A

APPENDIX A

2008-2018 Aquatic Plant Frequencies

	Scientific Name	Common Name	LFOO (%)									
			2008	2009	2010	2011	2013	2014	2015	2016	2017	2018
Dicots	*Myriophyllum spicatum*	Eurasian watermilfoil	20.7	10.8	0.0	0.3	3.2	11.9	37.7	4.4	14.3	40.1
	Ceratophyllum demersum	Coontail	23.4	22.0	21.0	22.0	14.5	14.2	9.2	11.9	18.6	15.1
	Myriophyllum sibiricum	Northern watermilfoil	28.3	12.9	0.0	0.0	1.9	8.6	11.8	5.1	4.2	7.2
	Bidens beckii	Water marigold	14.1	6.3	0.6	2.3	3.9	6.6	6.6	2.7	2.0	7.2
	Myriophyllum alterniflorum	Alternate-flowered watermilfoil	3.6	4.9	0.6	1.0	1.0	1.0	5.2	3.4	5.9	8.9
	Ranunculus aquatilis	White water crowfoot	0.7	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.7	0.0
	Myriophyllum tenellum	Dwarf watermilfoil	1.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3
Non-dicots	*Vallisneria americana*	Wild celery	60.9	45.6	53.7	58.2	37.3	36.8	43.6	47.8	45.0	33.2
	Potamogeton gramineus	Variable-leaf pondweed	46.7	49.1	37.5	40.5	46.6	52.6	38.4	46.1	44.3	29.6
	Najas flexilis	Slender naiad	33.2	33.1	17.2	43.4	14.8	25.8	33.4	27.8	40.7	35.2
	Potamogeton robbinsii	Fern-leaf pondweed	31.3	34.5	33.7	31.6	30.9	30.8	24.9	26.1	25.7	16.4
	Chara spp.	Muskgrasses	29.3	18.1	22.3	25.3	14.5	16.9	32.1	40.3	40.7	13.8
	Elodea canadensis	Common waterweed	24.7	27.9	15.5	31.3	15.1	12.9	14.1	24.4	34.5	10.5
	Potamogeton zosteriformis	Flat-stem pondweed	31.3	27.5	5.2	16.1	26.0	20.9	12.5	7.5	10.4	10.5
	Potamogeton richardsonii	Clasping-leaf pondweed	11.5	18.5	16.8	18.1	10.9	8.9	7.9	13.2	21.5	11.2
	Heteranthera dubia	Water stargrass	22.4	9.4	4.2	8.6	9.6	13.6	16.1	10.2	11.7	9.2
	Potamogeton praelongus	White-stem pondweed	10.5	10.5	10.4	7.6	10.3	5.6	7.9	10.2	9.8	8.2
	Potamogeton pusillus, P. strictifolious, P. freissi	Thin-leaved pondweeds	21.4	16.0	3.2	4.6	10.9	8.9	4.6	3.1	2.6	7.2
	Potamogeton pusillus	Small pondweed	18.4	13.2	2.6	1.6	10.3	8.9	4.3	2.0	2.0	6.6
	Eleocharis acicularis	Needle spikerush	5.9	4.2	5.5	1.6	3.2	7.0	9.8	4.4	5.2	6.3
	Isoetes spp.	Quillwort spp.	3.9	2.8	5.5	3.3	0.6	1.3	6.9	6.1	10.4	8.6
	Nitella spp.	Stoneworts	2.3	2.8	0.6	2.6	0.0	0.7	0.0	2.0	7.5	5.3
	Potamogeton amplifolius, P.amp x praelongus	Large-leaf pondweed & Hybrid pondweed	0.7	0.3	1.6	0.7	1.9	3.0	0.0	2.4	4.2	3.6
	Potamogeton amplifolius	Large-leaf pondweed	3.6	3.8	0.6	0.7	1.0	0.3	0.0	2.4	2.3	0.3
	Potamogeton hybrid 1	Pondweed Hybrid 1	0.7	0.3	1.0	0.0	1.9	2.6	0.0	0.0	2.0	3.3
	Potamogeton friesii	Fries' pondweed	9.2	3.5	0.6	0.3	0.0	0.0	0.0	0.0	0.0	0.7
	Potamogeton strictifolius	Stiff pondweed	0.0	0.3	0.3	2.6	0.6	0.0	0.7	1.0	0.7	0.0
	Potamogeton hybrid 2	Pondweed Hybrid 2	0.0	0.0	0.0	0.0	0.0	3.6	0.0	0.0	0.0	0.0
	Sagittaria sp. (rosette)	Arrowhead sp. (rosette)	2.0	0.7	0.0	0.0	0.3	0.3	0.0	0.0	0.0	0.0
	Juncus pelocarpus	Brown-fruited rush	0.0	0.0	0.3	1.6	0.0	0.0	0.7	0.3	0.0	0.0
	Sagittaria graminea	Grass-leaved arrowhead	1.3	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Stuckenia pectinata	Sago pondweed	0.0	0.0	0.3	0.0	0.0	0.0	0.3	0.0	0.0	0.0
	Potamogeton illinoensis	Illinois pondweed	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.3	0.0	0.0
	Eriocaulon aquaticum	Pipewort	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0
	Elatine minima	Waterwort	0.0	0.3	0.0	0.0	0.0	0.0	0.0	0.0	0.0	0.0

