked plants that did not appear to be growing out of a large root crown. It is unclear if this represents plants that survived the treatment or was a result of germination from a seedbank or sprouting from asexual turions (i.e. winter buds).

Surveys conducted in 2017 on Silver Lake show that the large-scale herbicide treatments conducted in 2016 were successful in meeting the control objectives. Professional hand-harvesting efforts in 2017 likely aided in maintaining the HWM population at a relatively low level. The native plant community exhibited some reduction in 2017, likely from a combination of the large scale fluridone treatment and environmental factors from the large amounts of precipitation and corresponding increase in water levels.

1.2 2018 HWM Management Strategy

Based on the HWM population that was documented during the late-summer 2017 mapping survey, a management strategy that including professional hand-harvesting was determined to be the most appropriate method for implementation in 2018. A set of HWM mapping surveys were used within this project to coordinate and qualitatively monitor the hand-harvesting efforts (Figure 1.2-1). The first monitoring event on Silver Lake in 2018 was the Early Season Aquatic Invasive Species Survey (ESAIS). This late-spring/early-summer survey provides an early look at the lake to help guide the hand-harvesting management to occur on the system. Following the hand-harvesting, Onterra ecologists completed the Late-Summer EWM Peak-Biomass Survey, the results of which serve as a post-treatment assessment of the hand-harvesting. The SLMD initiated an aggressive hand harvesting program with the goal of reducing the EWM population within the target areas.

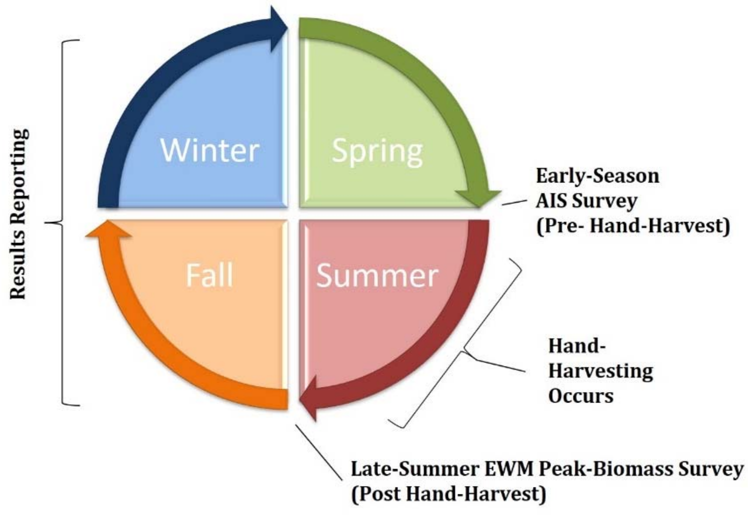


Figure 1.2-1. Hand-Harvesting Strategy Timeline. Includes potential hand-harvesting efforts which may or may not take place.

Additionally, a point-intercept survey was conducted in the summer of 2018 to be compared to previous surveys which will allow for an understanding of the aquatic plant populations two years following treatment.

2.0 2018 AQUATIC PLANT MONITORING RESULTS

2.1 Early-Season AIS Survey (ESAIS) (Pre-Hand-Harvesting)

Onterra field crews completed the ESAIS survey on Silver Lake on June 22, 2018. Crews noted favorable weather conditions during the survey with partly sunny skies and light winds. During the survey, the entire littoral area of the lake was surveyed through visual observations from the boat and the HWM & CLP populations were mapped. Field crews supplemented the visual survey by deploying a submersible camera along with periodically doing rake tows at locations in which dense HWM colonies were mapped in previous surveys. The HWM and CLP population was mapped using sub-meter GPS technology by using either 1) point-based or 2) area-based methodologies. Large colonies >40 feet in diameter are mapped using polygons (areas) and were qualitatively attributed a density rating based upon a five-tiered scale from *Highly Scattered* to *Surface Matting*. Point-based techniques were applied to AIS locations that were considered as *Small Plant Colonies* (<40 feet in diameter), *Clumps of Plants*, or *Single or Few Plants*.

The curly-leaf pondweed (CLP) population that was mapped during the survey was found to be of relatively low densities and present in several areas of the lake. Slightly more CLP was observed compared to the 2017 ESAIS survey. The largest area of CLP that was mapped in 2018 was near Fox Tail Bay on the southeast end of the lake where a *highly scattered* colony approximately 1.3 acres in size was delineated. The majority of the-wide population consisted of *clumps of plants* or *single or few plants* (Map 1).

The HWM population mapped during the ESAIS survey are displayed on Map 2. A *highly scattered* colony was located within Fox Tail Bay whereas all other occurrences in the lake were mapped with point-based methodologies as *singles, clumps* or *small plant colonies*. From this survey, a final 2018 hand-harvesting strategy was developed that included seven priority harvesting sites as well as 37 additional dive sites (see table embedded on Map 2). The seven priority harvesting sites were broken up further into 1st, 2nd and 3rd level priority sites and target the largest known populations of HWM in the lake and totaled 3.1 acres. The 37 dive sites were carried over from the sites that were developed as a part of the 2017 hand-harvesting/dive reconnaissance program. Each of these sites are located in areas that had previously contained some of the largest and most dense HWM colonies in the lake before the large-scale herbicide treatment. It was recommended that professional harvesting efforts focus on the highest priority sites first before moving onto other sites. Onterra provided the spatial data reflecting the ESAIS results to the professional harvesting firm to aid in the removal efforts.

2.2 Professional Hand-Harvesting Actions

The SLMD contracted with Aquatic Plant Management, LLC in 2018 to provide professional hand-harvesting services of HWM. AIS removal specialists from APM completed DASH services over twelve days between August 13 and August 29, harvesting approximately 621.3 cubic feet of HWM from Silver Lake (Table 2.2-1). The harvesting teams characterized most of the EWM growth in the lake as *clumps of plants surrounded by scattered individual plants* (APM Summary Report, Appendix A). The harvesting crews from APM also noted heavy native plants in Fox Tail Bay that ultimately resulted in discontinuing targeting this site due to concerns for diver safety. Additional details related to the professional harvesting actions are included in Appendix A.

Table 2.2-1. Silver Lake, 2018 professional hand-harvesting activities.

Site	Time spent underwater (Hours)	HWM Harvested (Cubic Feet)
A	14.29	181.5
B	5.31	59.5
C	16.17	138
D	3.92	60
E	6.09	37.5
F	14.96	76.8
G	4.42	23
Fox Tail	8	23
Other	5.33	22
Totals	**78.49**	**621.3**

2.3 Late-Summer HWM Peak-biomass Survey (Post Hand-Harvesting)

The HWM population was mapped on October 4, 2018 following the completion of the hand-harvesting efforts. During the survey, Onterra field crews meandered the littoral zone of the lake and mapped HWM populations using sub-meter GPS technology. Conditions were favorable during the survey with mostly sunny skies and light winds. In addition to visually scouring the lake from the surface, Onterra field crews lowered a submersible camera at each of the scuba reconnaissance survey locations.

The results of the late-summer HWM mapping survey are displayed on Map 3. The largest concentration of HWM plants was found in the shallower waters in Fox Tail Bay where approximately 1.8 acres of colonized plants were mapped. Outside of Fox Tail Bay, no other HWM locations in the lake were mapped with area-based methodologies (polygons) but rather, were mapped with point-based methods including *single or few plants*, *clumps of plants*, or *small plant colonies*. It is possible that some HWM escaped detection during the survey in cases where new growth characterized by short-statured plants would not have been visible.

Professional Hand-Harvesting Site Assessments

The sites that were targeted for professional harvesting are highlighted in Figures 2.3-1 through 2.3-3 where the left frame shows the pre-harvesting HWM population mapped in June 2018 and the right frame show the post-harvesting HWM population mapped in October 2018.

Site A-18: Before harvesting efforts, site A-18 contained numerous *single or few plant* HWM occurrences in addition to several *clumps of plants*. 181.5 cubic feet of HWM were removed from site A-18 during professional hand-harvesting efforts (Table 2.2-1). During the HWM peak-biomass survey, the site contained several areas where *single or few plants* and *clumps* were still present, as well as one *small plant colony* located on the east side of the permitted area (Figure 2.3-1).

Site B-18: During the 2018 ESAIS survey, site B-18 contained multiple *single or few plant* points. Approximately 59.5 cubic feet of HWM were removed during hand-harvest operations, and the HWM peak-biomass survey showed a reduced number of *single or few plant* points, in addition to one *clump of plants* (Figure 2.3-1).

Site C-18: Site C-18 was given first priority for hand-harvesting control efforts in 2018. The ESAIS survey recorded multiple *small plant colonies*, *clumps*, and *single plants* throughout the site. After over 16 hours underwater and 138 cubic feet of HWM removed, no *small plant colonies* were mapped in the permitted area. *Clumps of plants* and *single or few plant* occurrences were recorded within the site during the HWM peak-biomass survey (Figure 2.3-1).

Site D-18: The 2018 ESAIS survey indicated a *small plant colony*, a *clump of plants*, and a handful of *single or few plants* within site D-18. After harvesting 60 cubic feet of HWM, no HWM plants were mapped at site D-18 during the HWM peak-biomass survey (Figure 2.3-2).

Site E-18: A number of *single or few plant* points were mapped in the 2018 ESAIS survey within site E-18. After harvesting efforts yielded 37.5 cubic feet of HWM, only one *single or few plants* point was mapped in the site during the HWM peak-biomass survey (Figure 2.3-2).

Site F-18: A number of *single or few plants* points were mapped in Site F-18 during the 2018 ESAIS. Harvesting efforts removed 76.8 cubic feet of HWM from the site over just under 15 hours of dive time. During the HWM peak- biomass survey, one *single or few plants* point was mapped within the permitted area (Figure 2.3-2).

Site G-18: Before harvesting efforts, a handful of *single or few plant* points were mapped within site G-18. A total of 23 cubic feet of HWM was removed and the HWM peak-biomass survey found one *clump* of HWM present in the site (Figure 2.3-3).

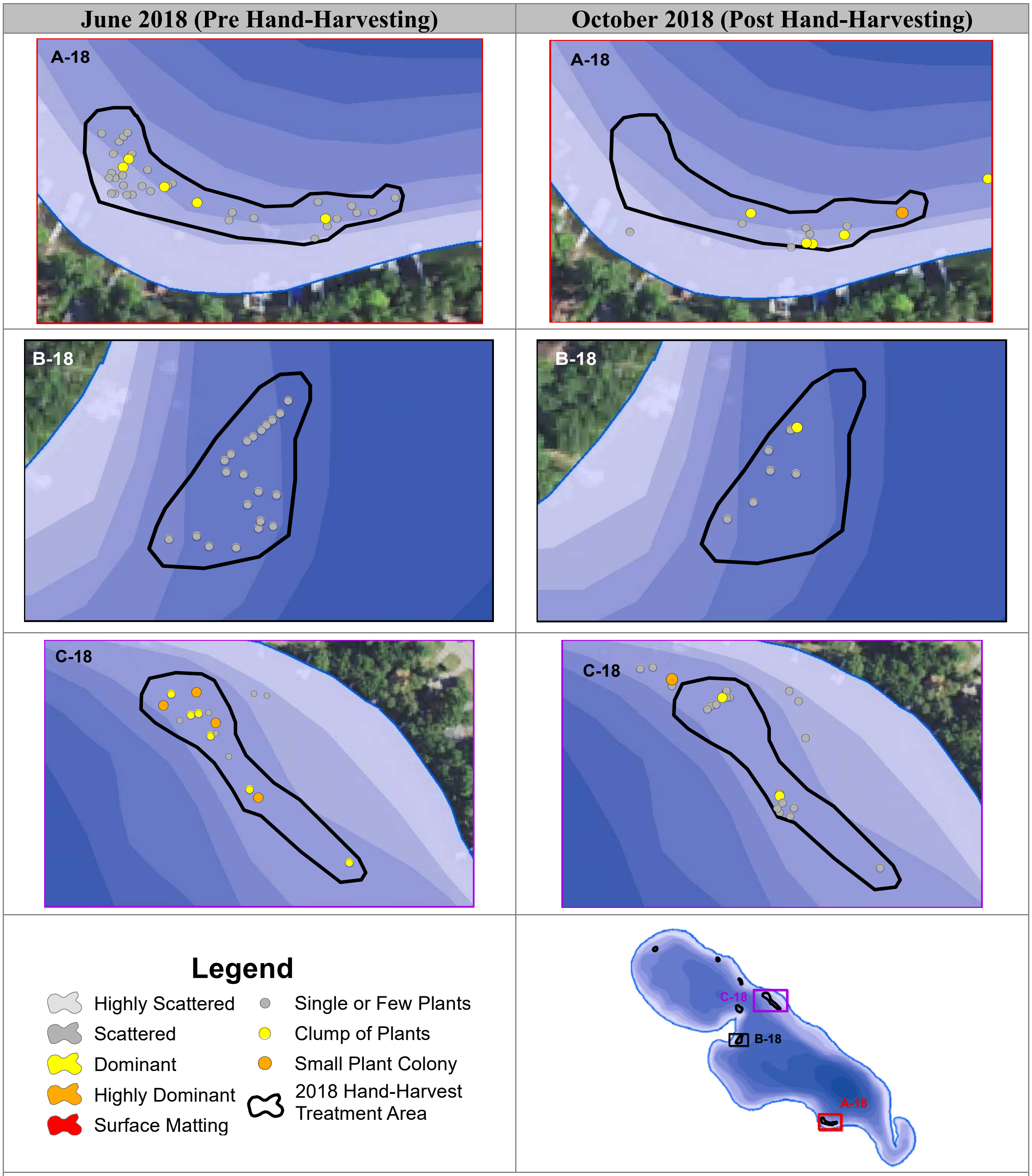


Figure 2.3-1. HWM Populations from before (June 2018) and after (October 2018) Professional Hand-Harvesting Efforts at sites A-18, B-18 & C-18 in Silver Lake.

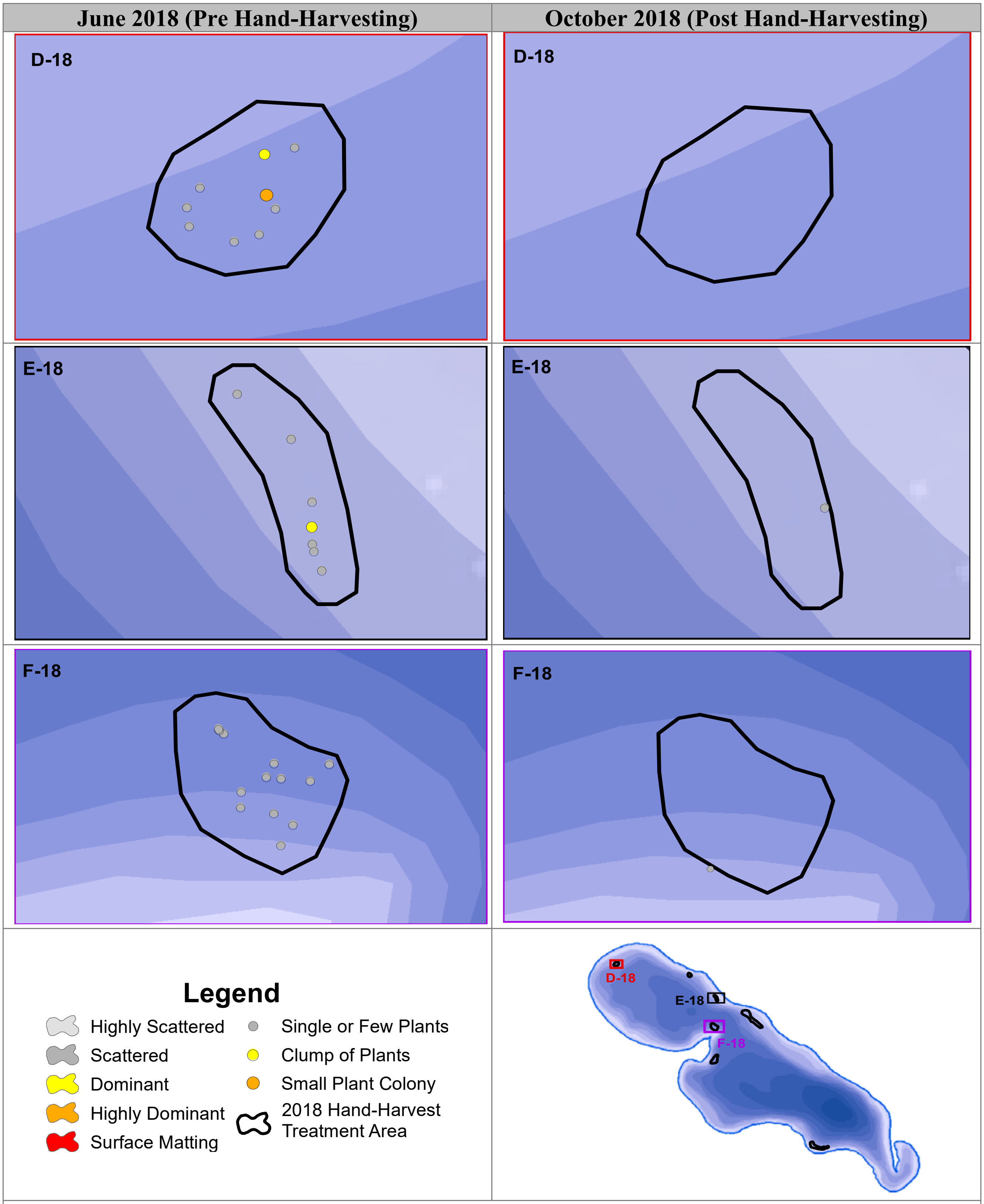


Figure 2.3-2. HWM Populations from before (June 2018) and after (October 2018) Professional Hand-Harvesting Efforts at sites D-18, E-18 & F-18 in Silver Lake.