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

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

B

APPENDIX B

<u>Fluridone Materials</u>

- **WDNR Chemical Fact Sheet on fluridone**
- **SePRO Sonar Risk Assessment**

January 2012

Fluridone Chemical Fact Sheet

Formulations

Fluridone is an aquatic herbicide that was initially registered with the EPA in 1986. The active ingredient is 1-methyl-3-phenyl-5-3-(trifluoromethyl)phenyl|-41H|-pyridinone. Both liquid and slow-release granular formulations are available. Fluridone is sold under the brand names Avast!, Sonar, and Whitecap (product names are provided solely for your reference and should not be considered endorsements).

Aquatic Use and Considerations

Fluridone is an herbicide that stops the plant from making a protective pigment that keeps chlorophyll from breaking down in the sun. Treated plants will turn white or pink at the growing tips after a week and will die in one to two months after treatment as it is unable to make food for itself. It is only effective if plants are growing at the time of treatment.

Fluridone is used at very low concentrations, but a very long contact time is required (45-90 days). If the fluridone is removed before the plants die, they will once again be able to produce chlorophyll and grow.

Fluridone moves rapidly through water, so it is usually applied as a whole-lake treatment to an entire waterbody or basin. There are pellet slow-release formulations that may be used as spot treatments, but the efficacy of this is undetermined. Fluridone has been applied to rivers through a drip system to maintain the concentration for the required contact time.

Plants vary in their susceptibility to fluridone, so typically some species will not be affected even though the entire waterbody is treated.

Plants have been shown to develop resistance to repeated fluridone use, so it is recommended to rotate herbicides with different modes of action when using fluridone as a control.

Fluridone is effective at treating the invasive Eurasian watermilfoil (*Myriophyllum spicatum*). It also is commonly used for control of invasive hydrilla (*Hydrilla verticillata*) and water hyacinth (*Eichhornia crassipes*), neither of which are present in Wisconsin yet. Desirable native species that are usually affected at concentrations used to treat the invasives include native milfoils, coontail (*Ceratophyllum demersum*), naiads (*Najas* spp.), elodea (*Elodea canadensis*) and duckweeds (*Lemna* spp.). Lilies (*Nymphaea* spp. and *Nuphar* spp.) and bladderworts (Utricularia spp.) also can be affected.

Post-Treatment Water Use Restrictions

There are no restrictions on swimming, eating fish from treated water bodies, human drinking water or pet/livestock drinking water. Depending on the type of waterbody treated and the type of plant being watered, irrigation restrictions may apply for up to 30 days. Certain plants, such as tomatoes and peppers and newly seeded lawn, should not be watered with treated water until the concentration is less than 5 parts per billion (ppb).

Herbicide Degradation, Persistence and Trace Contaminants

The half-life of fluridone (the time it takes for half of the active ingredient to degrade) ranges from 4 to 97 days depending on water conditions. After treatment, the fluridone concentration in the water is reduced through dilution due to water movement, uptake by plants, adsorption to the sediments, and break down from light and microbial action.

There are two major degradation products from fluridone: n-methyl formamide (NMF) and 3-trifluoromethyl benzoic acid. NMF has not been detected in studies of field conditions, including those at the maximum label rate.

Fluridone residues in sediments reach a maximum in one to four weeks after treatment and decline in four months to a year depending on environmental conditions. Fluridone adsorbs to clay and soils with high organic matter, especially in pellet form, and can reduce the concentration of fluridone in the water. Adsorption to the sediments is reversible; fluridone gradually dissipates back into the water where it is subject to chemical breakdown.

Impacts on Fish and Other Aquatic Organisms

Fluridone does not appear to have any apparent short-term or long-term effects on fish at application rates.

Fish exposed to water treated with fluridone absorb fluridone into their tissues. Residues of fluridone in fish decrease as the herbicide disappears from the water. The EPA has established a tolerance for fluridone residues in fish of 0.5 parts per million (ppm).

Studies on Fluridone's effects on aquatic invertebrates (i.e. midge and water flea) have shown increased mortality at label application rates.

Studies on birds indicate that fluridone would not pose an acute or chronic risk to birds. No studies have been conducted on amphibians or reptiles.