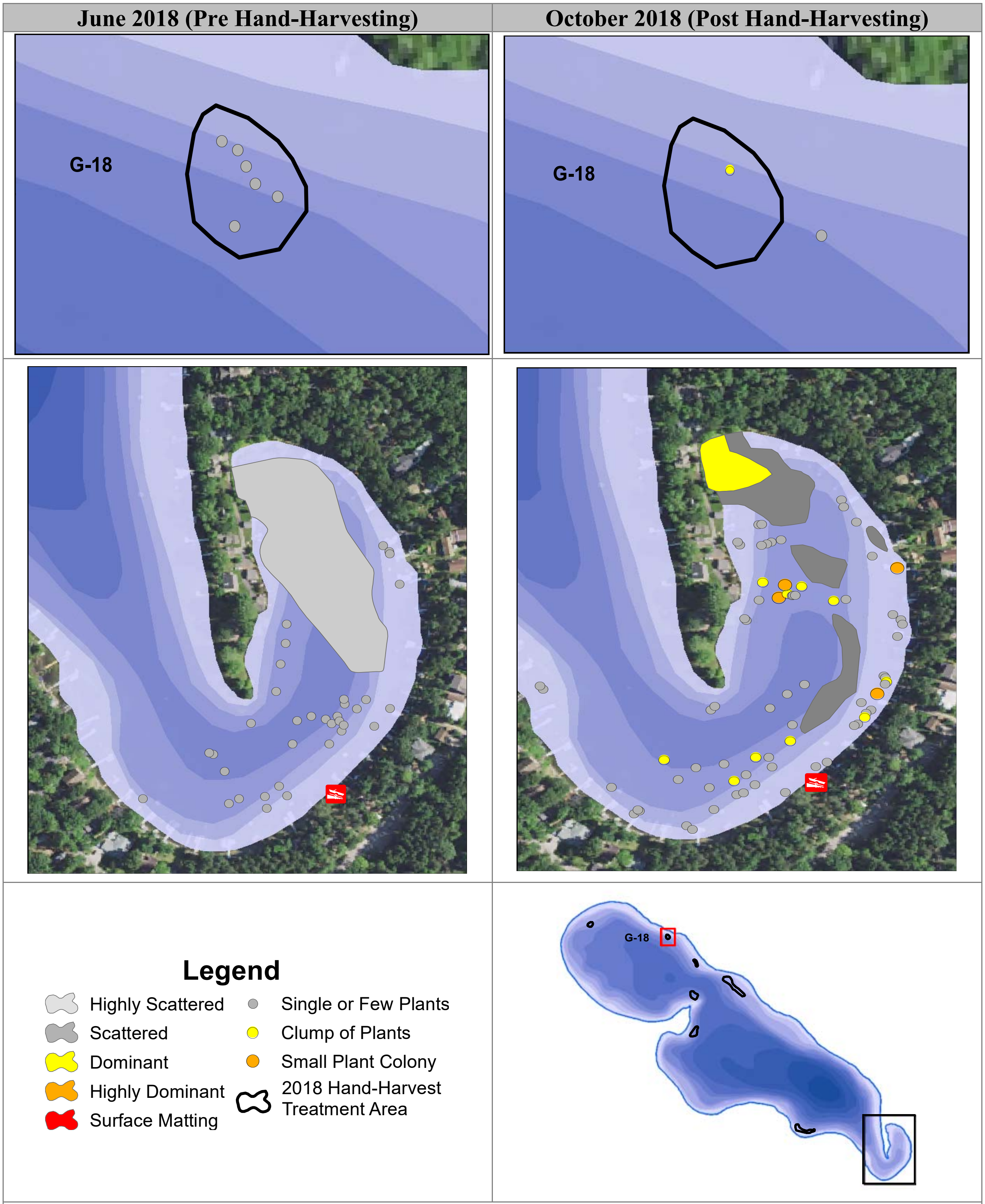


Figure 2.3-3. HWM Populations from before (June 2018) and after (October 2018) Professional Hand-Harvesting Efforts at sites G-18, & Fox Tail Bay in Silver Lake.

Site Fox Tail Bay: The 2018 ESAIS in Fox Tail Bay indicated a large, colonized area of *highly scattered* HWM plants. *Single or few plant* points were also mapped in the ESAIS survey. A total of 22 cubic feet of HWM was removed through 8-hours of hand-harvesting efforts. As discussed, the hand-harvesting efforts in Fox Tail Bay were abandoned due to dense aquatic plants and concerns for diver safety. The HWM peak-biomass survey showed a colonized HWM population of greater densities remained present within the bay. A total of 1.8 acres of *dominant* and *scattered* HWM colonies were mapped in Fox Tail bay. Numerous *small plant colonies*, *clumps*, and *single plants* were also mapped in the area (Figure 2.3-3).

2.4 2018 Point-Intercept Survey Results

One occurrence of HWM was recorded on the August 2017 (*1-year after treatment*) point-intercept survey in Silver Lake, representing a littoral frequency of occurrence of 0.2% The littoral frequency of occurrence of HWM exhibited a 99% decrease since the 2015 survey in which an occurrence of 20.0% was recorded (Figure 2.4-1). The 99% decrease in occurrence from 2015 (*year prior to treatment*) to 2017 (1-*year after treatment*) met lake managers expectation for successful control. The whole-lake point-intercept survey was replicated in 2018 to allow for further understanding of the native and non-native plant populations two-years after treatment. The 2018 survey found the HWM population had increased to 1.7%, representing a statistically valid increase in population since the 2017 survey (Figure 2.4-1). The 1.7% littoral frequency of HWM remains lower than any survey completed on Silver Lake prior to 2017 and remains 91.5% lower than the 2015 survey which was prior to the large-scale fluridone treatment.

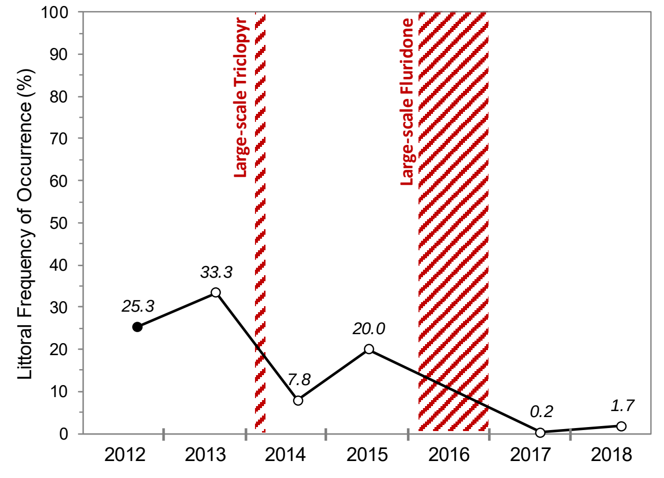


Figure 2.4-1. Littoral frequency of occurrence of HWM in Silver Lake. Open circle represents statistically valid change from previous survey (Chi-Square α = 0.05).

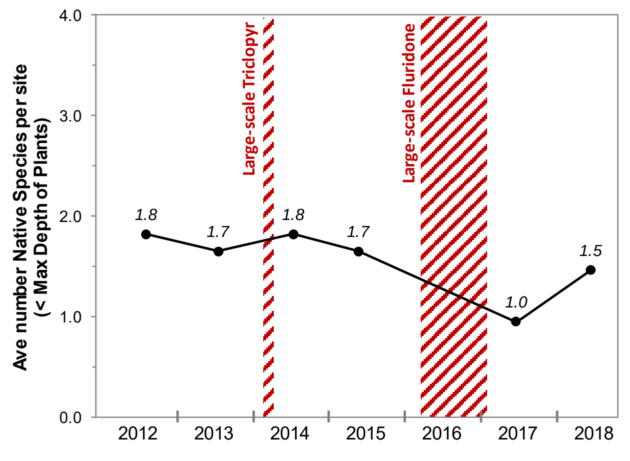


Figure 2.4-2. Average number of native species per sampling site from point-intercept surveys conducted from 2012-2018 on Silver Lake.

Along with understanding the level of HWM control achieved from the control action, the point-intercept data will also allow an understanding of non-target native plant impacts from the treatment. Figure 2.4-2 displays the average number of native aquatic plant species at each sampling site from point-intercept surveys conducted between 2012 – 2018. The average number of native species varied between 1.7 and 1.8 from 2012-2015 before declining to 1.0 in 2017. In 2018, the average number of native species per sampling site increased to 1.5, which is slightly less than the values from 2012-2015 (Figure 2.4-2).

Figure 2.4-3 shows a semi-quantitative analysis of the abundance of natives through looking at total rake fullness ratings (i.e. how full of plants is the sampling rake at each location). The TRF data collected during 2017 shows an overall reduction in rake fullness as compared to previous surveys. In 2017, 54% of the sampling points contained aquatic vegetation comparted to 80% in 2015. It is important to note that the aquatic plant fullness in 2017 is almost completely comprised of native plant species, whereas HWM was a large contributor to the aquatic plant biomass in past surveys (Figure 2.4-4). The 2018 TRF data indicate an increase in rake fullness since 2017 with 64% of the littoral sampling points containing vegetation.

Figure 2.4-4 displays the number of point-intercept sampling locations that contained either native plants, HWM or both native plants and HWM. These data demonstrate the reduction in sampling points that contained HWM and native plants in 2017 following the large-scale herbicide treatment as well as an increase in sampling points with native plants from 2017 to 2018.

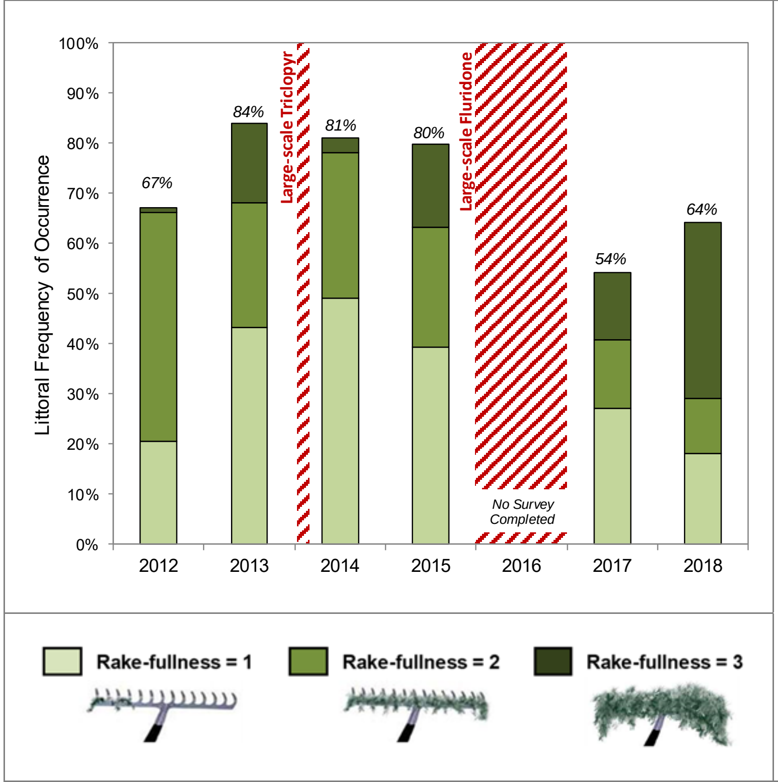


Figure 2.4-3. Silver Lake total rake fullness ratings from 2012 – 2018 point-intercept surveys. No survey was conducted in 2016.

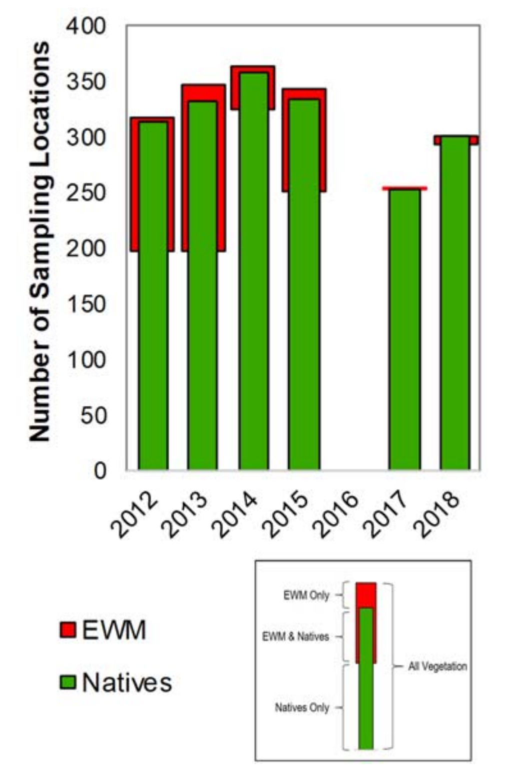


Figure 2.4-4. Number of sampling locations that contained either native plants or HWM from 2012-2018 point-intercept surveys.

Eight species exhibited a statistically valid decrease in occurrence from the 2015-2017 point-intercept surveys. The frequency of occurrence of common waterweed (*Elodea canadensis*, -100%) and coontail (*Ceratophyllum demersum*, -44.3%) represents a statistically valid decrease from 2015 to 2017 (Figure 2.4-5). According to a fluridone susceptibility analysis completed by the WDNR Science Service Department, common waterweed and coontail were shown to be particularly sensitive to fluridone treatments. Both coontail and common waterweed are free-floating or loosely rooted plant species that can utilize the biomass of other plant species as a "substrate" in which they become entangled and grow. It is suspected that with the nearly complete loss of structural habitat previously being supplied by the robust HWM population, may have compounded the direct impacts from the herbicide treatment strategy. A relatively robust population of coontail remained present in the lake during the 2017 point-intercept survey in which coontail was the second most common species with a frequency of 20.2%. Continued monitoring of these species in 2018 found the littoral frequency of occurrence of coontail increased to 25.5% and the species was once again the second most common species during the 2018 point-intercept survey (Figure 2.4-5). The population of common waterweed has been slow to recover after the fluridone treatment with the 2018 survey indicating a 1.3% littoral frequency of occurrence which is 93.2% lower than the 2015 occurrence of 19.1%.

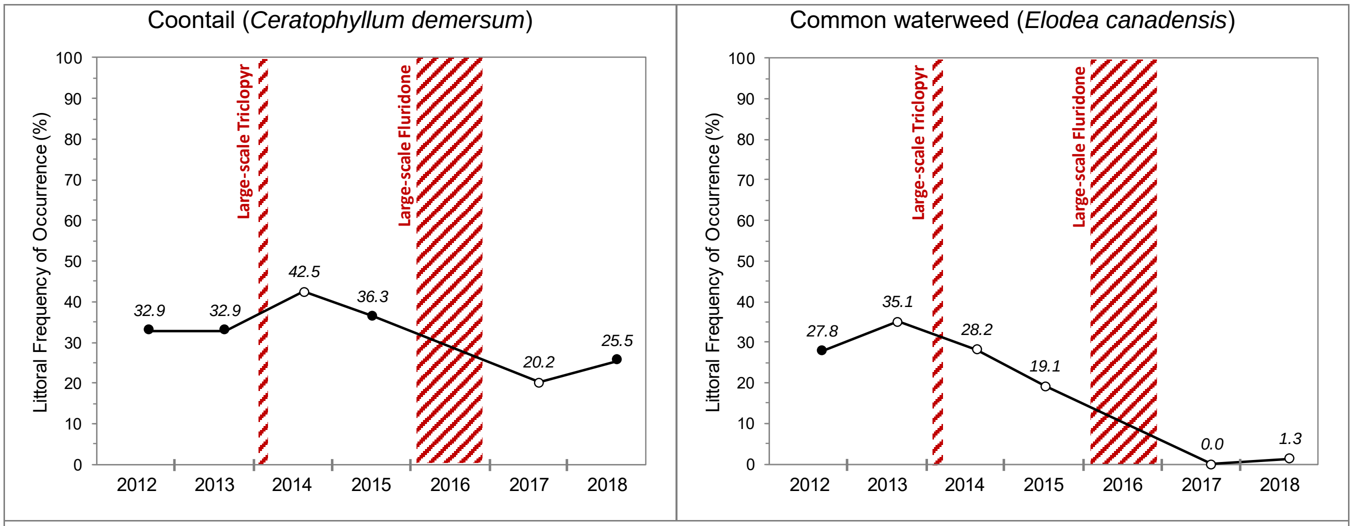


Figure 2.4-5. Littoral frequency of occurrence of coontail (Ceratophyllum demersum) and common waterweed (Elodea canadensis) from 2012-2018 in Silver Lake. Data from 2012-2018 point-intercept surveys. Open circle represents statistically valid change from previous survey (Chi-Square α = 0.05). 2013 large-scale triclopyr treatment and 2016 large-scale fluridone treatment indicated by red dashed lines.

The frequency of occurrence of southern naiad *(Najas guadalupensis)* decreased by a statistically valid 77.6 % in 2017, and slender naiad *(Najas flexilis)* decreased by a statistically valid 100% (Figure 2.4-6). Southern naiad is a hardy perennial that can be a nuisance at times and is suspected of expanding in population following auxin treatments (e.g. 2,4-D, triclopyr), whereas slender naiad is an annual that relies on seed production and has been shown to be particularly susceptible to auxin herbicides (Photo 2.4-1). However, slender naiad has shown to rebound quickly in most large-scale auxin treatments, often exceeding pretreatment levels during the *year after treatment*.

The 2018 point-intercept survey indicated that both southern naiad and slender naiad exhibited a statistically valid increase in littoral frequency of occurrence compared to the 2017 survey (Figure 2.4-6). Slender naiad populations in 2018 were approximately the same as the 2015 survey prior to the large-scale treatment. The littoral frequency of southern naiad remains lower than the 2015 survey, however has shown signs that the population is rebounding in 2018.

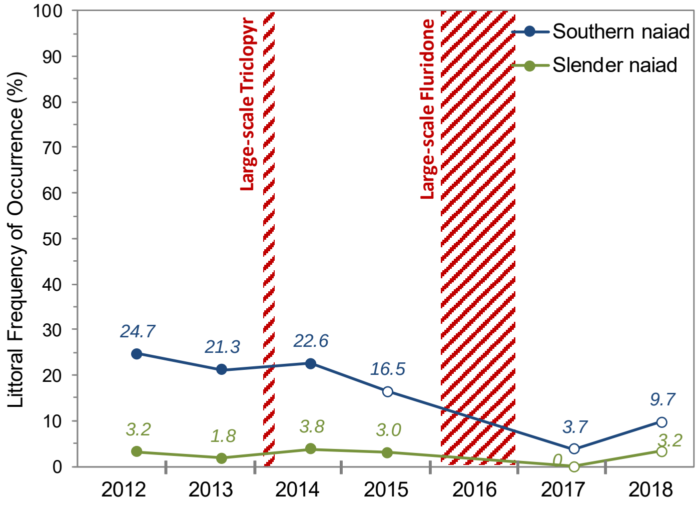


Figure 2.4-6. Littoral frequency of occurrence of slender naiad and southern naiad. Open circle represents statistically valid change from previous survey.

Photo 2.4-1. Slender naiad (Najas flexilis; left) and southern naiad (N. guadalupensis; right). Photo credit Onterra.

Other species that exhibited a statistically valid decrease in occurrence from 2015 to 2017 include: variable-leaf pondweed (*Potamogeton gramineus*), Illinois pondweed (*Potamogeton illinoensis*), Fries' pondweed (*Potamogeton friesii*), and white-stem pondweed (*Potamogeton praelongus*) (Figure 2.4-7). Variable leaf pondweed and white-stem pondweed exhibited statistically valid increases in littoral frequency of occurrence from the 2017 survey to the 2018 survey and are near or above pre-treatment levels. Variable-leaf pondweed was the third most encountered species during the 2018 point-intercept survey in Silver Lake with a littoral frequency of occurrence of 15.1%. Fries' pondweed and Illinois pondweed exhibited slightly lower littoral frequencies in 2018 compared to 2017, however they were not statistically different between the two surveys. Continued monitoring in the coming years will aid in understanding the population dynamics of these aquatic plant species following the large scale fluridone treatment.