Human Health

The risk of acute exposure to fluridone would be primarily to chemical applicators. The acute toxicity risk from oral and inhalation routes is minimal. Concentrated fluridone may cause some eye or skin irritation. No personal protective equipment is required on the label to mix or apply fluridone.

Fluridone does not show evidence of causing birth defects, reproductive toxicity, or genetic mutations in mammals tested. It is not considered to be carcinogenic nor does it impair immune or endocrine function.

There is some evidence that the degradation product NMF causes birth defects. However, since NMF has only been detected in the lab and not following actual fluridone treatments, the manufacturer and EPA have indicated that fluridone use should not result in NMF concentrations that would adversely affect the health of water users. In the re-registration assessment that is currently underway for fluridone, the EPA has requested additional studies on both NMF and 3-trifluoromethyl benzoic acid.

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/

Wisconsin Department of Health Services
http://www.dhs.wisconsin.gov/

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

Hamelink, J.L., D.R. Buckler, F.L. Mayer, D.U. Palawski, and H.O. Sanders. 1986. Toxicity of Fluridone to Aquatic Invertebrates and Fish. Environmental Toxicology and Chemistry 5:87-94.

Fluridone ecological risk assessment by the Bureau of Land Management, Reno Nevada: http://www.blm.gov/pgdata/etc/medialib/blm/wo/Planning_and_Renewable_Resources/veis.Par.91082.File.tmp/Fluridone%20Ecological%20Risk%20Assessment.pdf

SONAR*

An Effective Herbicide That Poses Negligible Risk To Human Health And The Environment

Sonar is a highly effective aquatic herbicide used to selectively manage undesirable aquatic vegetation in freshwater ponds, lakes, reservoirs, rivers and canals. Sonar is absorbed through the leaves, shoots, and roots of susceptible plants, and destroys the plant by interfering with its ability to make and use food. As with any substance introduced into the environment, concerns arise about possible harmful effects on humans who may come into contact with it, and about its effects on wildlife and plants that we wish to protect and preserve. The following discussion, presented in a "Question and Answer" format, provides information regarding Sonar and evidence that Sonar presents negligible risk[1] to human health and the environment when applied according to its legally allowed uses and label directions.

Q1. What are the legally approved uses of Sonar?

A1. Sonar has been approved for use by the U.S. Environmental Protection Agency (USEPA) since 1986 for the management of aquatic vegetation in freshwater ponds, lakes, reservoirs, drainage canals, irrigation canals and rivers. Four different formulations have been approved for use—an aqueous suspension known as Sonar A.S. (USEPA Registration Number 67690-4) and three pellet forms known as Sonar SRP (USEPA Registration Number 67690-3), Sonar PR Precision Release (USEPA Registration Number 67690-12), and Sonar Q Quick Release (USEPA Registration Number 67690-3). There are no USEPA restrictions on the use of Sonar-treated water for swimming or fishing when used according to label directions. The Agency has approved Sonar's application in water used for drinking as long as residue levels do not exceed 0.15 parts per million (ppm) or 150 part per billion (ppb). For reference, one (1) ppm can be considered equivalent to roughly one second in 12 days or one foot in 200 miles, and (0.1) ppm can be considered approximately equal to one second in 120 days or one foot in 2,000 miles.

Sonar's USEPA-approved labeling states that in lakes and reservoirs that serve as drinking water sources, Sonar applications can be made up to within one-fourth mile (1,320 feet) of a potable water intake. For the control of Eurasian watermilfoil, curlyleaf pondweed and hydrilla where treatment concentrations are 0.01 to 0.02 ppm (10 to 20 ppb), this setback distance of one-fourth mile from a potable water intake is not required. Note that these effective treatment concentrations are well below the 0.15 ppm (150 ppb) allowable limit in water used for drinking.

Local public agencies may require permits for use of an herbicide in public waters. Therefore, the Sonar label states that the user must consult appropriate state or local water authorities before applying the herbicide.

[1]Throughout this document, we use the phrases "negligible risk" or "no significant risk." We use these terms because it is beyond the capabilities of science to prove that a substance is absolutely safe, i.e., that the substance poses no risk whatsoever. Any substances, be it aspirin, table salt, caffeine, or household cleaning products, will cause adverse health effects at sufficiently high doses. Normal exposures to such substances in our daily lives, however, are well below those associated with adverse health effects. At

some exposure, risks are so small that, for all practical purposes, no risk exists. We consider such risks to be negligible or insignificant.

*Trademark of SePRO Corporation

Q2. How does a product such as Sonar gain approval for use? (How does it become registered?)

A2. Federal law requires that an aquatic herbicide be registered with the USEPA before it can be shipped or sold in the United States. To obtain registration, manufacturers are required to conduct numerous studies (i.e., over 120 studies depending upon the intended uses) and to submit a thorough and extensive data set to USEPA to demonstrate that, under its conditions of use, the product will not pose a significant risk to human health and the environment and that the herbicide is effective against the target weeds or plants.