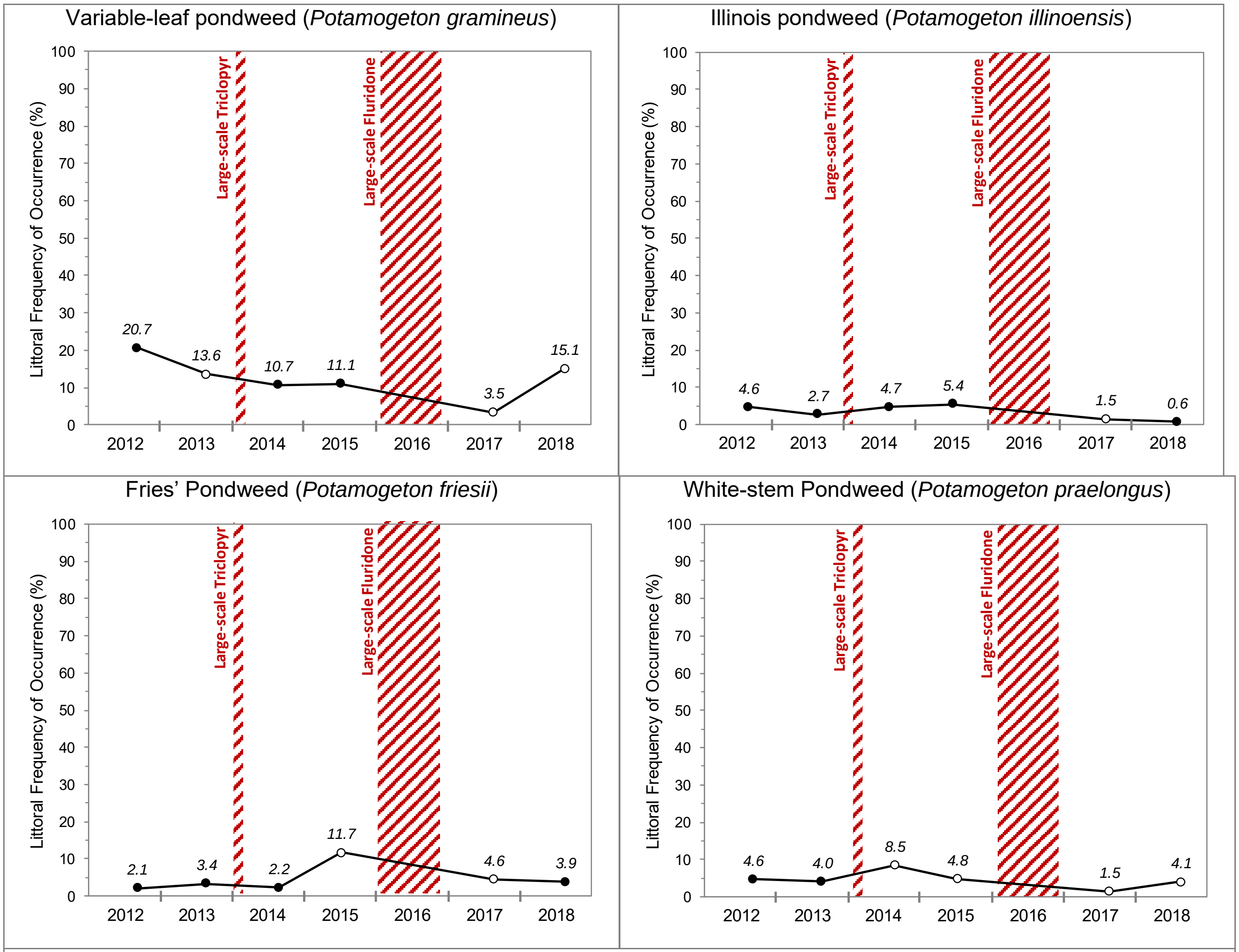


Figure 2.4-7. Littoral frequency of occurrence from of native aquatic plant species that exhibited a statically valid decrease in occurrence following the large-scale fluridone treatment in Silver Lake. Data from 2012-2018 point-intercept surveys. Open circle represents statistically valid change from previous survey (Chi-Square α = 0.05). 2013 large-scale triclopyr treatment and 2016 large-scale fluridone treatment indicated by red dashed lines.

Muskgrasses and stoneworts are genera of macroalgae (Photo 2.4-2). These macroalgae require lakes with good water clarity, and their large beds stabilize bottom sediments. Studies have also shown that muskgrasses sequester phosphorus in the calcium carbonate incrustations which form on these plants, aiding in improving water quality by making the phosphorus unavailable to phytoplankton (Coops 2002). As macroalgae, they are typically resilient to most herbicide use-patterns. Due to their morphological similarity muskgrasses (*Chara* spp.) and stoneworts (*Nitella* spp.) were combined for the analysis, but also shown separately (Figure 2.4-8). The populations of muskgrasses and stoneworts have remained relatively stable between 2012 and 2018. In the 2018 point-intercept survey, the combined occurrences of muskgrasses and stoneworts had the highest littoral frequency of any species in Silver Lake at 33.5% (Figure 2.4-8).

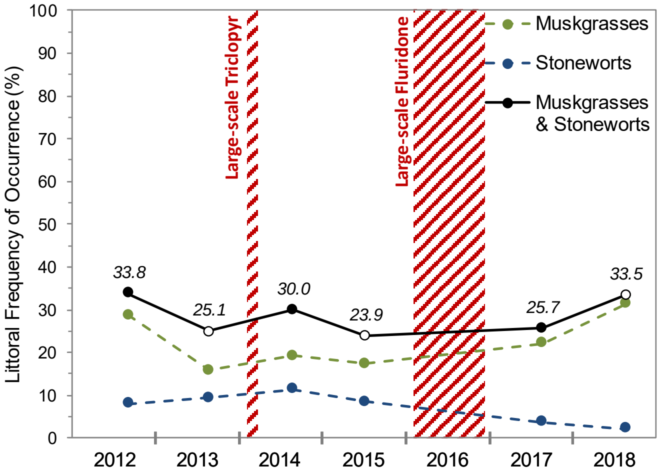


Figure 2.4-8. Littoral frequency of occurrence of muskgrasses (Chara spp.) and stoneworts (Nitella spp.) from 2012-2018 in Silver Lake. Blue dashed line indicates initiation of large-scale triclopyr treatment, red dashed line indicates initiation of large-scale fluridone treatment

Photo 2.4-2. The aquatic macroalgae muskgrasses (Chara spp.). Photo credit Onterra.

Small pondweed (*Potamogeton pusillus*) exhibited a statistically valid increase in occurrence from 2017-2018 (Figure 2.4-9). Wild celery (*Vallisneria americana*) also exhibited a statistically valid increase in littoral frequency of occurrence from 2017-2018 and was the fourth most encountered species during the 2018 survey. Five other native species that have been commonly found in point-intercept surveys in Silver Lake are displayed on Figure 2.4-10. A chi-square analysis for all species identified in the point-intercept surveys is included as an appendix to this report (Appendix B).

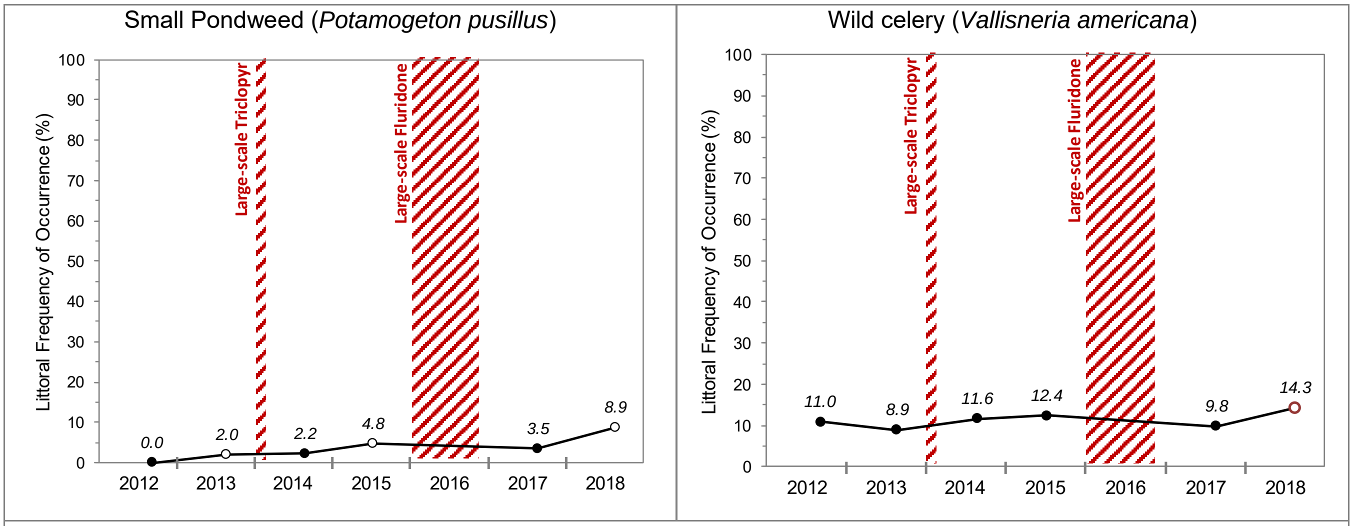
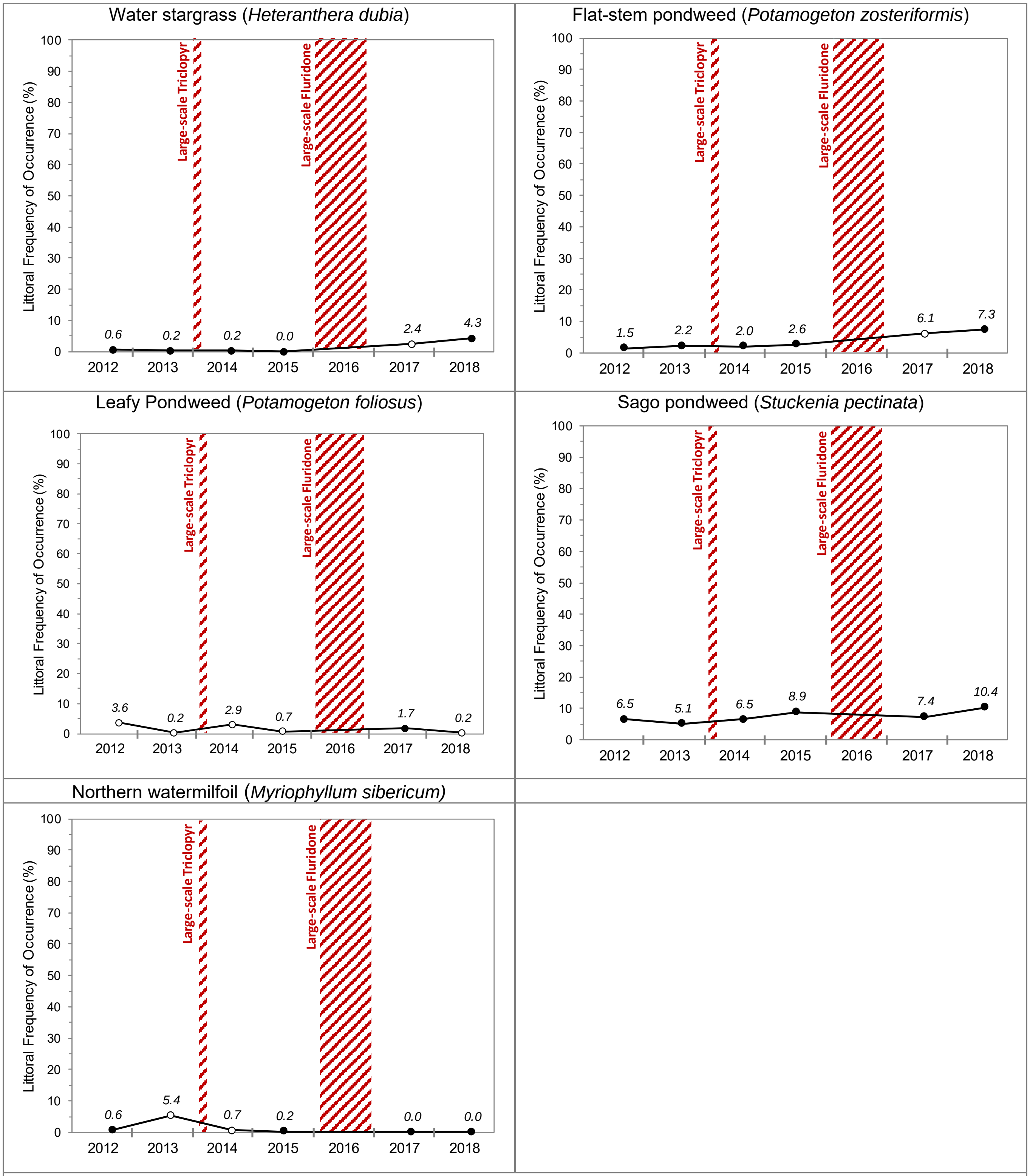


Figure 2.4-9. Littoral frequency of occurrence of small pondweed and wild celery from 2012-2018 in Silver Lake. Data from 2012-2018 point-intercept surveys. Open circle represents statistically valid change from previous survey (Chi-Square α = 0.05). 2013 large-scale triclopyr treatment and 2016 large-scale fluridone treatment indicated by red dashed lines.



Water stargrass (*Heteranthera dubia*)

Flat-stem pondweed (*Potamogeton zosteriformis*)

Leafy Pondweed (*Potamogeton foliosus*)

Sago pondweed (*Stuckenia pectinata*)

Northern watermilfoil (*Myriophyllum sibericum)*

Figure 2.4-10. Littoral frequency of occurrence of select native aquatic plant species from 2012-2018 in Silver Lake. Open circle represents statistically valid change from previous survey (Chi-Square α = 0.05). 2013 large-scale triclopyr treatment and 2016 large-scale fluridone treatment indicated by red dashed lines.

2.5 Acoustic Survey Results

In the summer of 2015 prior to treatment and the summer of 2017 following the treatment, Onterra ecologists conducted acoustic surveys with two primary goals: 1) to obtain accurate bathymetric data for the proposed 2016 treatment to ensure accurate herbicide dosing, and 2) to document the change in aquatic plant bio-volume from before and after the treatment. An additional benefit of the acoustic survey allowed quantitative comparisons of water levels and volumes between 2015 and 2017. In 2018, a partial acoustic survey was completed during August with the purpose of documenting the aquatic plant biomass in Silver Lake. The extents of the surveyed area during the 2018 acoustic survey included only the vegetated areas of the lake and these data are comparable to the data collected in 2015 and 2017.

The acoustic data confirms the water volume in 2017 was much higher than in 2015 as a result of the record precipitation. Volume calculations utilizing the data obtained from the 2017 survey indicated the water volume to be approximately 8,791 acre-feet, which is approximately 17% greater (1,275 acre-feet more) than the volume calculated from the 2015 acoustic survey. This correlates to about a 3.5-foot increase in water levels, which was approximately confirmed by difference in the maximum depth located in the 2015 (50 feet) and 2017 (53 feet) acoustic surveys. The 2018 acoustic survey did not evaluate water volumes.

The increased water depth of approximately 3 to 3.5 feet would have impacts to the littoral area of the lake (zone in which light is able to penetrate the water column allowing for plant growth). With a shifting littoral zone, plants that had been established at depths within the littoral zone most conducive to its growth may have been unable to adapt to the increased water level and decreased light availability. This factor may have compounded the impacts from the herbicide treatment in 2016 and may have contributed to the reduction in overall aquatic plant growth observed in 2017. This factor may also be making it more difficult for HWM populations to rebound in deeper areas. Water levels in Silver Lake were observed to be similarly high during much of the 2018 growing season.

Aquatic plant bio-volume is measured as a percentage of the water column that is occupied by aquatic plants. The results of these surveys are displayed in Figure 2.5-1. Prior to treatment, approximately 15.2% of the lake area contained a bio-volume 50% or greater, compared to approximately 3.1% for the same parameters in 2017 (Figure 2.5-1). These high biovolume areas in 2015 corresponded with the dense colonies of HWM that were greatly impacted by the fluridone treatment. In 2018, the data showed approximately 13% of the lake area contained a bio-volume of 50% or greater.

The 2015 acoustic survey showed approximately 60% of the lake area contained between a 0-10% bio-volume compared to 77.9% in 2017 that had a bio-volume between 0 and 10% (Figure 2.5-1). These data indicate that the lake has more areas with no/low vegetation in 2017 than in 2015. A portion of this increase is caused by a reduction in deeper areas that contained vegetation in 2015 that were unable to support vegetation in 2017 now that the lake-levels were increased by 3-3.5 feet. For example, deeper areas of the northwestern lobe of the lake contained areas with greater than 10% biovolume in 2015 but not in 2017. The 2018 acoustic survey showed that the area of the lake with between 0-10% biovolume was 61.2% and similar to the 2015 survey.

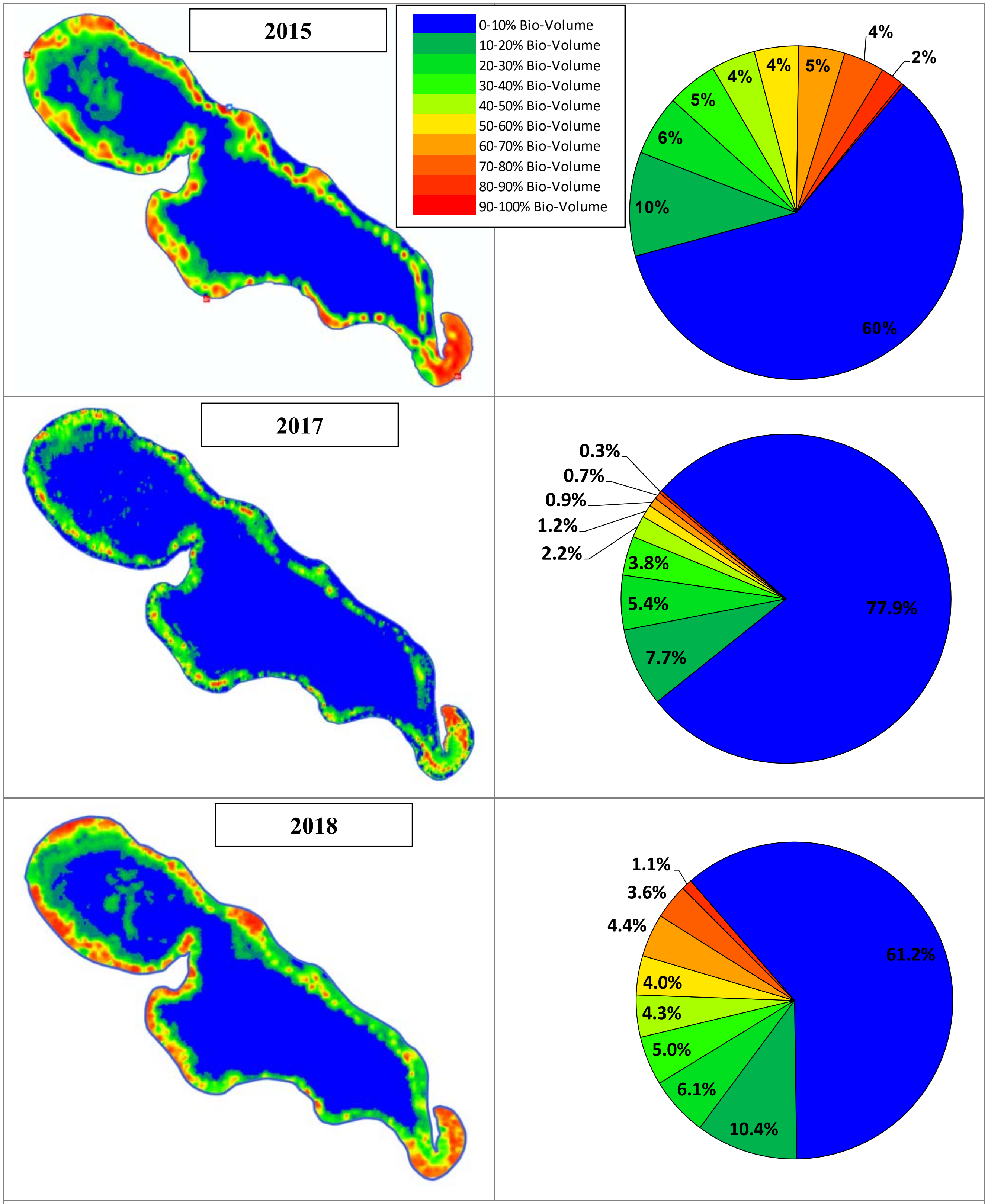


Figure 2.5-1. Bio-volume of aquatic vegetation within Silver Lake from bioacoustics surveys conducted in 2015, 2017, & 2018.

3.0 CONCLUSIONS AND DISCUSSION

Professional hand-harvesting efforts in 2018 were able to effectively maintain or reduce the HWM population in all of the sites in which efforts were undertaken with the exception of Fox Tail Bay. This method of HWM management continues to be evaluated as it relates to meeting HWM management expectations in Silver Lake, as early outcomes are promising.