Individual states can establish registration standards that are more strict than federal standards, but not less strict.

Q3. What types of information must be submitted to regulatory agencies before an herbicide is registered?

A3. To register a herbicide, the manufacturer must submit information that falls into the following categories: product chemistry (for example, solubility, volatility, flammability and impurities), environmental fate (for example, how the substance degrades in the environment), mammalian toxicology (studies in laboratory animals used to assess potential health risks to humans), and wildlife and aquatic (for example, bird and fish) toxicology. If there are any residues in the environment, their levels must be determined. A manufacturer also conducts studies of product performance (or efficacy as a herbicide).

Q4. Have all of the data required for registration of Sonar been submitted to regulatory agencies, and have those agencies found the data acceptable?

A4. The data required for registration of Sonar by the USEPA is complete and has been accepted by the USEPA and by all states.

Q5. What happens to Sonar when it is used according to approved labeling -- that is, what is its environmental fate or what happens to Sonar once it is released or applied to the water?

A5. Tests under field conditions show that Sonar disappears from treated water in a matter of weeks or months, depending on a number of environmental factors such as sunlight, water temperature and depth. In lakes, reservoirs, rivers and canals where only a portion of the water body is treated, dilution reduces the level of Sonar relatively quickly following application.

Sonar does not persist in the environment. Its disappearance from aquatic environments is accomplished by several processes. First, the plants that are being treated absorb Sonar, thereby removing a portion of it from the water. Second, Sonar degrades or breaks down in the presence of sunlight by a process called "photo degradation." Photo degradation is the primary process contributing to the loss of Sonar from water. Third, adsorption of Sonar to hydrosoil (sediments) also contributes to its loss from water. As Sonar is released from hydrosoil back into the water, it is photo degraded.

Study results indicate that Sonar has a low bioaccumulation potential and therefore is not a threat to the food chain. Specifically, studies have shown that Sonar does not accumulate in fish tissue to any significant degree. The relatively small amounts of Sonar that may be taken up by fish following application are eliminated as the Sonar levels in water decline. In a study of crops irrigated with Sonar treated water, no residues of Sonar were found in any human food crops, and only very low levels were detected in certain forage crops. Consumption by livestock of Sonar-treated water and crops irrigated with Sonar-treated water was shown to result in negligible levels of Sonar in lean meat and milk. Sonar-treated water can be used immediately for watering livestock.

To ensure that residue levels of Sonar pose no significant risk, USEPA has established tolerances, or maximum legally allowable levels, in water, fish, and crops irrigated with Sonar-treated water, and other agricultural products (including eggs, milk, meat, and chicken). For example, the 0.15 ppm (150 ppb) concentration in water mentioned in the answer to Question #1 is the tolerance limit for water that is used for drinking. The recommended application rates of Sonar (detailed on the label) are established to ensure the product will do its job and that tolerance limits won't be exceeded.

Q6. How might people come into contact with Sonar after it is applied to an aquatic site?

A6. People could come into contact with Sonar by swimming in water bodies treated with the herbicide, by drinking water from treated lakes or reservoirs, by consuming game fish taken from treated waters, and by consuming meat, poultry, eggs or milk from livestock that were provided water from treated surface water sources.

Q7. Is it likely that people will be harmed because of those contacts?

A7. Extensive studies have demonstrated that contact with Sonar poses negligible health risks when the herbicide is used according to label instructions. The label for Sonar carries no restrictions for swimming or fishing in treated water or against drinking water treated with Sonar. Sonar does not build up in the body.

The conclusion that Sonar poses negligible health risks is evidenced by USEPA's toxicity rating for Sonar. The USEPA classifies herbicides according to their acute toxicity or potential adverse health effects and requires that a "signal word" indicating the relative toxicity of the herbicide be prominently displayed on the product label. Every herbicide carries such a signal word. The most acutely toxic herbicide category requires the signal word DANGER. However, if the product is especially toxic, the additional word POISON is displayed. Herbicides of moderate acute toxicity require the signal word WARNING. The least toxic products require the signal word CAUTION. Sonar labels display the word CAUTION, the USEPA's lowest acute toxicity rating category.

Q8. How do we know that humans are not likely to experience any harmful effects from Sonar's temporary presence in the environment?

A8. Companies that develop new herbicides are required to: 1) conduct extensive investigations of the toxicology of their product in laboratory animals; 2) characterize the ways by which people may contact the herbicide after it has been applied to an aquatic site; 3) determine the amount of exposure resulting from these possible contacts; and 4) demonstrate the fate of the herbicide in the environment. Before USEPA will register a herbicide, the Agency must establish with a high degree of certainty that an ample safety margin exists between the level to which people may be exposed and the level at which adverse effects have been observed in the toxicology studies.

Investigations of the toxicity of Sonar have been performed in laboratory animals under a variety of exposure conditions, including exposure to very high doses for short periods (acute studies), as well as repeated exposures to lower doses (which are still far in excess of any exposures that humans might actually receive) throughout the lifetime of the laboratory animals (chronic studies). Other special studies have been performed to evaluate the potential for Sonar to cause reproductive effects, cancer, and genetic damage. Study results indicate a low order of toxicity to mammalian species following acute exposures and repeat-dose exposures for up to a lifetime. In addition, repeated doses of Sonar did not result in the development of tumors, adverse effects on reproduction or on development of offspring, or genetic damage.

In characterizing the toxicity of a compound and its safety margin for exposures of humans and wildlife, toxicologists attempt to identify the maximum dose at which a chemical produces no toxicity. Another way of stating this is how much of the chemical can an organism be exposed to before it reaches a toxic level (recall from the footnote to the introduction on page 1 that all substances are toxic at some dose or level). This maximum non-toxic dose is usually established by studies in laboratory animals and is reported as the "no-observed-effect level" or NOEL. The dietary NOEL for Sonar (that is, the highest dose at which no adverse effects were observed in laboratory animals fed Sonar) is approximately 8 milligrams of Sonar per kilogram of body weight per day, abbreviated 8 mg/kg/day. This NOEL was derived from a study in rats that were fed Sonar in their regular diets every day for their entire two-year lifetime.

To put this NOEL into perspective, a 70-kg adult (about 150 pounds) would have to drink over 1,000 gallons of water containing the maximum legally allowable concentration of Sonar in potable water (0.15 ppm) daily for a significant portion of their lifetime to receive a dose equivalent to the 8 mg/kg/day NOEL. At most, adults drink about 2 quarts (one-half gallon) of water daily, which means that even if a person were drinking water with the maximum legally allowable concentration of Sonar, their margin of safety would still be at least 2,000. Similarly, a 20-kg child (about 40 pounds) would have to drink approximately 285 gallons of Sonar-treated water every day to receive a dose equivalent to the NOEL. Because children drink only about one quart of water daily, this provides a safety margin of greater than 1,000.

The above example calculation of safety margins is based on the assumption that potable water will contain levels of Sonar at its maximum allowable concentration of 0.15 ppm (150 ppb). In fact, the Sonar concentration achieved under typical applications is closer to 0.02 ppm (20 ppb), thereby providing a safety margin seven times greater. The point is that adults and children who drink water from potable water sources that have been treated with Sonar according to label instructions are at negligible risk.

Similarly, the levels of Sonar allowed in various food products pose negligible risk to human health. For example, even if Sonar were present at the maximum allowable limit of 0.05 ppm in meat, poultry, eggs, and milk, a 70-kg adult would have to consume almost 25,000 pounds of these foods daily (and again for a significant portion of a lifetime) to receive a dose equivalent to the dietary NOEL for Sonar. A child would have to consume over 7,000 pounds of these foods daily.

Because Sonar is used only intermittently in any one area, and because it disappears from the environment, there is virtually no way that anyone will be exposed continuously for a lifetime. Because the NOEL derives from a study involving daily exposures for a lifetime, the actual safety margin for people is, in fact, much greater than is suggested by the above illustrative examples.

Q9. How complete is the toxicology information upon which this conclusion rests?

A9. All toxicity studies required by the USEPA to obtain registration approval for Sonar have been completed.

Q10. What about the people who apply Sonar—are they at risk?

A10. The Sonar label states that individuals who use Sonar should avoid breathing spray mist or contact with skin, eyes, or clothing; should wash thoroughly with soap and water after handling; and should wash exposed clothing before reuse. These precautions are the minimum recommendations for the application of any pesticide. If Sonar is used according to label instructions, exposures to the product should be minimal and use should pose negligible risks to applicators.

Sonar has been shown to be of low acute toxicity in laboratory animal studies (that is, toxicity from a high dose exposure for a short period of time). Therefore, any exposure to the product (even undiluted) that might occur during use is unlikely to lead to adverse effects as long as label instructions are followed. As discussed in Question #7, Sonar's label carries the signal word CAUTION that corresponds to the USEPA's lowest acute toxicity rating category.

Studies in laboratory animals show that the lethal dose from a single oral exposure of Sonar is greater than 10,000 mg/kg. To put this into perspective, an adult would have to drink over one million gallons of Sonar-treated water (at the 0.15 [150 ppb] ppm maximum allowable limit) to receive a dose of 10,000 mg/kg; a 20-kg child would have to drink approximately 350,000 gallons.