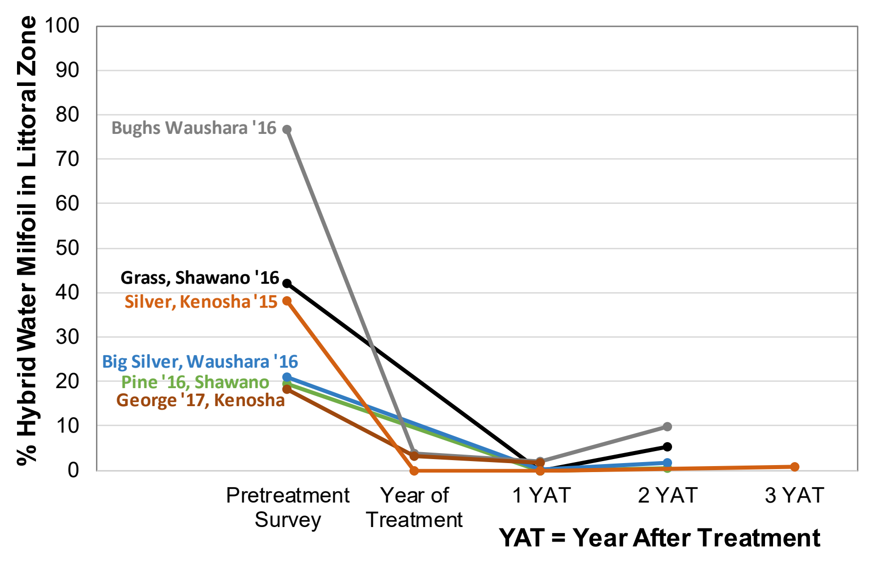


Figure 3.0-1. Littoral frequency of occurrence of HWM in lakes managed with whole-lake pelletized fluridone treatments.

While the hand-harvesting efforts were effective at managing the HWM populations in areas where is was applied, some HWM population increases were observed in areas of the lake that were not targeted. The overall HWM population continues to be much lower level than was observed prior to the large-scale fluridone treatment. Figure 3.0-1 shows the level of HWM control from six Wisconsin pelletized fluridone treatments, including for Silver Lake (Big Silver) Please note that a point-intercept survey was not completed during the year of treatment on some lakes (Big Silver, Pine, and Grass), as the lakes were still in the process of being treated (i.e. had active herbicide concentrations). During the year after treatment (YAT), all lakes contained HWM populations below 2% of the littoral zone. HWM rebound has been the greatest on Bughs Lake, with all other lakes containing approximately 5% or less HWM at 2 YAT. Please note that Bughs Lake has a past history of fluridone treatment, whereas the others have not. Silver Lake in Kenosha County is the only lake that has progressed to 3 YAT, with 0.8% of the littoral zone containing HWM.

As a part of an ongoing Integrated Pest Management (IPM) strategy following the 2016 large-scale treatment, the SLMD intends to continue to implement follow-up management actions that include hand-harvesting and/or herbicide spot treatments in an effort to maintain the HWM population at lower levels in Silver Lake. Following this management strategy, professional hand-harvesting as well as a spot herbicide treatment are discussed below. The SLMD will be submitting an application for a WDNR AIS-Established Population Control Grant to assist with funding the IPM strategy.

3.1 Herbicide Spot-Treatment

One area of Silver Lake was originally considered for herbicide control of HWM in 2019. Fox Tail Bay, on the southeast end of the lake has contained some dense populations of HWM in recent years and was shown to have *scattered* and *dominant* density HWM communities during the late-summer 2018 survey. Hand-harvesting in this area of Silver Lake in 2018 proved to be challenging as dense native aquatic plants hindered the professional diver's removal efforts and resulted in falling short of meeting control expectations. The protected nature of this bay of the lake is believed to aid in limiting herbicide dissipation out of the application area and is theorized to allow for sufficient concentration exposure times to result in HWM control. However, this area has a history of various herbicide treatments with mixed results. Because of the stage of recovery/rebound of the HWM population, the SLMD considered a potentially more aggressive management approach to this population. This included evaluation of several herbicide herbicides that require short exposure times (diquat, florpyrauxifen-benzyl [ProcellaCOR™]) and herbicide combinations (diquat/endothall, 2,4-D/endothall, etc.).

At the time of this report, the Lake District is trending towards florpyrauxifen-benzyl, commercially available as ProcellaCOR™ (SePRO). This herbicide is specifically designed to control invasive milfoil populations. ProcellaCOR™ is in a new class of synthetic auxin mimic herbicides (arylpicolinates) with short concentration and exposure time (CET) requirements compared to other systemic herbicides. Uptake rates of ProcellaCOR™ into EWM were two times greater than reported for triclopyr (Haug 2018, Vassios et al. 2017). ProcellaCOR™ is primarily degraded by photolysis (light exposure), with some microbial degradation. The herbicide is relatively short-lived in the environment, with half-lives of 4-6 days in aerobic environments and 2 days in anerobic environments (WSDE 2017). The product has a high affinity for binding to organic materials (i.e. high KOC).

Netherland and Richardson (2016) and Richardson et al. (2016) indicated control of select non-native plant species with the active ingredient in ProcellaCOR™, including invasive watermilfoils (EWM and HWM) at low application rates compared with other registered spot treatment herbicides. The majority of native plants tested to date also suggest greater tolerance to this mode of action. Water lilies, pickerelweed, arrowheads, and native watermilfoils have shown sensitivity to ProcellaCOR™. Coontail may also be impacted at higher application rates. Because this is a new herbicide, data available from field trials is relatively limited.

The use of any aquatic herbicide poses environmental risks to non-target plants and aquatic organisms. The majority of available toxicity data has been conducted as part of the EPA product registration process. These laboratory studies are attempted to mimic field settings, but can underestimate or overestimate the actual risk (Fairbrother and Kapuska 1996). Federal and state pesticide regulations and strict application guidelines are in place to minimize impacts to non-target organisms based on the organismal studies. The use of aquatic herbicides includes regulatory oversight and must comply with the following list:

- Labeled and registered with U.S. EPA's office of Pesticide Programs;
- Registered for sale and use by the Department of Agriculture, Trade, and Consumer Protection (DATCP);
- Permitted by the Wisconsin Department of Natural Resources (WDNR); and
- Applied by a DATCP-certified and licensed applicator,

The EPA Ecological Risk Assessment places the risk to non-target wildlife into the "no risk concern" category and the impacts to bees, birds, reptiles, amphibians, and mammals in the "practically non-toxic" category. The EPA has also indicated that there are no risks of concern to human health. There are no restrictions on swimming, drinking, fish consumption, or turf irrigation. However, there would be an approximate 1-day waiting period of the proposed application for shoreland irrigation due to concerns of herbicidal impacts.

Additional information from the WDNR related to aquatic herbicide regulation and the WDNR's Chemical fact sheet for florpyrauxifen-benzyl are included in Appendix C. Appendix C also includes the relevant chapter on ProcellaCOR™ from the State of Washington Department of Ecology *Final Supplemental Environmental Impact Statement for State of Washington Aquatic Plant and Algae Management* (August 14, 2017).

A preliminary strategy of ProcellaCOR™ treatment in Fox Tail Bay for 2019 was presented to local WDNR biologists (Ted Johnson) in mid-January targeting 7.7 acres (Map 3). This site is divided into two areas that contain differing average depths. For this site, SePRO recommends 3 Prescription Dose Units (PDU) of ProcellaCOR™ EC per acre-foot, which equates to 5.8 ppb. The maximum application rate of this formulation of ProcellaCOR™ is 25 PDU with a typical use rate for invasive watermilfoils being 3-5 PDU. Because the product has a high affinity for binding to organic materials, SePRO recommends closely spaced application transects for this treatment as well as an early application timing.

If grant funds are being used or new-to-the-region herbicide strategies are being considered, the WDNR may request a quantitative evaluation monitoring plan be constructed that is consistent with Appendix D of the WDNR Guidance Document, *Aquatic Plant Management in Wisconsin* (WDNR 2010). This generally consist of collecting quantitative point-intercept sub-sampling during the summer before the treatment (pre) and summer following the treatment (post). For perspective treatment in 2019, the pretreatment data would need to be collected during the summer of 2018. Because the whole-lake point-intercept survey was conducted on Silver Lake in 2018, investigating a sub-set of these data was originally considered for the quantitative monitoring strategy (Figure 3.1-1, left frame).

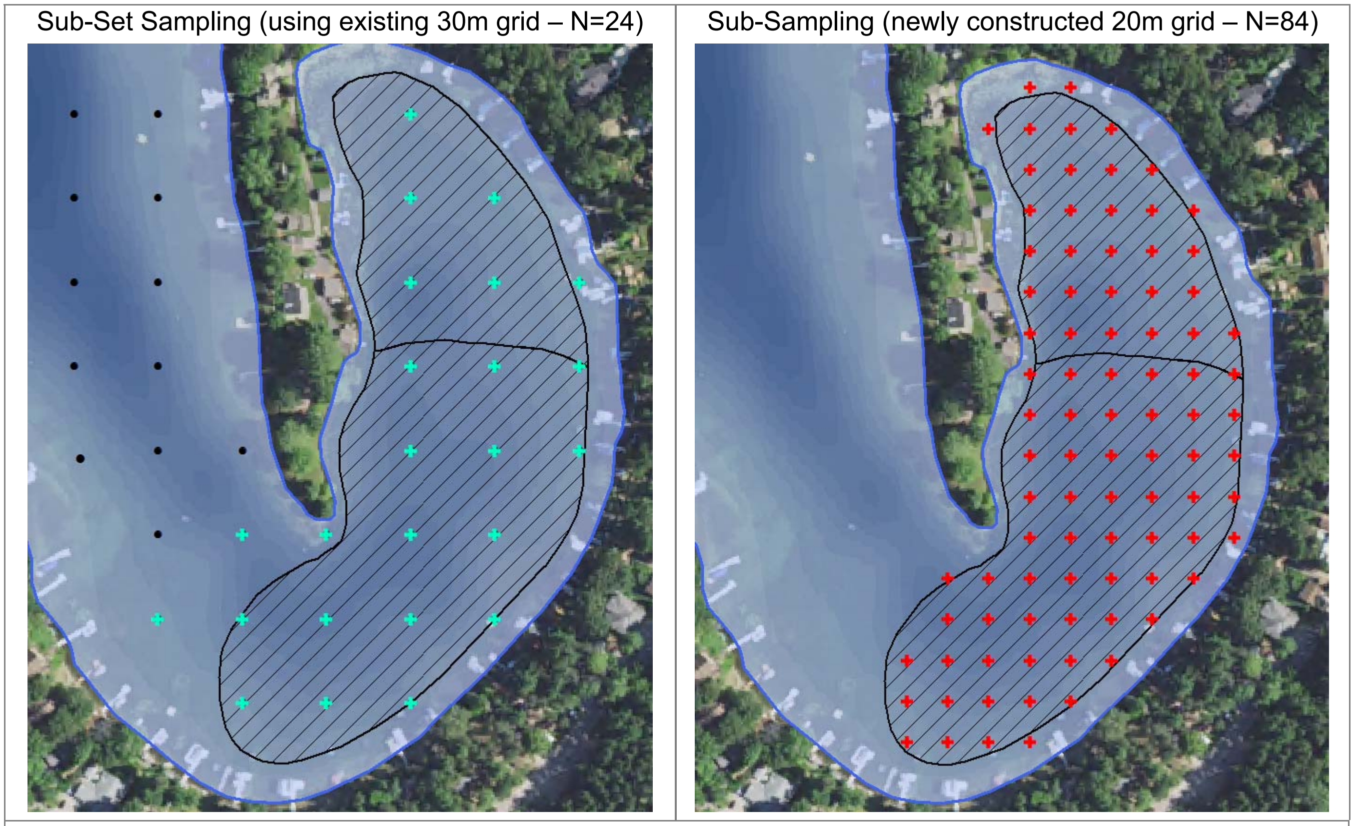


Figure 3.1-1. Quantitative sampling plans. Left frame shows sub-set of whole-lake point-intercept (green cross) and right frame shows newly developed sub-set sampling plan (red cross).

Mr. Johnson raised concerns that this was an insufficient amount of sampling intensity in this area to understand the impacts of this experimental strategy. Ultimately Mr. Johnson requested the herbicide treatment strategy be postponed until the spring of 2020 for concerns of limited quantitative pretreatment data, low abundance of target plants in that area, and a newer herbicide requiring additional WDNR technical review. The SLMD understands these concerns and will initiate a quantitative monitoring at an increased intensity during the summer of 2019 to serve as a pretreatment dataset for herbicide treatment during spring of 2020 (Figure 3.1-1, right frame). As was proposed for 2019, herbicide concentration monitoring would accompany the 2020 treatment, with a preliminary design investigating 6 intervals (1, 3, 6, 12, 24, and 48 hours after treatment) at 4 locations.

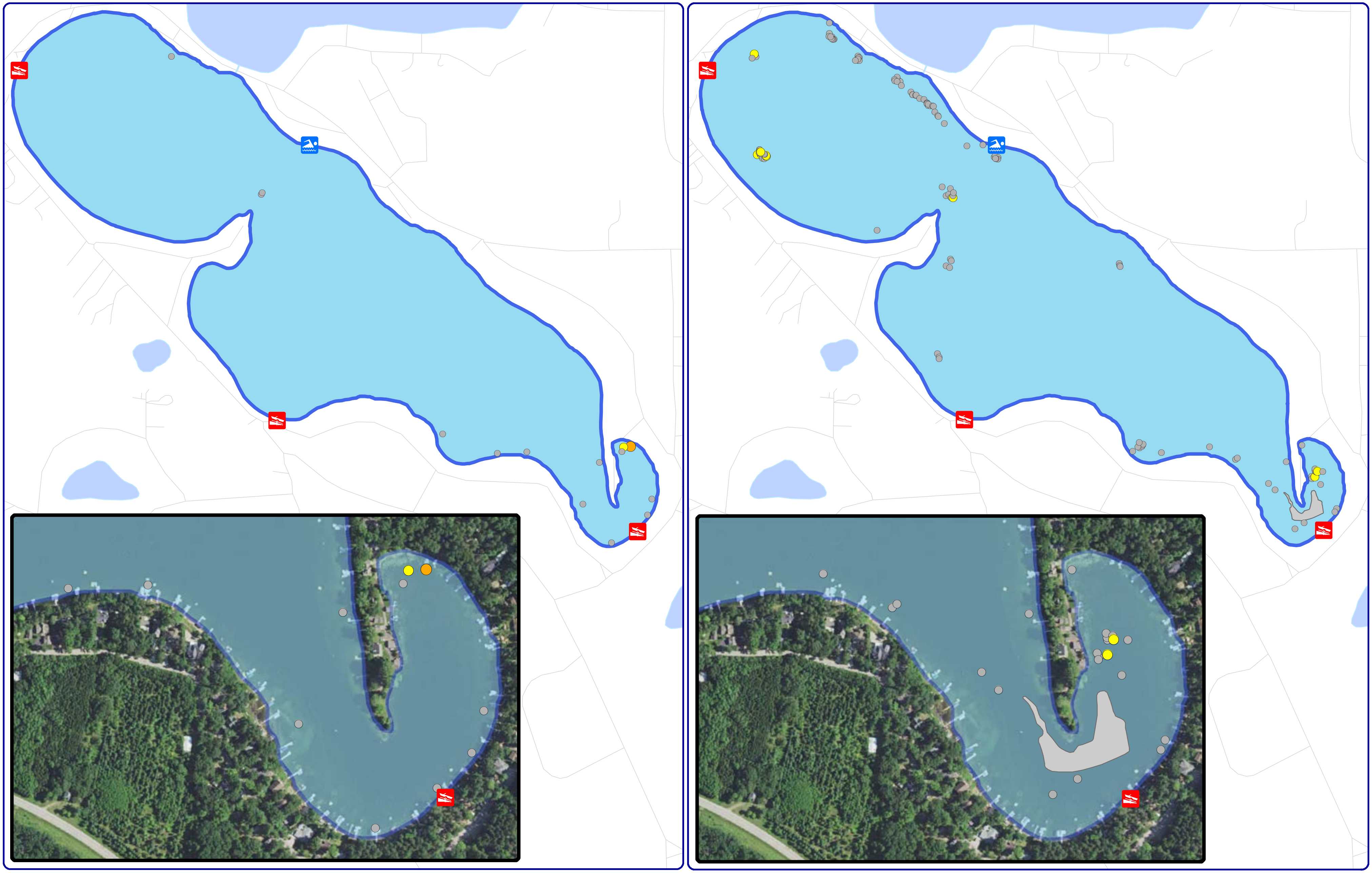
3.2 Hand-Harvesting

The Lake District has contracted with a professional hand-harvesting firm (Aquatic Plant Management, LLC) to implement four weeks of DASH services in 2019 with the possibility of adding a fifth week if necessary. This includes a dedicated week of effort specifically targeting Fox Tail Bay during 2019. Hand-harvesting in Fox Tail Bay during 2018 proved to be challenging as dense native aquatic plants hindered the professional diver's removal efforts, falling short of meeting control expectations. The District believes that greater HWM population management strides in this bay may be achieved by implementing the strategy earlier in the growing season (early June) when HWM and native plants are at an earlier growth stage and thus DASH operations are scheduled to begin on June 17, 2019.

The complete preliminary DASH strategy for 2019 is offered on Map 4 in which a total of 24.15 acres over 40 work sites in the lake are proposed for HWM management utilizing DASH methods. Similar to 2018, the sites are broken into six primary harvesting areas that surround any HWM occurrences in the main body of the lake in which a small plant colony was located during the late-summer 2018 survey. A seventh primary hand-harvesting area is delineated for Fox Tail Bay.

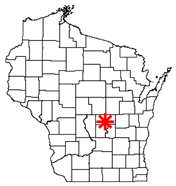
Additionally, 33 dive sites are proposed as second priority DASH sites. These dive sites encompass all other areas of the lake that were found to contain at least a *clump of plants* during the 2018 late-summer survey or they are located at a location in which some of the densest colonies of HWM had been present before the large scale fluridone treatment.

The SLMD is also considering piloting a Residential AIS Removal Project in 2020 in an effort to encourage riparian owners to remove AIS along their frontage through contracted professional hand-harvesting. The basic framework includes the District securing a hand-harvesting contractor and permits if utilizing DASH methodologies. The District is considering a cost share for the project if specific criteria are met, likely based upon water depth or relationship to the pierhead (i.e. end of the dock). Riparians wishing to have AIS removal closer to their shorelines (i.e. shallower water) would be responsible for associated costs but would use the District's contracted service purveyor. Further discussions about this program will be made during upcoming SLMD board meetings.