Because applicators are more likely to contact the undiluted material than the general population, questions about the toxicity of Sonar following direct skin contact have been raised. A laboratory study of the toxicity of an 80 percent solution of Sonar applied to rabbit skin (a standard model to predict effects in humans) suggests that Sonar is minimally toxic by this route. In this study, when Sonar was repeatedly applied to the skin of rabbits for 21 days (in the largest amounts that could be applied practically), there were no signs of toxicity and only slight skin irritation was observed. Further, the dermal administration of the 80 percent solution of Sonar did not induce sensitization in guinea pigs.

Q11. Has there been any investigation of the possible harmful effects of Sonar on fish, wildlife, pets and livestock?

A11. The toxicity of Sonar has been investigated in laboratory studies in birds (including the bobwhite quail and mallard duck), in the honey bee (as a representative insect) and in the earthworm (as a representative soil organism), in five different species of freshwater and marine fish, and in other aquatic animals. These studies have involved exposures to high concentrations for brief periods as well as exposures lasting as long as an entire lifetime, including during reproduction.

Extensive studies have also been performed to evaluate the effects of Sonar on various aquatic and terrestrial plants (both those considered undesirable aquatic weeds and those native plants that we wish to protect). Studies in laboratory animals designed primarily to assess potential health risk in humans are also relevant to the assessment of potential health effects in mammalian wildlife, livestock, and pets.

In addition, **Sonar** has been monitored in water, plants and fish during field trials. This provides firsthand information on residue levels in the environment following application of Sonar.

Q12. What do these investigations reveal?

A12. A combination of the toxicity studies and residue monitoring data reveals that Sonar poses negligible risks to aquatic animals including fish, wildlife, pets, and livestock when used according to label directions.

As was done with laboratory mammals, toxicity studies were conducted to establish a dietary no-observed effect level (NOEL) for birds. This maximum, non-toxic chronic dose is 1,000 ppm in the diet. One thousand (1,000) ppm is 2,500 times the highest average concentration of total residue found in fish (0.40 ppm), about 2,100 times the highest concentration found in aquatic plants (0.47 ppm), and about 11,500 times the highest average concentration of Sonar found in the water at field trial sites (0.087 ppm). Because the residue levels in these “bird food” items are so far below the NOEL, it can be concluded is that there are negligible risks to birds that might be exposed to Sonar in their diet following application of Sonar.

The highest average Sonar concentration found in Sonar-treated water is below the lowest NOEL values for both short and long term exposures from freshwater and marine fish. Honeybees and earthworms are not particularly sensitive to Sonar. Sonar caused no deaths in honey bees when they were dusted directly with the herbicide, and earthworms were not affected when they were placed in soil containing more than 100 ppm Sonar.

Extensive testing of Sonar in laboratory animals used to assess potential risks to human health indicates that a large safety margin exists for mammalian species in general. Thus, Sonar poses negligible risk to pets, livestock, and mammalian wildlife that might drink from water treated with Sonar.

Q13. Can Sonar be used in environmentally sensitive areas?

A13. Sonar has been used in a wide range of aquatic environments in the United States without incident for almost 15 years. Florida canals and rivers are examples of environmentally sensitive areas that have been treated with Sonar. Some sites are habitats for the endangered Florida manatee. Although toxicity testing data for the manatee, or for other endangered species, cannot be collected directly, questions about whether Sonar treatment will pose any significant risk to the manatee can be answered with results of the mammalian toxicity studies.

The Florida manatee is an aquatic mammal that consumes up to 20% (one-fifth) of its body weight per day in aquatic plants. Treatment of canal water with Sonar according to label directions is expected to result in a maximum Sonar concentration of 0.15 ppm in the water and from 0.8 to 2.6 ppm in aquatic plants. Calculations show that it would be impossible for a manatee to ingest enough Sonar in its diet to cause any adverse effects, based on results of laboratory studies in other mammals. To reach the maximum non-toxic dose or NOEL for sensitive mammalian species, a manatee would have to drink more than 40 times its body weight per day in treated water, or eat at least 3 to 10 times its body weight per day in aquatic plants. This calculation indicates that treatment with Sonar in manatee habitats—as one example of an environmentally sensitive area—will pose negligible risk. In fact, application to Florida canals and rivers has been approved by the U.S. Fish and Wildlife Service, Florida Department of Environmental Protection, and the Florida Game and Fresh Water Fish Commission.

Sonar has also been used in other environmentally sensitive areas such as Disney World, Ducks Unlimited MARSH projects, Sea World, state and federal parks, and numerous fish and waterfowl management areas.

Q14. What is it that makes Sonar an effective aquatic herbicide while being a compound of relatively low toxicity to humans?