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, 2015;
processed by C-Map USA
Aquatic Plants: Onterra, 2018
Map Date: January 4, 2019 AMS
File name: *SilverWaushara_CLP_2017-2018.mxd*


Project Location in Wisconsin

Legend

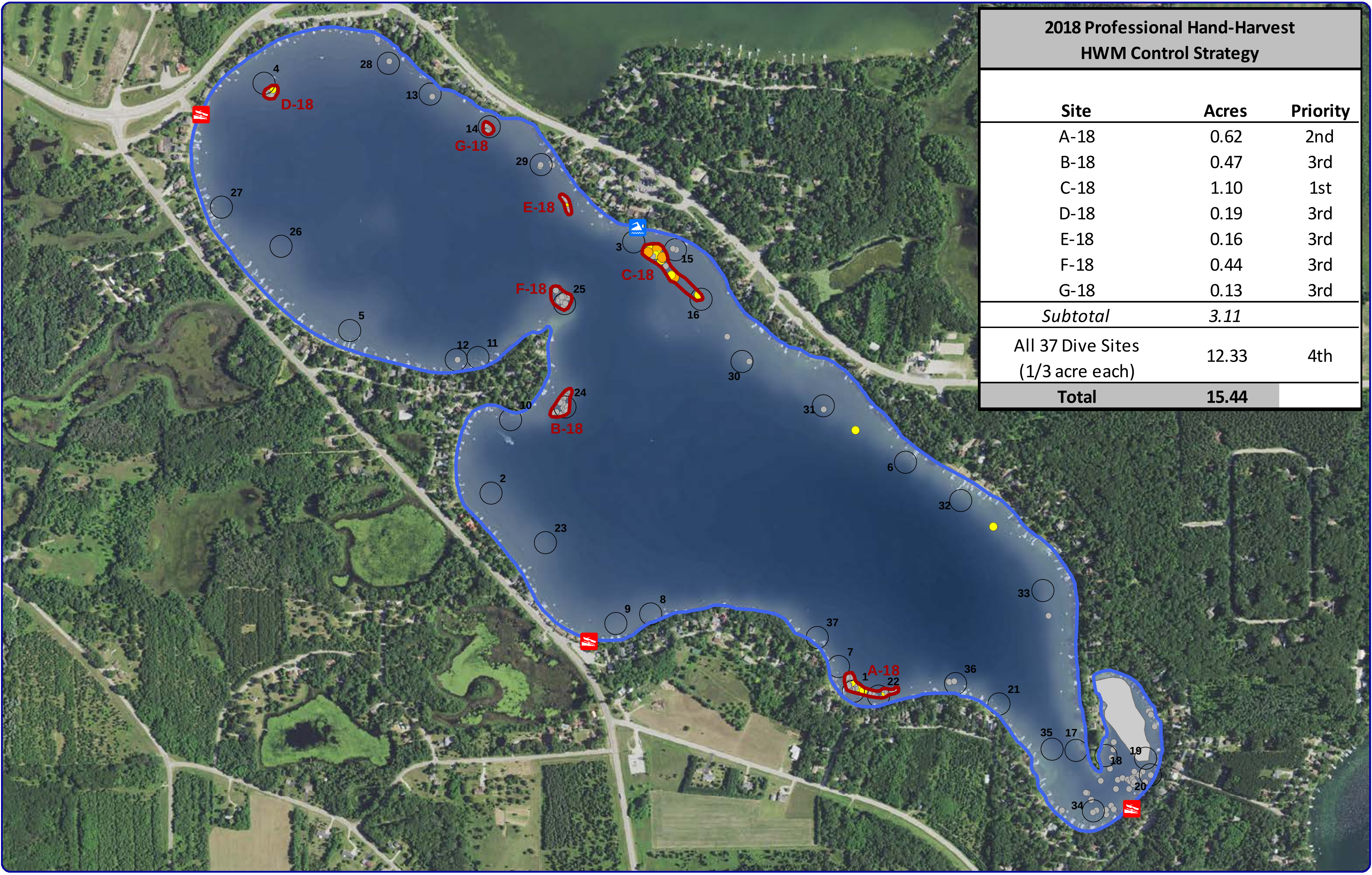
▨	Highly Scattered	●	Single or Few Plants
▨	Scattered *(none)*	●	Clump of Plants
▨	Dominant *(none)*	●	Small Plant Colony *(none)*
▨	Highly Dominant *(none)*		
▨	Surface Matting *(none)*		

Map 1

Silver Lake

Waushara County, Wisconsin

2017-2018 CLP Survey Results



2018 Professional Hand-Harvest HWM Control Strategy

Site	Acres	Priority
A-18	0.62	2nd
B-18	0.47	3rd
C-18	1.10	1st
D-18	0.19	3rd
E-18	0.16	3rd
F-18	0.44	3rd
G-18	0.13	3rd
Subtotal	*3.11*	
All 37 Dive Sites (1/3 acre each)	12.33	4th
Total	**15.44**	

N
W E
S

950
Feet

Onterra LLC
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815 Prosper Road
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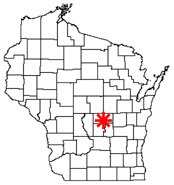
Sources:
Roads and Hydro: WDNR
Bathymetry: Onterra, 2015
Aquatic Plants: Onterra, 2018
Map Date: July 31, 2018 - EJH


Project Location in Wisconsin

Legend

June 2018 HWM/EWM Locations

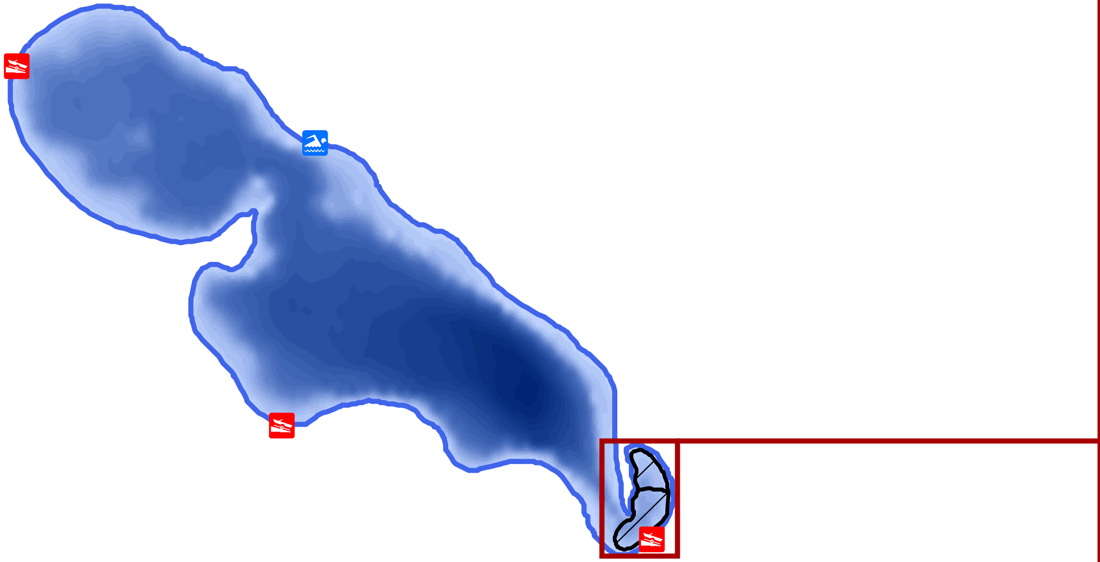
- ▨ Highly Scattered
- ▨ Scattered *(none)*
- ▨ Dominant *(none)*
- ▨ Highly Dominant *(none)*
- ▨ Surface Matting *(none)*
- • Single or Few Plants
- ● Clump of Plants
- ● Small Plant Colony
- ○ Dive Recon & DASH Removal Site
- ○ DASH Removal Site

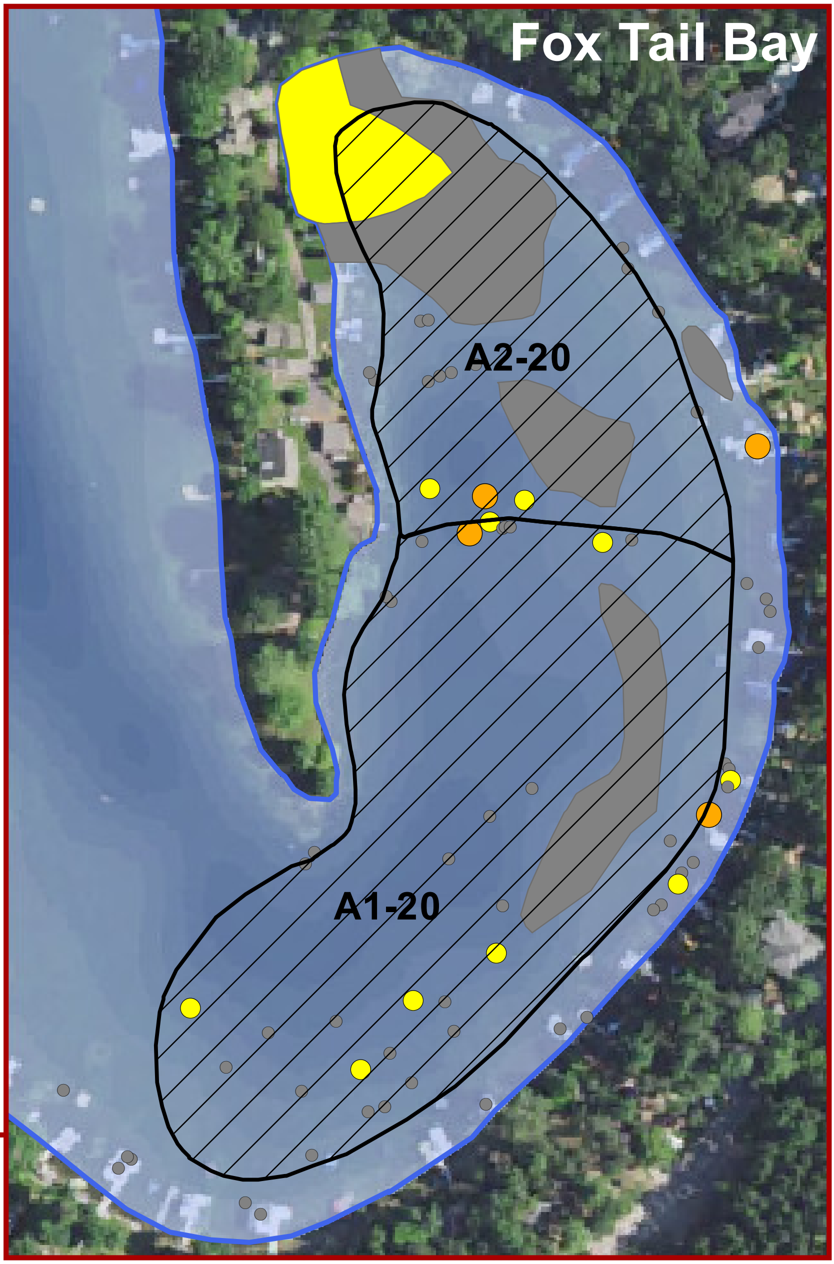
Map 2
Silver Lake
Waushara County, Wisconsin

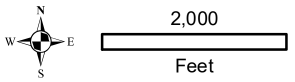
2018 HWM DASH Work Zones v.2

2020 Preliminary Control Strategy
Herbicide Spot Treatment

Site	Proposed Acres	Avg Depth (ft)	Volume (acre/ft)
A-1	5.1	10.8	55.1
A-2	2.6	7.3	19.0
Total	**7.7**		**74.1**







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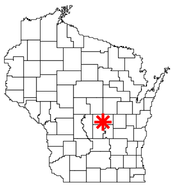
www.onterra-eco.com

Sources:
Roads and Hydro: WDNR
Bathymetry: Onterra, 2015;
Aquatic Plants: Onterra, October 4, 2018
Map Date: October 5, 2018 - EJH



Project Location in Wisconsin

Legend

2018 Late-Season EWM Survey Results

▨ Highly Scattered *(none)*
▨ Scattered
▨ Dominant
▨ Highly Dominant *(none)*
▨ Surface Matting *(none)*

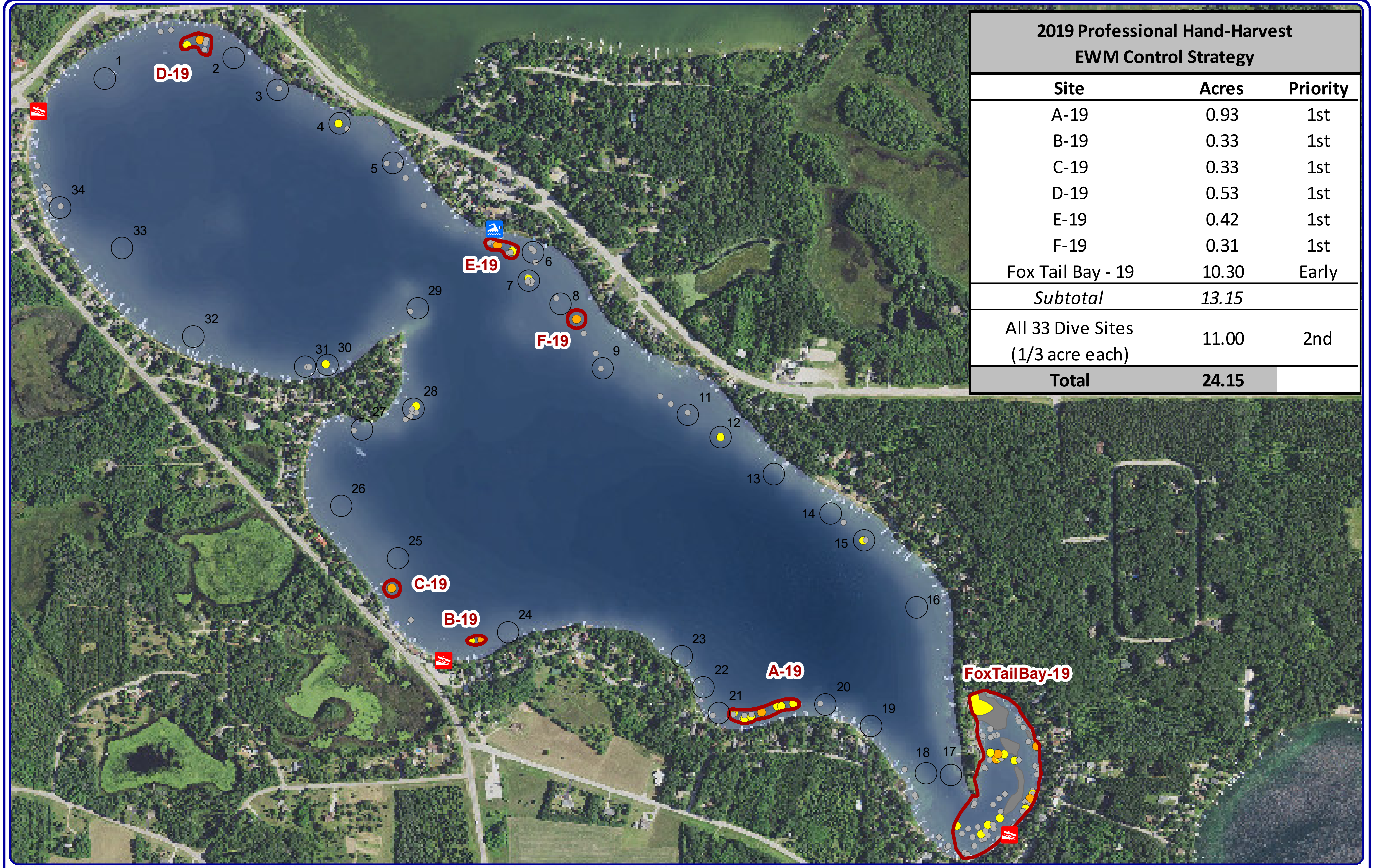
● Single or Few Plants
● Clump of Plants
● Small Plant Colony

Map 3

Silver Lake

Waushara County, Wisconsin

Preliminary 2020 HWM Treatment Strategy



2019 Professional Hand-Harvest EWM Control Strategy

Site	Acres	Priority
A-19	0.93	1st
B-19	0.33	1st
C-19	0.33	1st
D-19	0.53	1st
E-19	0.42	1st
F-19	0.31	1st
Fox Tail Bay - 19	10.30	Early
Subtotal	*13.15*	
All 33 Dive Sites (1/3 acre each)	11.00	2nd
Total	**24.15**	

N
W E
S

1,000
Feet

Onterra LLC
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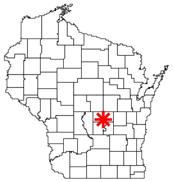
Sources:
Roads and Hydro: WDNR
Bathymetry: Onterra, 2015
Aquatic Plants: Onterra, 2018
Map Date: December 11, 2018 AMS


Project Location in Wisconsin

Legend

October 2018 HWM/EWM Locations

▨	Highly Scattered	•	Single or Few Plants
▨	Scattered	●	Clump of Plants
🟨	Dominant	●	Small Plant Colony
🟧	Highly Dominant *(none)*		
🟥	Surface Matting *(none)*		

○ Dive Recon & DASH Removal Site

2019 Preliminary HH Site

Map 4
Silver Lake
Waushara County, Wisconsin
2019 Preliminary Hand-Harvest Sites v2

A

APPENDIX A

Aquatic Plant Management, LLC 2018 Silver Lake Report on EWM Hand-Harvesting

Silver Lake EWM Treatment Report
2018

PO Box 1134 Minocqua, WI 54548

Silver Lake EWM Treatment Summary 2018

Summary: On August 13th-16th, 20th-24th, and 27th-29th, Aquatic Plant Management LLC (APM) Conducted Diver-Assisted Suction Harvesting (DASH) services of Eurasian Watermilfoil (EWM) on Silver Lake in Waushara County, WI. Utilizing GPS coordinates provided by Onterra LLC, our divers were able to successfully remove 621.3 cubic feet of EWM from the lake. During the week of August 13th, our crew focused on sites C-18 and A-18. During the week of August 20th, our crews focused on sites A-18, B-18, C-18, D-18, E-18, F-18, G-18, and several locations within Foxtail Bay. During the week of August 27th, our crew focused on sites E-18, F-18, Foxtail Bay, and several clumps of plants along the southeast shoreline. Most of the EWM growth in the lake was characterized by clumps of plants surrounded by scattered individual plants.

Conditions:

- 8/13/18: Clear, light winds, 88 degrees.
- 8/14/18: Clear, light winds, 89 degrees.
- 8/15/18: Clear, light winds, 85 degrees.
- 8/16/18: Mostly cloudy, light winds, 80 degrees.
- 8/20/18: Mostly cloudy, mild winds, 75 degrees.
- 8/21/18: Overcast, mild winds, 72 degrees.
- 8/22/18: Partly cloudy, mild winds, 75 degrees.
- 8/23/18: Clear, light winds, 80 degrees.
- 8/24/19: Rain, mild winds, 63 degrees.
- 8/27/18: Mostly cloudy, mild winds, 82 degrees.
- 8/28/18: Overcast, light winds, 80 degrees.
- 8/29/18: Overcast, light winds, 65 degrees.

Recommendations: In several of the dive locations, heavy native aquatic plant growth may be obscuring smaller EWM plants. The abundance of native aquatic plant growth in Foxtail Bay was also a hindrance to the progress of our DASH team in that particular area. It is our recommendation that any future DASH efforts be undertaken earlier in the summer, ideally beginning in June or early July, before the native growth reaches its peak density. While we were able to remove nearly all of the visible EWM from the 2018 control areas that we visited, EWM is present in small amounts in other locations around the lake. Due to this fact, continued monitoring and management efforts are vital to preventing proliferation of EWM throughout Silver Lake.

Map of Silver Lake Dive Sites

● **Dive Site**

Detailed Diving Activities

EWM Treatment Results:

Date	Location	Latitude	Longitude	Time Underwater (Hrs)	Water Depth	Substrate Type	Plant Condition	Native Growth	Estimated EWM Removed (Cubic Feet)
8/13/2018	C-18	44.05777	-89.23354	1.00	10.0	Organic	Intermediate	Abundant	12.0
8/13/2018	C-18	44.05790	-89.23331	1.50	12.0	Organic	Intermediate	Abundant	15.0
8/13/2018	C-18	44.05784	-89.23357	1.50	14.0	Organic	Intermediate	Abundant	15.0
8/13/2018	C-18	44.05779	-89.23331	1.25	12.0	Organic	Intermediate	Abundant	12.0
8/13/2018	C-18	44.05790	-89.23351	1.00	10.0	Organic	Intermediate	Abundant	15.0
8/14/2018	C-18	44.05732	-89.23286	1.58	10.0	Organic	Intermediate	Abundant	7.0
8/14/2018	C-18	44.05732	-89.23286	1.83	10.0	Organic	Intermediate	Abundant	16.0
8/14/2018	C-18	44.05696	-89.23218	0.43	8.0	Organic	Intermediate	Abundant	1.0
8/14/2018	C-18	44.05773	-89.23317	0.75	8.0	Organic	Intermediate	Abundant	6.0
8/14/2018	C-18	44.05788	-89.23330	1.00	12.0	Organic	Intermediate	Abundant	15.0
8/14/2018	C-18	44.05788	-89.23330	1.08	12.0	Organic	Intermediate	Abundant	15.0
8/15/2018	C-18	44.05784	-89.23357	1.67	11.0	Organic	Intermediate	Abundant	4.5
8/15/2018	C-18	44.05777	-89.23336	1.58	11.0	Organic	Intermediate	Abundant	4.5
8/15/2018	A-18	44.04902	-89.22797	1.33	12.0	Organic	Intermediate	Abundant	4.5
8/15/2018	A-18	44.04902	-89.22800	1.08	12.0	Organic	Intermediate	Abundant	18.0
8/16/2018	A-18	44.04908	-89.22793	0.83	12.0	Organic	Intermediate	Abundant	10.0
8/20/2018	A-18	44.04881	-89.22703	1.42	11.0	Organic	Intermediate	Abundant	27.0
8/20/2018	A-18	44.04881	-89.22703	0.83	11.0	Organic	Intermediate	Abundant	15.0
8/20/2018	A-18	44.04893	-89.22769	1.08	12.0	Organic	Intermediate	Abundant	18.0
8/20/2018	B-18	44.05478	-89.23603	1.67	15.0	Organic	Intermediate	Abundant	24.0
8/20/2018	A-18	44.04890	-89.22797	1.33	9.0	Organic	Intermediate	Abundant	24.0
8/20/2018	A-18	44.04897	-89.22778	1.33	9.0	Organic	Intermediate	Abundant	15.0
8/20/2018	A-18	44.04895	-89.22789	0.58	9.0	Organic	Intermediate	Abundant	7.5
8/20/2018	A-18	44.04895	-89.22778	0.33	9.0	Organic	Intermediate	Abundant	3.0
8/20/2018	A-18	44.04895	-89.22781	1.80	9.0	Organic	Intermediate	Abundant	9.0
8/20/2018	A-18	44.04889	-89.22763	0.88	9.0	Organic	Intermediate	Abundant	15.0
8/20/2018	A-18	44.04889	-89.22763	0.75	9.0	Organic	Intermediate	Abundant	12.0
								Sub-Total	**340.0**

Detailed Diving Activities

EWM Treatment Results:

Date	Location	Latitude	Longitude	Time Underwater (Hrs)	Water Depth	Substrate Type	Plant Condition	Native Growth	Estimated EWM Removed (Cubic Feet)
8/21/2018	B-18	44.05463	-89.23605	0.98	14.0	Gravel	Intermediate	Abundant	10.0
8/21/2018	B-18	44.05465	-89.23618	1.58	12.0	Organic	Intermediate	Abundant	18.0
8/21/2018	B-18	44.05501	-89.23601	1.08	11.0	Organic	Intermediate	Abundant	7.5
8/21/2018	E-18	44.05875	-89.23582	1.75	11.0	Organic	Intermediate	Abundant	18.5
8/21/2018	E-18	44.05893	-89.23593	0.50	11.0	Organic	Intermediate	Abundant	7.5
8/21/2018	A-18	44.04880	-89.22701	1.00	7.0	Organic	Intermediate	Abundant	1.0
8/21/2018	A-18	44.04880	-89.22737	0.72	7.0	Organic	Intermediate	Abundant	2.5
8/21/2018	D-18	44.06114	-89.24417	2.25	14.0	Organic	Intermediate	Abundant	19.0
8/21/2018	D-18	44.06116	-89.24413	1.00	14.0	Organic	Intermediate	Sparse	23.0
8/21/2018	D-18	44.06116	-89.24413	0.67	14.0	Organic	Intermediate	Sparse	18.0
8/22/2018	F-18	44.05704	-89.23624	2.33	13.0	Gravel	Intermediate	Abundant	4.5
8/22/2018	F-18	44.05697	-89.23608	1.25	12.0	Gravel	Intermediate	Abundant	5.0
8/22/2018	F-18	44.05692	-89.23595	1.08	12.0	Gravel	Intermediate	Abundant	5.0
8/22/2018	F-18	44.05714	-89.23622	1.67	0.0	Gravel	Intermediate	Abundant	14.5
8/22/2018	E-18	44.05899	-89.23605	2.42	10.5	Organic	Intermediate	Abundant	5
8/22/2018	G-18	44.06028	-89.23799	1.42	10.0	Organic	Intermediate	Abundant	6
8/22/2018	G-18	44.06037	-89.23806	1.25	9.0	Organic	Intermediate	Abundant	10.5
8/22/2018	G-18	44.06043	-89.23812	1.75	9.0	Organic	Intermediate	Abundant	6.5
8/23/2018	F-18	44.05706	-89.23619	2.75	9.0	Gravel	Intermediate	Sparse	10.5
8/23/2018	F-18	44.05706	-89.23619	1.50	10.0	Gravel	Intermediate	Sparse	3.5
8/23/2018	F-18	44.05706	-89.23619	1.55	11.0	Gravel	Intermediate	Sparse	12.5
8/23/2018	F-18	44.05692	-89.23600	0.58	15.0	Organic	Intermediate	Abundant	0.3
8/24/2018	Fox	44.04744	-89.22063	2.25	12.0	Organic	Intermediate	Abundant	7.0
8/24/2018	Fox	44.04768	-89.22072	1.42	8.0	Organic	Intermediate	Abundant	7.0
8/24/2018	Fox	44.04744	-89.22064	1.00	10.0	Organic	Intermediate	Abundant	1.0
8/24/2018	Fox	44.04820	-89.22050	1.00	12.0	Organic	Intermediate	Abundant	3.5
								Sub-Total	**227.3**

Detailed Diving Activities

EWM Treatment Results:

Date	Location	Latitude	Longitude	Time Underwater (Hrs)	Water Depth	Substrate Type	Plant Condition	Native Growth	Estimated EWM Removed (Cubic Feet)
8/27/2018	Fox	44.04657	-89.22049	1.00	10.0	Organic	Healthy	Abundant	1.5
8/27/2018	Fox	44.04848	-89.22034	1.33	13.0	Organic	Intermediate	Abundant	3.0
8/27/2018	F-18	44.05685	-89.23600	1.58	14.0	Organic	Intermediate	Abundant	15.0
8/27/2018	F-18	44.05685	-89.23600	0.67	13.0	Organic	Intermediate	Abundant	6.0
8/28/2018	E-18	44.05876	-89.23582	0.92	13.0	Organic	Healthy	Abundant	1.0
8/28/2018	E-18	44.05876	-89.23582	0.50	13.0	Organic	Healthy	Abundant	5.5
8/29/2018	Clump	44.05219	-89.22390	1.33	15.0	Organic	Intermediate	Abundant	6.0
8/29/2018	Clump	44.05417	-89.22778	1.00	13.0	Organic	Intermediate	Abundant	4.5
8/29/2018	Clump	44.05219	-89.22390	0.75	15.0	Organic	Intermediate	Abundant	6.0
8/29/2018	Clump	44.05419	-89.22778	0.92	13.0	Organic	Intermediate	Abundant	3.0
8/29/2018	Clump	44.05219	-89.22390	0.33	15.0	Organic	Intermediate	Abundant	2.5
								Sub-total	**54.0**
								Grand Total	**621.3**

Native By-Catch (Cubic Feet)	Pondweeds	Elodea	Northern Milfoil	Eelgrass	Coontail
8/13/2018	0.50	0.25	1.25	0.25	0.25
8/14/2018	0.25	2.50	0.00	0.50	1.00
8/15/2018	0.25	0.50	2.50	1.25	0.00
8/16/2018	0.75	0.25	0.50	0.25	0.00
8/20/2018	0.50	0.25	0.75	2.50	1.00
8/21/2018	1.50	1.50	0.75	0.50	1.25
8/22/2018	4.50	0.75	0.75	1.0	2.0
8/23/2018	0.50	0.75	0.25	1.50	0.00
8/27/2018	0.75	0.50	0.50	2.50	0.50
8/28/2018	2.50	1.50	0.00	1.25	0.00
8/29/2018	0.75	0.75	0.00	0.00	0.00
Total	**12.75**	**9.5**	**7.25**	**11.5**	**6**

B

APPENDIX B

2012-2018 Aquatic Plant Frequencies

Appendix B. Silver Lake Chi Square Analysis.

			LFOO (%)						2015-2017		2017-2018	
	Scientific Name	**Common Name**	**2012**	**2013**	**2014**	**2015**	**2017**	**2018**	**% Change**	**Direction**	**% Change**	**Direction**
Dicots	*Myriophyllum sibiricum X spicatum*	Hybrid watermilfoil	25.3	33.3	7.8	20.0	0.2	1.7	-98.9	▼	694.8	▲
	Ceratophyllum demersum	Coontail	32.9	32.9	42.5	36.3	20.2	25.5	-44.3	▼	26.1	△
	Myriophyllum sibiricum	Northern watermilfoil	0.6	5.4	0.7	0.2	0.0	0.0	-100.0	▽		-
	Bidens beckii	Water marigold	0.0	0.0	0.2	0.0	0.4	1.1		△	148.4	△
	Ranunculus aquatilis	White water crowfoot	0.0	0.0	0.2	0.4	0.0	0.0	-100.0	▽		-
	Utricularia geminiscapa	Twin-stemmed bladderwort	0.0	0.0	0.0	0.0	0.0	0.2		-		△
	Nymphaea odorata	White water lily	0.2	0.0	0.0	0.0	0.0	0.0		-		-
	Ceratophyllum echinatum	Spiny hornwort	0.0	0.0	0.0	0.0	0.2	0.0		△	-100.0	▽
Non-dicots	*Potamogeton crispus*	Curly-leaf pondweed	0.2	0.0	0.0	0.2	1.1	0.0	400.0	△	-100.0	▼
	Chara spp. & Nitella spp.	Muskgrasses & Stoneworts	33.8	25.1	30.0	23.9	25.7	33.5	7.3	△	30.5	▲
	Chara spp.	Muskgrasses	28.7	15.9	19.2	17.4	22.2	31.3	27.5	△	41.2	▲
	Elodea canadensis	Common waterweed	27.8	35.1	28.2	19.1	0.0	1.3	-100.0	▼		▲
	Najas guadalupensis	Southern naiad	24.7	21.3	22.6	16.5	3.7	9.7	-77.6	▼	163.0	▲
	Potamogeton gramineus	Variable-leaf pondweed	20.7	13.6	10.7	11.1	3.5	15.1	-68.6	▼	334.7	▲
	Vallisneria americana	Wild celery	11.0	8.9	11.6	12.4	9.8	14.3	-21.1	▽	45.7	▲
	Stuckenia pectinata	Sago pondweed	6.5	5.1	6.5	8.9	7.4	10.4	-17.1	▽	40.3	△
	Fissidens spp. & Fontinalis spp.	Aquatic Moss	5.5	10.1	7.4	4.8	0.0	9.7	-100.0	▼		▲
	Nitella spp.	Stoneworts	8.0	9.4	11.4	8.5	3.7	2.2	-56.4	▼	-41.6	▽
	Potamogeton friesii	Fries' pondweed	2.1	3.4	2.2	11.7	4.6	3.9	-61.1	▼	-14.8	▽
	Potamogeton praelongus	White-stem pondweed	4.6	4.0	8.5	4.8	1.5	4.1	-68.2	▼	169.7	▲
	Potamogeton pusillus	Small pondweed	0.0	2.0	2.2	4.8	3.5	8.9	-27.3	▽	154.6	▲
	Potamogeton zosteriformis	Flat-stem pondweed	1.5	2.2	2.0	2.6	6.1	7.3	133.3	▲	20.6	△
	Filamentous algae	Filamentous algae	7.0	2.2	3.8	13.3	1.1	0.2	-91.8	▼	-80.1	▽
	Potamogeton illinoensis	Illinois pondweed	4.6	2.7	4.7	5.4	1.5	0.6	-72.0	▼	-57.4	▽
	Najas flexilis	Slender naiad	3.2	1.8	3.8	3.0	0.0	3.2	-100.0	▼		▲
	Heteranthera dubia	Water stargrass	0.6	0.2	0.2	0.0	2.4	4.3		▲	80.6	△
	Potamogeton foliosus	Leafy pondweed	3.6	0.2	2.9	0.7	1.7	0.2	166.7	△	-87.6	▼
	Potamogeton strictifolius	Stiff pondweed	0.0	0.0	0.0	0.0	1.5	0.9		▲	-43.2	▽
	Potamogeton amplifolius	Large-leaf pondweed	0.6	0.0	0.7	0.0	0.0	0.6		-		△
	Potamogeton natans	Floating-leaf pondweed	0.0	0.7	0.4	0.9	0.0	0.2	-100.0	▼		△
	Eleocharis acicularis	Needle spikerush	0.0	0.0	0.2	0.7	0.4	0.2	-33.3	▽	-50.3	▽
	Spirodela polyrhiza	Greater duckweed	0.0	0.0	0.0	0.0	0.4	0.0		△	-100.0	▽
	Schoenoplectus pungens	Three-square rush	0.0	0.0	0.0	0.0	0.0	0.2		-		△
	Potamogeton spirillus	Spiral-fruited pondweed	0.0	0.0	0.0	0.0	0.4	0.0		△	-100.0	▽
	Elodea nuttallii	Slender waterweed	0.0	0.4	0.0	0.0	0.0	0.0		-		-
	Freshwater sponge	Freshwater sponge	0.2	0.0	0.0	0.0	0.0	0.0		-		-

▲ or ▼ = Change Statistically Valid (Chi-square; α = 0.05)
△ or ▽ = Change Not Statistically Valid (Chi-square; α = 0.05)

C

APPENDIX C

ProcelleCOR™ Materials

- **WDNR Chemical Fact Sheet on florpyrauxifen-benzyl (ProcelleCOR™)**
- **Final Supplemental Environmental Impact Statement for State of Washington Aquatic Plant and Algae Management. State of Washington Department of Ecology. August 14, 2017.** Full report found at:

  https://fortress.wa.gov/ecy/publications/documents/1710020.pdf

Wisconsin Department of Natural Resources

July 2018

Florpyrauxifen-benzyl Chemical Fact Sheet

Formulations

Florpyrauxifen-benzyl was registered with the EPA for aquatic use in 2017. The active ingredient is 2-pyridinecarboxylic acid, 4-amino-3-chloro-6-(4-chloro-2-fluoro-3-methoxyphenyl)-5-fluoro-, phenyl methyl ester. The current Wisconsin-registered formulation is a liquid (ProcellaCOR™ EC) solely manufactured by SePRO Corporation.

Aquatic Use and Considerations

Florpyrauxifen-benzyl is a systemic herbicide that is taken up by aquatic plants. The herbicide is a member of a new class of synthetic auxins, the arylpicolinates, that differ in binding affinity compared to other currently registered synthetic auxins. The herbicide mimics the plant growth hormone auxin that causes excessive elongation of plant cells that ultimately kills the plant. Susceptible plants will show a mixture of atypical growth (larger, twisted leaves, stem elongation) and fragility of leaf and shoot tissue. Initial symptoms will be displayed within hours to a few days after treatment with plant death and decomposition occurring over 2 – 3 weeks. Florpyrauxifen-benzyl should be applied to plants that are actively growing; mature plants may require a higher concentration of herbicide and a longer contact time compared to smaller, less established plants.

Florpyrauxifen-benzyl has relatively short contact exposure time (CET) requirements (12 – 24 hours typically). The short CET may be advantageous for localized treatments of submersed aquatic plants, however, the target species efficacy compared to the size of the treatment area is not yet known.

In Wisconsin, florpyrauxifen-benzyl may be used to treat the invasive Eurasian watermilfoil (*Myriophyllum spicatum*) and hybrid Eurasian watermilfoil (*M. spicatum* X *M. sibiricum*). Other invasive species such as floating hearts (*Nymphoides* spp.) are also susceptible. In other parts of the country, it is used as a selective, systemic mode of action for spot and partial treatment of the invasive plant hydrilla (*Hydrilla verticillata*). Desirable native species that may also be negatively affected include waterlily species (*Nymphaea* spp. and *Nuphar* spp.), pickerelweed (*Pontederia cordata*), and arrowhead (*Sagittaria* spp.).

It is important to note that repeated use of herbicides with the same mode of action can lead to herbicide-resistant plants, even in aquatic plants. Certain hybrid Eurasian watermilfoil genotypes have been documented to have reduced sensitivity to aquatic herbicides. In order to reduce the risk of developing resistant genotypes, avoid using the same type of herbicides year after year, and utilize effective, integrated pest management strategies as part of any long-term control program.

Post-Treatment Water Use Restrictions

There are no restrictions on swimming, eating fish from treated waterbodies, or using water for drinking water. There is no restriction on irrigation of turf. Before treated water can be used for non-agricultural irrigation besides turf (such as shoreline property use including irrigation of residential landscape plants and homeowner gardens, golf course irrigation, and non-residential property irrigation around business or industrial properties), follow precautionary waiting periods based on rate and scale of application, or monitor herbicide concentrations until below 2 ppb. For agricultural crop irrigation, use analytical monitoring to confirm dissipation before irrigating. The latest approved herbicide product label should be referenced relative to irrigation requirements.

Herbicide Degradation, Persistence and Trace Contaminants

Florpyrauxifen-benzyl is broken down quickly in the water by light (i.e., photolysis) and is also subject to microbial breakdown and hydrolysis. It has a half-life (the time it takes for half of the active ingredient to degrade) ranging from 1 – 6 days. Shallow clear-water lakes will lead to faster degradation than turbid, shaded, or deep lakes.

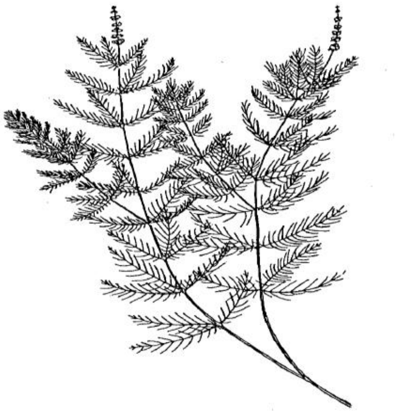
Florpyrauxifen-benzyl breaks down into five major degradation products. These materials are generally more persistent in water than the active herbicide (up to 3 week half-lives) but four of these are minor metabolites detected at less than 5% of applied active ingredient. EPA concluded no hazard concern for metabolites and/or degradates of florpyrauxifen-benzyl that may be found in drinking water, plants, and livestock.

Florpyrauxifen-benzyl binds tightly with surface sediments, so leaching into groundwater is unlikely. Degradation products are more mobile, but aquatic field dissipation studies showed minimal detection of these products in surface sediments.

Impacts on Fish and Other Aquatic Organisms

Toxicity tests conducted with rainbow trout, fathead minnow, water fleas (*Daphnia* sp.), amphipods (*Gammarus* sp.), and snails (*Lymnaea* sp.) indicate that florpyrauxifen-benzyl is not toxic for these species. EPA concluded florpyrauxifen-benzyl has no risk concerns for non-target wildlife and is considered "practically non-toxic" to bees, birds, reptiles, amphibians, and mammals.

Florpyrauxifen-benzyl does not bioaccumulate in fish or freshwater clams due to rapid metabolism and chemical depuration.