A14. Sonar inhibits a plant's ability to make food. Specifically, Sonar inhibits carotenoid synthesis, a process specific only to plants. Carotenoids (yellow, orange and red pigments) are an important part of the plant's photosynthetic (food making) system. These pigments protect the plant's green pigments (called chlorophyll) from photo degradation or breakdown by sunlight. When carotenoid synthesis is inhibited, the chlorophyll is gradually destroyed by sunlight. As a plant's chlorophyll decreases, so does its capacity to produce carbohydrates (its food source) through photosynthesis. Without the ability to produce carbohydrates, the plant dies.

Humans do not have carotenoid pigments. Therefore, the property of Sonar that makes it an effective herbicide at low doses does not affect the human body.

Q15. Will Sonar have an adverse effect on water quality?

A15. Extensive testing of a wide range of water bodies has shown no significant changes in water quality after Sonar treatment. In fact, Sonar has a practical advantage over certain other aquatic herbicides in this area. Specifically, the dissolved oxygen content of the water does not change significantly following Sonar treatment because the relatively slow herbicidal activity of the product permits a gradual decay of the treated vegetation. Maintaining adequate dissolved oxygen levels are critical to fish and other aquatic animals, which require oxygen to survive. This contrasts with the changes in water quality that can arise from the application of certain other aquatic herbicides that are “fast-acting.” The sudden addition of large amounts of decaying plant matter to the water body can lead to decreased oxygen levels and result in a fish kill. To avoid depressions in dissolved oxygen content, label directions for certain “fast-acting” aquatic herbicides recommend that only portions of areas of dense weeds be treated at a time. Because Sonar does not have any substantial impact on dissolved oxygen, it is possible to treat an entire water body with Sonar at one time.

Q16. Is there any reason for concern about the inert ingredients used in Sonar?

A16. Inert ingredients are those components of the product that do not exhibit herbicidal activity; that is, the components other than Sonar. Water is the primary inert ingredient in Sonar A.S., making up approximately 45% of the formulation. The second largest (approximately 10%) inert is propylene glycol; a compound used in facial creams and other health and beauty products. Other inert ingredients are added to serve as wetters, dispersants, and thickeners in the formulation. Trace amounts of an antifoaming agent and a preservative are also added. The primary inert ingredient in the pelleted formulations is clay, which makes up approximately 89% of the formulation. Small amounts of a binder or coating solution are also added to reduce the dustiness of the pellets. None of the inert ingredients in Sonar formulations are on the USEPA’s list of “Inerts of Toxicological Concern” or list of “Potentially Toxic Inerts/High Priority for Testing.” Thus, there is no reason for concern about the inert ingredients used in Sonar.

Q17. Is it important to follow label directions for use and disposal of Sonar?

A17. Yes. It is a violation of federal law to use products, including Sonar, in a manner inconsistent with product labeling or to improperly dispose of excess products or rinsate. Although the results of extensive toxicity testing in the laboratory and in field trials indicate a low order of toxicity to non-target plants, animals, and people, Sonar, like all chemicals, will cause adverse effects at sufficiently high exposure levels. Failure to follow label directions for use and disposal of Sonar could result in environmental levels that exceeds the tolerances for Sonar established to be protective of human health and the health of pets, livestock and other wildlife. In addition, improper use of Sonar could result in unintended damage to non-target plants.

Q18. If Sonar is used in conformance with label directions, is there any reason to be concerned that Sonar will pose risk to human health or the environment?

A18. As discussed in the answers to the previous questions, results of laboratory and field studies and extensive use experience with Sonar in a wide range of water bodies strongly support the conclusion that Sonar will pose negligible risks to human health and the environment when used in conformance with label directions.

In summary, it can be said that Sonar has a favorable toxicological profile for humans. It has an overall low relative toxicity and it is not a carcinogen, mutagen or reproductive toxicant. Sonar also has a very good environmental profile for an aquatic product because of: 1) its low toxicity to non-target organisms; 2) its non-persistent behavior when applied to water bodies (i.e., it readily breaks down to carbon, hydrogen, oxygen, nitrogen and fluorine); and 3) its low bioaccumulation potential, which means it does not build up in the body or in the food chain.

C

APPENDIX C

<u>ProcelleCOR™ Materials</u>

- **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

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™ (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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D

APPENDIX D

Agency Comments on November 15, 2018 Draft Report

- Onterra Response Comments to WDNR and GLIFWC Comments
- SePRO Response Comments to WDNR & GLIFWC
- Lake District & WDNR Response Comments to GLIFWC Comments
- Lake District Analysis of Native Plants

Technical Review Team Comments to North and South Twin Lake 2018 EWM Monitoring & Control Strategy Development Report
November 2018

Technical Review Team: Tom Aartila, Marsha Burzynski, Mary Gansberg, Steve Gilbert, Jodi Lepsch, Ali Mikulyuk, Ashleigh McCord, Michelle Nault, Scott Van Egeren, Ty Krajewski, Carol Warden, Sue Graham, Susan Knight, & Kevin Gauthier.