Human Health

EPA has identified no risks of concern to human health since no adverse acute or chronic effects, including a lack of carcinogenicity or mutagenicity, were observed in the submitted toxicological studies for florpyrauxifen-benzyl regardless of the route of exposure. EPA concluded with reasonable certainty that drinking water exposures to florpyrauxifen-benzyl do not pose a significant human health risk.

For Additional Information

Environmental Protection Agency Office of Pesticide Programs
www.epa.gov/pesticides

Wisconsin Department of Agriculture, Trade, and Consumer Protection
http://datcp.wi.gov/Plants/Pesticides/

Wisconsin Department of Natural Resources
608-266-2621
http://dnr.wi.gov/lakes/plants/

National Pesticide Information Center
1-800-858-7378
http://npic.orst.edu/

Washington State Department of Ecology. 2017.
https://fortress.wa.gov/ecy/publications/documents/1710020.pdf

4.3 EVALUATION OF PROCELLACOR™ (FLORPYRAUXIFEN-BENZYL)

NOTE: GEI Consultants, Inc. executed a confidential non-disclosure agreement with SePRO Corporation to obtain and review proprietary studies and data. SePRO is working in partnership with Dow AgroSciences to develop this technology for aquatic weed control. In the absence of peer-reviewed journal articles or other scientific literature, these studies—many of which were performed in support of EPA's Office of Pesticide Programs (OPP) registration requirements—were used to prepare the evaluation of the candidate aquatic herbicide.

4.3.1 Registration Status

PROCELLACOR^{TM} (Procellacor™) Aquatic Herbicide (2-pyridinecarboxylic acid, 4-amino-3-chloro-6-(4-chloro-2-fluoro-3-methoxyphenyl)-5-fluoro-, phenylmethyl ester also known as Rinskor™; common name: florpyrauxifen-benzyl) has not yet been registered nationally by the EPA or in Washington State by the WSDA under 15.58 Revised Code of Washington (RCW). This SEIS provides technical, environmental, and other information required by Ecology to determine whether to add Procellacor™ to existing water quality NPDES permits, which will allow this herbicide to be discharged to the waters of the State as allowed under the Clean Water Act.

Procellacor™ (florpyrauxifen-benzyl)was granted Reduced Risk status by EPA under the Pesticide Registration Improvement Act (PRIA) Version 3 (https://www.epa.gov/pria-fees/pria-overview-and-history#pria3) in early 2016 (Denny, Breaux, 2016; also see notification letter at Attachment A) because of its promising environmental and toxicological profiles in comparison to currently registered herbicides utilized for partial treatment of hydrilla, invasive watermilfoils, and other noxious plant species. EPA concluded that the overall profile appeared more favorable when compared to the registered alternatives for the proposed use patterns for these noxious species, and that the reduction in risk pertaining to human health was the driving factor in this determination. As discussed later in the document, Procellacor™ shows excellent selectivity with few or limited impacts to native aquatic plants such as aquatic grasses, bulrush, cattail, pondweeds, naiads, and tapegrass. In its review, EPA also noted that the overall profile for the herbicide appears favorable when compared to currently registered alternative herbicides (e.g. 2,4-D, endothall, triclopyr) for this aquatic use pattern. Procellacor™ represents an alternative mode of chemical action which is more environmentally favorable than currently registered aquatic herbicides. Florpyrauxifen-benzyl would be expected to offer improvements in IPM for control of noxious aquatic weeds. The alternative mode of action should also help to prolong the effectiveness of many aquatic herbicide solutions by offering a new rotation or combination alternative as part of herbicide resistance management strategies.

The new candidate aquatic herbicide is under expedited review from EPA under the PRIA per the Reduced Risk status designation discussed above, with an anticipated registration date of summer 2017. As part of the review, EPA's OPP is also currently conducting human health and ecological risk assessments with an expected date of release in late spring 2017. This SEIS document relies on information currently available at this time, much of which necessarily is limited to data provided by Dow AgroSciences and SePRO Corporation in developing and testing the herbicide. It can be revised with more updated information following the release of EPA review information as well as other peer-reviewed literature expected to be released later in 2017. Dow AgroSciences has also concurrently applied to EPA for registration of the florpyrauxifen-benzyl active ingredient for weed control in rice paddies. The initial Procellacor™ formulation is expected to be a 300 g TGAI/L suspension concentrate. Control of hydrilla and invasive watermilfoils can be achieved at in-water spot/partial treatment rates of 10 to 50 µg a.i./L with Procellacor™, as opposed to rates of 1,000 to 5,000 µg a.i./L for endothall, 2,4-D, and triclopyr (Getsinger 2016, Beets and Netherland 2017a *in review*, Netherland et al 2017 *in prep*).

This analysis considers florpyrauxifen-benzyl's (Procellacor™'s) mode of action, efficacy, and range of in-water treatment concentrations required to achieve control across different water exchange / exposure scenarios. The review discusses results of mesocosm and other field studies conducted in partial site and whole pond treatments, described in more detail below.

To help expedite development and future adoption of the technology, SePRO has been working with numerous partners and collaborators to conduct experimental applications to confirm field efficacy on a variety of target aquatic vegetation, as well as to document non-target effects or impacts. As an unregistered product that does not have a federal experimental use permit, EPA guidelines require that field testing be limited to one acre or less of application per target pest species and that uses of water potentially affected by this application such as swimming, fishing, and irrigation be restricted. The discussion below provides a summary of the herbicides' physical properties, mammalian and ecotoxicological information, environmental fate, and other requirements for EPA registration. Most of these studies have been conducted by Dow AgroSciences and SePRO Corporation in fulfillment of EPA's OPP pesticide registration requirements under FIFRA (as represented by Heilman 2016). As noted above, few peer-reviewed publications have yet been released, although more are expected later in 2017 and beyond.

4.3.2 Description

Procellacor™ is the aquatic trade name for use of a new active ingredient (florpyrauxifen-benzyl), which is one chemistry in a novel class of herbicides known as the arylpicolinates. The primary end-use formulation anticipated for in-water application at time of registration is a 300 g active ingredient/liter suspension concentrate, but other aquatic use formulations are being considered for registration shortly after the initial EPA decision.

Aquatic herbicides are grouped by contact (controls plant shoots only) vs. systemic (controls entire plant), and by aqueous concentration and exposure time (CET) requirements. In general, contact products are quicker acting with shorter CET requirements, while systemic herbicides are slower acting with longer CET requirements. In light of this, Procellacor™ is quick-acting, has relatively short CET requirements, is systemic, and requires low application rates compared to other currently registered herbicides. Moreover, it has shown short persistence in both water and sediment relative to currently registered herbicides such as endothall, 2,4-D, and triclopyr, is species-selective, and has minimal non-target effects to both plant and animal species. Its effective chemical mode of action and high selectivity for aquatic invasive and noxious plants provides a significant impetus for its development and eventual registration. Procellacor™ has demonstrated this selective, systemic activity with relatively short CET requirements on several major aquatic weed species, including hydrilla and invasive watermilfoils. Netherland and Richardson (2016) and Richardson *et al*. (2016) investigated the sensitivity of numerous aquatic plant species to the compound, and provided verification of Procellacor™'s activity on key invasives and greater tolerance by the majority of native aquatic plants tested to date. Additional government and university research has documented high activity and different selectivity patterns relative to possible impacts to non-target aquatic vegetation compared to other currently registered, well-documented herbicides such as triclopyr, endothall, and/or 2,4-D (Beets and Netherland 2017a *in review*, Beets and Netherland 2017b *in prep*, Haug and Richardson 2017 *in prep*).

4.3.2.1 Environmental Characteristics: Product Use and Chemistry

Procellacor™ shows excellent activity on several major US aquatic weeds including hydrilla (*H. verticillata*) and multiple problematic watermilfoils (*Myriophyllum spp.)*, including Eurasian (EWM) and hybrid Eurasian (*M. spicatum X M. sibiricum*), parrotsfeather (*M. aquaticum*), and variable-leaf milfoil (*M. heterophyllum*). Procellacor™ provides a new systemic mode of action for hydrilla control and a new class of auxin-mimic herbicide chemistry for selective management of invasive watermilfoils. It also has in-water or foliar herbicidal activity on a number of noxious emergent and floating aquatic plants such as water hyacinth and invasive floating hearts (*Nymphoides spp*.). Procellacor™ has low application rates (50 µg/L or less) for systemic activity with short CET requirements (12 – 72 hours depending on rate and target weed) allowing for spot and/or partial in-water applications. For such treatments, Procellacor™ provides selective control with several hundred times less herbicide use versus current in-water, spot treatment herbicides such as endothall (5,000 µg/L maximum use rate for dipotassium salt form) and 2,4-D (4,000 µg/L maximum use rate). Procellacor™ also appears to show high selectivity with few impacts to native aquatic plants such as aquatic grasses, bulrush, cattail, pondweeds, naiads, and tapegrass (see discussion on selectivity below).

Procellacor™ is effective in controlling hydrilla, and offers a new pattern of selectivity for removing hydrilla from mixed aquatic-plant communities. The strong activity of this new alternative mode of action supports its development for selective hydrilla control. Mesocosm studies summarized by Heilman (2016) and in preparation or under active review for peer-reviewed publication have shown that control of standing biomass of hydrilla and EWM can be achieved in two to three weeks, with high activity even on 2,4-D and triclopyr-tolerant stands of hybrid EWM (Beets and Netherland 2017a *in review*, Netherland et al. 2017 *in prep*). Multiple small-scale laboratory screening studies were conducted to support both target weed activity and regulatory consideration of potential effects of Procellacor™ on non-target aquatic vegetation. The test plant EC_{50} response (herbicide concentration having 50% effect) to static exposures of Procellacor™ was determined for 12 different plant species: the general EC_{50} range was approximately 0.11 µg/L to greater than 81 µg/L (Netherland and Richardson, 2016; Richardson *et al*., 2016). Similar small-scale comparative efficacy testing of Procellacor™ vs. 2,4-D and triclopyr on multiple invasive watermilfoils confirms orders of magnitude greater activity with Procellacor™ versus the older auxin herbicides, including activity on hybrid EWM with documented tolerance to the older herbicides (Beets and Netherland 2017b *in prep*). These findings are promising for Procellacor™, as they support significantly lower herbicide application rates combined with a favorable environmental profile, discussed in more detail below.

4.3.2.2 Environmental Mobility and Transport

Procellacor™/Rinskor is known to have low water solubility (laboratory assay of TGAI: 10 to 15 µg/L at pH 5 to 9, 20°C), low volatility (vapor pressure approx. 10^{-7} mm Hg), with moderately high partition coefficients (log K_{ow} values of approximately 5.4 to 5.5), which describe an environmental profile of low solubility and relatively high affinity for sorption to organic substrates.

The environmental fate of the herbicide in soil and water has been characterized as part of the registration package and is well understood. The parent compound is not persistent and degrades via a number of pathways including photolysis, aerobic soil degradation, aerobic aquatic degradation, and/or hydrolysis to a number of hydroxyl, benzyl-ester, and acid metabolites. In aerobic soil, Procellacor™ degrades moderately quickly, with half-lives ranging from 2.5 to 34 days, with an average of 15 days. Anaerobic soil metabolism studies also show relatively rapid degradation rates, with half-lives ranging from 7 to 15 days, and an average of 9.8 days. The herbicide is short-lived, with half-lives ranging from 4 to 6 days and 2 days, respectively, in aerobic and anaerobic aquatic environments, and in total water-sediment systems such as mesocosms. These half-lives are consistently rapid compared to other currently registered herbicides such as 2,4-D, triclopyr, and endothall. Degradation in surface water is accelerated when exposed to sunlight, with a reported photolytic half- life in laboratory testing of 0.07 days.

In two outdoor aquatic dissipation studies, as summarized by Heilman (2016), the SC formulation of the herbicide was directly injected into outdoor ponds at nominal rates of 50 and 150 µg/L as the active ingredient. Water phase dissipation half-lives of 3.0 – 4.9 days were observed, which indicates that the material does not persist in the aquatic environment. With conditions similar to wetland and marsh habitat, results from another field dissipation study in rice paddies that incorporated appropriate water management practices for both wet-seeded and dry-seeded rice (also reported by Heilman 2016) resulted in aquatic-phase half-lives ranging from 0.15 to 0.79 days, and soil phase half-lives ranging from 0.0037 to 8.1 days These results do not indicate a tendency to persist in the aquatic environment. The herbicide can be classified as generally immobile based on soil log K_{oc} values in the order of 10^{-5}, and suggest that the potential for off-site transport is minimal. This is consistent with numerous observations that Procellacor™ undergoes rapid degradation in the soil and aqueous environments via a number of degradation mechanisms, summarized above.

4.3.2.3 Field Surveys and Investigations

A human health and ecological risk assessment is currently being conducted by EPA Office of Pesticide Programs. Results of this assessment are expected to be released during spring of 2017 (Denny, 2016), and these conclusions will either support or refute data already collected for Procellacor™. There are no preliminary findings to report, but based on the current understanding of available environmental fate, chemistry, toxicological, and other data, there is little to no cause for concern to human health or ecotoxicity for acute, chronic, or subchronic exposures to Procellacor™ formulations.

4.3.2.4 Bioconcentration and Bioaccumulation

A fish bioconcentration factor study and magnitude of residue studies for clam, crayfish, catfish, and bluegill support that, as anticipated from its physical chemistry and organic affinity, Procellacor™/Rinskor will temporarily bioaccumulate but is rapidly depurated and/or metabolized within freshwater organisms within 1 – 3 days after exposure to high concentrations (150 µg/L or higher). Based on these findings and the low acute and chronic toxicity to a wide variety of receptor organisms, summarized below, bioconcentration or bioaccumulation are not expected to be of concern for the

Procellacor™ aquatic use. EPA's forthcoming human health and ecological risk assessment will include exposure scenarios that will help to further clarify and refine the understanding of bioconcentration or bioaccumulation potential for Procellacor™.

4.3.2.5 Toxicological Profile

Mammalian and Human Toxicity

Extensive mammalian toxicity testing of Procellacor™ has been conducted by the proposed registrant, and results have shown little evidence of acute or chronic toxicity. Acute mammalian toxicity testing for Procellacor™ showed very low acute toxicity by oral or dermal routes (LD_{50} values greater than 5,000 mg/kg). Acute toxicity is also reported low via the inhalation route of exposure (LC_{50} value greater than 5.2 mg/L). Procellacor™ is reported not to be an irritant to eyes or skin and only demonstrated a weak dermal sensitization potential in a mouse local lymph node assay (EC_3 of 19.1%).

Absorption, distribution, metabolism, and elimination profiles have been developed for Procellacor™. In summary, Procellacor™ has demonstrated rapid absorption (T_{max} of 2 hours), with higher absorption rates at lower doses (36 to 42% of the administered dose), rapid hydrolysis, and rapid elimination via the feces (51 to 101%) and urine (8 to 42%) during the first 24 hours following administration to laboratory mammals. In general, the lower doses tested would be more representative of levels potentially encountered by people, mammals, or other organisms.

Based on laboratory testing, Procellacor™ is not genotoxic, and there was no treatment-related toxicity even up to the highest doses tested in the acute, short-term, two generation reproduction or developmental toxicity studies or in the acute or subchronic neurotoxicity studies. Chronic administration of the herbicide did not show any carcinogenicity potential and did not cause any adverse effects in mice, rats or dogs, at the highest doses tested. In summary, studies conducted in support of EPA registration indicate there is little or no concern for acute, short term, subchronic or chronic dietary risk to humans from Procellacor™ applications. Tests have shown no evidence of genotoxicity/carcinogenicity, immunotoxicity, neurotoxicity, subchronic or chronic toxicity, reproductive or developmental toxicity, and only showed evidence of low acute toxicity.

Several studies conducted on both mice and rats, over the course of 1-2 years have indicated no treatment-related (post-necropsy) clinical observations or gross histopathological lesions. An 18-month mouse study was conducted, and no chronic toxicity, carcinogenicity, or other adverse effects were observed, even in those male and female mice receiving the highest doses tested. A 1-year dog study is also ongoing; similar to the above mammalian toxicity tests, no treatment-related toxicity or pathology has yet been observed during this study. Reproductive, developmental, and endocrine toxicity (immunotoxicity) has also been tested, and results of all these tests showed no evidence of toxicity. Although no specific human testing has been conducted for Procellacor™, based on extensive laboratory testing on mammalian species, little to no acute or chronic toxicity would be expected in association with environmental exposures.

General Ecotoxicity

Procellacor™ has undergone extensive ecotoxicological testing and has been shown to be nearly non-toxic to birds in acute oral, dietary, and reproduction studies. Similar to the mammalian testing summarized above, no toxicity was observed for avian, fish, or other species exposed to the herbicide in acute and long-term studies, with endpoints set at the highest concentration tested, which are well above those actually released as part of label-specified application of Procellacor™. As would be expected for an herbicide, toxicity has been observed to certain sensitive terrestrial and aquatic plants (see plant discussion below).

As noted above, the TGAI of Procellacor™ exhibits low water solubility, and in laboratory aquatic ecotoxicity studies, the highest concentration of TGAI that could be dissolved in the test water (or functional solubility) was approximately 40-60 µg/L in freshwater. The acute and/or chronic endpoints for freshwater fish and invertebrates are generally at, or above, the limit of functional solubility. Additional evaluations indicate a lack of toxicity of the aquatic end-use product (greater functional solubility than the TGAI) and metabolites up to several orders of magnitude above the typical in-water use rates of Procellacor™ (50 µg/L or less).

Fish Ecotoxicity

A variety of fish tests have been conducted in cold and warm water fish species using the TGAI as well as the end-use formulation and various metabolites. Acute toxicity results using rainbow trout (*O. mykiss*, a standard cold water fish testing species) indicated LC_{50} values of greater than 49 µg/L, and greater than 41 µg/L for fathead minnow (*P. promelas*, a standard warm water species). The pure TGAI would not be expected to be released into the environment, and comparable acute ecotoxicity testing was performed for carp using an end-use formulation for Procellacor™. Results indicate an LC_{50} value of greater than 1,900 µg/L for carp (*C. carpio*), indicating much lower acute toxicity potential. A marine toxicity test was identified, where sheepshead minnows (*C. variegatus*) were tested for acute toxicity, and a LC_{50} value of greater than 40 µg/L was produced, which is comparable to freshwater species tested for acute toxicity. This value is indicative of slight acute toxicity potential if environmental concentrations were to be present at these levels, which is unlikely. Comparable acute ecotoxicity testing using various Procellacor™ metabolites indicated LC_{50} values uniformly greater than 1,000 µg/L, indicating a minimal potential for acute toxicity from metabolites. Salmonid toxicity data also indicated no overt toxicity to juvenile rainbow trout at limit of solubility for both the TGAI and end-use formulation at the maximum application rate (40 µg/L). If fish were to occupy a plant-infested littoral zone that was treated by Procellacor™, no toxic exposure would be expected to occur, as toxicity thresholds would not be exceeded by the concentrations predicted to be allowed for use by the FIFRA label.

Fish toxicity testing, in addition to that summarized above, has been planned and is currently under way for sensitive and ESA-listed aquatic species and habitat considerations in the Pacific Northwest, as reported by Grue (2016 and 2017). The emphasis for this aquatic toxicity testing is on salmonid species (Chinook salmon, bull trout, coho salmon, etc.), which are the most frequently listed and probably the most representative fish species in the Northwest under ESA. The most commonly accepted surrogate fish test species for salmonids is the cold water salmonid rainbow trout (*O. mykiss*), but to help alleviate additional uncertainty, this additional testing will use age- and species- appropriate salmon species, and is intended to replicate pre-registration toxicity tests with trout using environmentally representative exposure concentrations. Test endpoints include acute mortality, growth, and other sublethal and behavioral endpoints (e.g. erratic swimming, on-bottom gilling, etc.) to evaluate more subtle toxicological effects potentially associated with Procellacor™. Preliminary results from this testing indicate little to no effects associated with exposure to florpyrauxifen-benzyl, and a final report on this work will be forthcoming later in 2017.

This testing will screen comparable treatments to the trout testing (0, 40 and 80 µg/L Procellacor™, with the latter being well in excess of anticipated maximum labeled use rate). Testing will follow standard guidelines (ASTM, 2002; EPA, 1996) as did the earlier testing (e.g. Breaux, 2015), to ensure comparability. Results from this additional testing are expected to become available by late spring 2017, and will be useful in expanding our understanding of the toxicological properties of Procellacor™ when used in salmon-bearing waters.

Avian Toxicity

As noted above, Procellacor™ has been shown to be of low acute and chronic toxicity to birds as shown in a series of acute oral, dietary, and reproduction studies (Breaux, 2015). Little to no toxicity was observed for avian species exposed to the herbicide in both acute and longer-term chronic studies, with the highest test concentrations exceeded expected labeled rates, a common practice in laboratory toxicology. Bird testing was conducted to include standard test species including mallard duck (*A. platyrhynchos*), the passerine (songbird) species zebra finch (*T. guttata*), and bobwhite quail (*C. virginianus*). Tests involved oral administration for acute and chronic testing and reproductive studies, eggshell thinning, life cycle testing, and other endpoints. In summary, acute oral testing using bobwhite quail and zebra finch yielded LD_{50} values of greater than 2,250 mg/kg-day for both species. Two five-day acute dietary tests were also conducted, which both yielded LC_{50} values of greater than 5,620 mg/kg-day. Subchronic reproductive tests were also conducted for bobwhite quail and mallard ducks both yielded NOEC values of 1,000 mg/kg in the feed. All of these results are highly indicative of little to no toxicity to each of the avian species tested.

No amphibian or reptile toxicity testing was required by EPA Office of Pesticide Programs registration requirements, or conducted as part of the testing regimen for Procellacor™. EPA guidelines generally assert that avian testing is an adequate surrogate for amphibian or reptile testing, and invertebrate and mammalian test results are available as well to support projection of minimal toxicity of Procellacor™ to amphibians or reptiles.

Invertebrate Ecotoxicity

Acute and chronic testing of Procellacor™ with honey bees, the only insect species tested, has indicated no evidence of ecotoxicity to this species (Breaux, 2015). Concerning aquatic invertebrates, acute testing was performed for both the daphnid *D. magna* and the midge *Chironomus* sp. Tests were conducted using both the TGAI and end-use formulation for Procellacor™, as well as various metabolites. Acute toxicity results for the TGAI using *D. magna* indicated LC_{50} values of greater than 62 µg/L, and greater than 60 µg/L for *Chironomus.* This is generally consistent with acute toxicity testing conducted for the freshwater amphipod *Gammarus* sp., for which a NOEC value of 42 µg/L was developed. These results are indicative of little to no acute toxicity to these species. Comparable acute ecotoxicity testing was performed for *D. magna* using a Procellacor™ end-use formulation, and results indicated an LC_{50} value of greater than 80,000 µg/L, also indicating negligible acute toxicity potential. Acute ecotoxicity testing using various metabolites of the herbicide indicated LC_{50} values uniformly greater than 980 µg/L, with most values exceeding 10,000 µg/L, indicating little to no potential for acute toxicity for the metabolites.

Life cycle testing was also completed for a freshwater (*D. magna*) for both the TGAI and metabolites, and results showed a Lowest Observable Adverse Effect Concentration (LOAEC) and an NOAEC of 38 µg/L (both endpoints) showing low toxicity potential for the TGAI in an artificial scenario of static exposure using a renewal protocol design. The spot/partial use pattern of the herbicide and instability of TGAI under natural conditions project to a lack of chronic exposure to aquatic fauna. Comparable testing with metabolites showed LOAEC/NOAEC values both exceeding 25,000 µg/L, indicating negligible levels of toxicity for metabolites. Whole sediment testing using the TGAI for a freshwater invertebrate (chironomid midge) was also conducted for acute (10 day) and chronic (28 day) duration. The chronic test spiked water overlying sediments to a target concentration as the means to initiate exposure. Results of the whole sediment testing indicated an acute 10-day LOAEC of 10.5 mg ai/kg sediment and 28-day NOEC level of 78.5 µg/L (overlying water target concentration), which would generally be indicative of very low to negligible aquatic ecotoxicity.

Additionally, acute screening was recently performed by North Carolina State University (Principal Investigator: Dr. Greg Cope, cited as Buczek *et al.* 2017) on the juvenile life stage of a representative freshwater mussel (*L. siliquoidea*) with the TGAI, a primary metabolite (acid metabolite), and two TEP / formulations (the SC above and a 25 g/L EC formulation). The study showed no toxicity to juvenile mussels in any test with formulated results showing No Effect Concentrations (NOEC) that were 25 – 50 times greater than anticipated maximum application rate for the new herbicide (Cope *et al.* 2017 *in prep*).

Although the proposed registration for Procellacor™ in Washington State will be for freshwater application, it is possible that Procellacor™ would be applied near marine or estuarine habitats for weed control. Acute toxicity testing, using TGAI, conducted on the eastern oyster (*C. gigas*) produced an NOEC of greater than 24 µg ai/L and a comparable NOEC value for mysid shrimp (*M. bahia*) of greater than 26 µg ai/L, both the highest rates tested due to solubility limits with assays. Comparable NOEC values developed for primary aquatic end-use formulation were greater than 1,100 and 1,350 µg/L as formulated product (>289 and >362 µg/L as active ingredient), respectively, for the oyster and shrimp.

Marine invertebrate life cycle testing was conducted using the TGAI on a mysid shrimp) and a chronic NOAEC of 7.8 µg/L (LOAEC of 13 µg/L) was developed, which is potentially indicative of chronic toxicity to marine or estuarine invertebrates if these sustained concentrations were attained in environmental settings. Acute NOECs for oyster and mysids tested with the TGAI were set at the highest mean measured rate of tested material. There were no adverse effects noted in those studies. There are potential unknowns with possible effects with acute exposures to concentrations greater than 24-26 µg/L, but range finding-finding toxicity testing demonstrated that this range of concentrations were the highest limits to maintain solubility of TGAI in the assays.

In practice, due to rapid degradation of the TGAI in the field, rapid dilution from spot applications (main use pattern), and not labelling for estuarine and marine sites will mitigate any chance of acute exposures to marine invertebrates above the range of mid-20 µg/L. Chronic toxicity results for mysid shrimp do suggest possible chronic effects at 7.8 µg/L, with extended exposures to the TGAI. Again, however, the use pattern is not intended for estuarine/marine application with the initial labelling. The use pattern in freshwater is spot/partial treatments with negligible chance of sustained TGAI concentrations migrating downstream to estuarine habitat even if the freshwater site was in close proximity to an estuarine area. In general, the labeled freshwater use for spot/partial applications (high dilution potential) to control noxious freshwater aquatic plants and the rapid degradation of the TGAI suggest minimal risk to marine and estuarine invertebrates following application to a nearby freshwater site. Metabolite testing with marine species yielded NOECs of greater than 25,000 µg/L, indicating negligible toxicity.

Data Gaps

No data gaps have been identified for the basic environmental profile, including environmental fate, product chemistry, toxicology and ecotoxicology, and field studies required by EPA for pesticide registration. However, a number of recent trials are currently in review (e.g., Beets and Netherland 2017a) or in preparation for publication (e.g. Beets and Netherland, 2017b, Netherland *et al.* 2017, Haug *et al.* 2017). These, along with the continued use of Procellacor™ under a variety of plant management scenarios, will add valuable information that can be incorporated into the product labels, improved treatment profiles and potentially required mitigation measures.

4.3.3 Environmental and Human Health Impacts

4.3.3.1 Earth

Soil and Sediments

Procellacor™ has moderately high measured K_{ow} and K_{oc} partition coefficients, with log K_{ow} and K_{oc} values of approximately 5.4 to 5.5, or about 10^{-5}, which supports low solubility and demonstrates a relatively high affinity for sorption to organically enriched substrates such as soils or sediments. However, as noted above, in aerobic soil Procellacor™ degrades quickly, with half-lives ranging from 2.5 to 34 days, with an average of 15 days. Anaerobic soil metabolism studies are similar, showing relatively rapid degradation rates with half-lives ranging from 7 to 15 days, and an average of 9.8 days. This rapid degradation in the soil and sediment environment strongly suggests low persistence in these media. Due to the low acute and chronic toxicity described below, low to negligible impacts are expected in soils and sediments adjoining Procellacor™ treatment areas. The herbicide can be classified as largely immobile based on soil log K_{oc} values in the order of 10^{-5}, and that potential for off-site transport would be minimal.

Agriculture

At anticipated use concentrations, irrigation or flooding of crops with water treated with Procellacor™ are not expected to damage crops or non-target wild plants, except under scenarios not addressed in the forthcoming EPA label.

Terrestrial Land Use

At anticipated use concentrations, water reentry or swimming in water treated with Procellacor™ is not expected to cause dermal, eye, or other irritation or toxicity to human or wildlife species.

4.3.3.2 Water

Surface Water and Runoff

Procellacor™ is known to have low water solubility (about 15 µg/L in lab testing) and the parent compound is not persistent and is known to quickly degrade via a number of well-established pathways. As discussed above, the herbicide is short lived in aerobic and anaerobic aquatic environments in a total water-sediment system. When exposed to direct sunlight, degradation in surface water is even more accelerated, with a reported photolytic half-life as little as 0.1 days.

The two outdoor aquatic dissipation studies summarized above further support this rapid dissipation and low impact. Both studies show that when Procellacor™ was directly injected into outdoor freshwater ponds at nominal rates of 50 and 150 µg/L, very rapid water-phase dissipation half-lives (3 to 4.9 days) were observed. These characteristics strongly suggest that the potential for off-site transport or mobility is minimal. As noted above, Procellacor™ undergoes rapid degradation in both soil and aqueous-phase environments via a number of degradation mechanisms.

No use for aquatic vegetation management in marine or estuarine water using Procellacor™ will be labeled at this time in Washington State (Heilman, 2016).

No specific studies or exposure scenarios were identified where drift or runoff were specifically investigated, but the forthcoming EPA risk assessment for Procellacor™ is expected to address these scenarios. For drift, the low vapor pressure (approximately 10^{-7} mm Hg) indicates that the material is not prone to volatilize following application, thus minimizing drift potential, and the low water solubility, low acute and chronic toxicity, along with minimal potential for persistence suggest that potential hazards associated with surface water runoff would be minimal.

Groundwater and Public Water Supplies

Few studies have yet been completed for groundwater, but based on known environmental properties concerning mobility, solubility, and persistence, Procellacor™ is not expected to be associated with potential environmental impacts or problems in groundwater.

In laboratory aquatic ecotoxicity studies, the highest concentration of TGAI that could be dissolved in the test water (or functional solubility) was approximately 40-60 µg/L in freshwater and 20-40 µg/L in saltwater. This is due to the low water solubility of the active ingredient and limits the range for which these toxicity tests can be conducted. This finding suggests that the water chemistry of Procellacor™ would limit potential environmental impacts to groundwater or surface water.

Impacts to public water supplies are expected to be low to negligible based on the low solubility, low persistence, and low acute and chronic toxicity of Procellacor™. Section 4.3.4 discusses possible measures or best management practices (BMPs) that could be used to further reduce potential impacts to public water supplies. The Ecology permit has mitigation that requires permittees to obtain an approval letter for this treatment prior to obtaining coverage under the permit.

4.3.3.3 Wetlands

The habitat and aquatic structure found in rice paddies is similar to those in a wetland and marsh environments, making the studies reported by Heilman (2016a) and Netherland and Richardson (2016) important tools for this analysis. The wetland and marsh study, discussed above in Section 4.3.2.2., incorporated appropriate water management practices for both wet-seeded and dry-seeded rice, and reported rapid aquatic-phase half-lives ranging from 0.15 to 0.79 days, and soil phase half-lives were also rapid, ranging from less than 0.01 to 8.1 days.

4.3.3.4 Plants

Algae

Limited ecotoxicity testing using a growth endpoint was conducted for two species of freshwater algae, including a diatom and green algae. These tests showed EC_{50} values using the TGAI of greater than 40 and 34 µg/L, respectively (solubility limit of assays). These results indicate that Procellacor™ is generally not toxic to green algae, freshwater diatoms, or blue-green algae at the anticipated label rate. Metabolite testing showed little toxicity to these algae, with no EC_{50} value less than 450 µg/L. Comparable growth testing was also conducted using the end-use formulation for aquatic algal plant growth, and results showed an EC_{50} greater than 1,800 µg/L (480 µg/L as active), with a NOAEC of 420 µg/L of formulation (111 µg/L as active), again showing a lack of toxicity to algae within anticipated label use rates. A comparable test of the TGAI was performed for cyanobacteria (blue-green algae), and results showed an EC_{50} of greater than 45 µg/L, with a calculated NOAEC value of 23.3 µg/L, showing little evidence of toxicity for any of these species.

Higher Plants and Crops

Procellacor™ is known to have strong herbicidal activity on key target aquatic invasive species, and testing shows that many native plants are able to tolerate Procellacor™ at exposure rates greater than what is necessary to control key target invasives. Data collection is still underway for specific toxicity to non-target plant species. Initial results of a 2016 collaborative mesocosm study conducted in Texas, for which results will be formally available later in 2017 indicate favorable selectivity by Procellacor™ of multiple invasive watermilfoils in the presence of representative submersed aquatic native plants (Netherland *et al.* 2017 *in prep*). Aquatic native plants challenged in this study included tapegrass, Illinois pondweed, American pondweed, waterweed, and water stargrass. Using aboveground biomass as a response endpoint, no significant treatment effects were observed with tapegrass or American/Illinois pondweed. Similarly, no statistically significant treatment effects were observed with stargrass, although injuries were observed at higher rates and exposures, although it was much more tolerant than the two target milfoil species. Other mesocosm studies have shown similar responses in white water lily with other non-target species including Robbins pondweed, American pondweed, and multiple bladderwort species showing little or no discernible impact. Richardson *et al*. (2016) and Haug and Richardson (2017 *in prep*) report that Procellacor™ provides a new potential for selectivity for removing hydrilla from mixed aquatic-plant communities. They recommend that further research should be conducted to further characterize observed patterns of selectivity.

4.3.3.5 Habitat

Impacts to critical habitat for aquatic plant or animal species are expected to be minimal, and may benefit critical habitat overall by supporting plant selectivity. Procellacor™ is generally of a low order or acute and chronic toxicity to plants and animals and generally does not persist in the environment. Due to its documented selectivity, Procellacor™ would allow many native non-target plants to thrive and thus enhance quality habitat. Removing noxious aquatic plants creates open spaces in the littoral zone that may be recolonized by not only native plants but other invasive plant species.

For example, when left unchecked, dense stands of unwanted weeds such as watermilfoil, parrotsfeather, hydrilla, or numerous other noxious plant species can negatively impact critical salmonid or other habitat used at all life stages, as well as habitats to a wide variety of plant and animal species, including vulnerable life stages. Stands of invasive weeds can reduce water flow and circulation, thus impeding navigation for migrant salmonids. Such stands can also provide ambush cover for predatory species such as bass, which prey on critical juvenile and other salmonid life stages. Moreover, noxious plants may outcompete native plant species, thus reducing overall biodiversity and reducing overall habitat quality. Dense stands may also be conducive to creating warmer water (through reduced circulation and dissolved oxygen sags), and could become subject to wide fluctuations in water quality (e.g. temperature, dissolved oxygen (DO)) on a diurnal/seasonal basis.

4.3.4 Mitigation

4.3.4.1 Use Restrictions

Procellacor™ should only be used for the control of aquatic plants in accordance with label specifications. No data gaps have been identified for the basic environmental profile required by EPA for pesticide registration, although continued use of Procellacor™ under a variety of plant management scenarios will add valuable information that can be incorporated into improved treatment profiles and possible mitigation measures. For potential future irrigation with Procellacor™-treated water, final EPA labeling will include guidance on appropriate water use. Such restrictions can be refined once the human health and ecological risk assessment currently being conducted by EPA are released in spring 2017. The proposed label language is expected to reflect fewer application-related restrictions than other herbicides. Lower levels of personal protective equipment (PPE) for workers will be required, which is consistent with lower use rates, lower water use restrictions, and minimal effects to crops or other non-target species.

4.3.4.2 Swimming and Skiing

Recreation activities such as swimming, water skiing and boating are expected to be unaffected by applications or treatments using Procellacor™ herbicide formulations.

4.3.4.3 Irrigation, Drinking and other Domestic Water Uses

Ecology's Aquatic Plant and Algae permit provides specific mitigation measures for irrigation water and water rights. Following registration, however, no water use restrictions are anticipated for the product use label except for some forms of irrigation. Any such restrictions will be specified on the final label language in collaboration with EPA.

Drinking water is not expected to be affected by Procellacor™ applications.

4.3.4.4 Fisheries and Fish Consumption

Neither fisheries nor human fish consumption are expected to be affected by application of Procellacor™ herbicides. If there is potential to impact listed salmonid species (e.g. salmon, steelhead, bull trout, etc.) Ecology would enforce a fish timing window that would be protective of those species. Guidance for such timing windows are found at: http://www.ecy.wa.gov/programs/wq/pesticides/final_pesticide_permits/aquatic_plants/permitdocs/wdfwtiming.pdf.

4.3.4.5 Endangered Species

Data are limited for specific listed threatened or endangered species under the ESA, however, a number of carefully designed and relevant laboratory toxicity tests for endangered species are currently under way, as discussed above. These tests will increase available testing data and enhance our understanding of how to more effectively protect non-target listed and vulnerable species, with particular emphasis on ESA-listed salmonid species such as salmon species, steelhead, and bull trout.

4.3.4.6 Wetlands or Non-Target Plants

Ecology's APAM permit outlines specific restrictions on what can be treated in wetlands. For example, in identified wetlands, the APAM specifies that the permittee "may treat only *high use areas* to provide for safe *recreation* (e.g., *defined swimming corridors*) and boating (e.g., *defined navigation channels*) in *identified and/or emergent wetlands.* The permittee must also limit the treated area to protect native wetland vegetation. However, final mitigation measures and best management practices concerning potential effects to beneficial or desirable wetland plant species will be developed in conjunction with testing on higher plants, some of which may occur in wetlands.

In general, effects to wetlands are anticipated to be minimal. Toxicity to fish, invertebrates, wildlife, and non-target plants would not generally be expected, and persistence (and thus food chain effects) would also be minimal. No specific toxicity testing was required or conducted for amphibians or reptiles which are ubiquitous in wetlands, but test results from invertebrate, avian, mammalian and other test species would be expected to serve as representative surrogate species for amphibians and reptiles.

Regarding potential impacts to rare or endangered plants occurring in wetlands, Ecology uses the Washington Department of Natural Resources (WDNR) Natural Heritage Site guidelines to determine if rare plants are likely to occur in the treatment area. If rare plants may be present at the treatment site, Ecology would require a field survey, and if such plants are found mitigation would be required.

4.3.4.7 Post-treatment Monitoring

EPA, Ecology, and other agencies routinely require both short- and long-term post-treatment monitoring for the purpose of evaluating non-target effects from herbicides such as Procellacor™. For Ecology, this post-treatment monitoring would be required under the permit, and would be a permit condition requiring monitoring to determine potential non-target impacts. These requirements will be incorporated into both label and permit, as appropriate, in conjunction with pesticide registration prior to application.

4.3.5 References

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