Response Comments by Eddie Heath, Onterra

- Several team members noted that the report does not mention mechanical harvesting as a potential management option for South Twin Lake and wanted to know if this mgmt. option has been seriously considered and discussed amongst stakeholders. Regional staff indicated that some verbal discussions were had on mechanical harvesting, but that this technique was not supported by the lake group at this time. It would be good to document in this report when and with whom these discussions occurred and what the outcomes were (i.e., why was this technique not supported) to provide some additional information that this management technique was indeed considered and critically evaluated alongside other control options (such as large-scale herbicide) which are discussed within the report. Additional elaboration of the goal development process from the *Comprehensive Management Planning Project* is now included in the report (Section 2.3).

- Certain native species (e.g., NWM, water marigold, water stargrass, etc.) are still currently below 'pre-treatment' (i.e., 2008) frequencies and there are concerns that another whole-lake herbicide treatment may further suppress these native species ability to recover. The fluridone treatment may affect different natives than impacted by 2,4-D. These statements are fact and discussed within the report. Lakewide suppression of natives following repeated large-scale herbicide treatments may provide a better opportunity for EWM to expand once it begins to recover following treatment, potentially exasperating the problem. This core ecological principal (empty niche hypotheses, ecological succession of pioneer species) is acknowledged. However, in a recent study of 1,100 Minnesota lakes, researchers concluded that more diverse communities were not more resistant or resilient to invaders (Muthukrishnan et al. 2018).

- The average number of natives per site was 4.2 species/site in 2008 and has ranged from 2.5-3.5 species/site since large-scale chemical management began to occur in South Twin Lake (a net reduction of 0.7-1.7 species/site). Statement of fact, explored in Figure 2.2-7.

- There are also concerns that repeated whole-lake herbicide treatments may impact other aquatic organisms, either directly (e.g., toxicity) or indirectly (e.g., habitat loss). Statement of fact, explored in Section 2.4.

- Tribes have asked that herbicide application occur after spearfishing concludes. However, fluridone is anticipated to still be present at detectable levels into spring 2020 and may overlap with spearfishing and early season spawning activities. Statement of fact, outlined in Figure 2.4-1 and explored in Section 2.5.

- Include a column in Table 2.3-1 for plants known to be affected by fluridone as listed on the herbicide product label. Label indicates that 20 of 27 species (~75%) found in South Twin could be affected by this treatment. Table column added. The max label rate of 150 ppb is much higher than an exposure of 2-3 ppb which is discussed here. For this reason, the report investigates the available case studies for dose-dependent responses to better understand the potential impacts.

- Much of the previous data on the use of pelletized fluridone is from presentations given at conferences (i.e., Heath et al. 2018a, 2018b). Are these available publicly or in another format? Much of the justification for use of fluridone in South Twin is based on the information from a few limited Wisconsin treatments within these reports and it would be beneficial to have wider access to this data. The WDNR has all the individual lake reports on file, as well as the presentations cited. The report contains some of the data presented within these citations (i.e. Figure 2.4-1, 2.4-2, 2.4-3).

- Native impacts based on previous studies (i.e., Heath et al. 2018a, 2018b) were stated, but it is not clear how these previously documented impacts were actually taken into consideration when deciding to pursue fluridone. There seems to be a disconnect between listing likely native impacts and then still recommending this particular treatment strategy. For clarity, the NSTLPRD contracted with Onterra to provide technical direction as they pursue their EWM management goals. The report includes technical direction, not recommendations. Need a more throughout analysis of pros and cons of this treatment approach; it seems that the lake group believes the pro of controlling EWM outweighs any potential cons to native plants that may be associated with this treatment. Perhaps consider incorporating native plant 'goals' in addition to EWM? [and more than just a generic 'strive to protect native communities' blanket statement; for hypothetical example: a whole-lake treatment will not occur if native plant populations [and/or species X] have been reduced to more than Y% of pre-treatment levels; this will allow for consideration of both EWM goals as well as native plant protection]. We look forward to working with the WDNR in attempt to incorporate such metrics into future lake management plans.

- Data from previous pelletized fluridone applications have been for HWM control. The efficacy of low dose pelletized fluridone on pure EWM is not known (but likely to be similar?). Yes, see Dr. Heilman's comments (Appendix D).

- "*Overall the selectivity of the pelletized fluridone strategy appears to be better than the liquid case studies.*" (p.12). This generalization is biased based upon the data presented. The liquid case studies (compiled by DNR SS) were at relatively higher application rates [i.e., 10+ ppb or 6-bump-6] than more current pelletized fluridone applications [i.e., 2-4 ppb] and so making a comparative statement on the selectivity of these two formulations is not an 'apples to apples' comparison since the liquid applications were at ~2-3x the rate of the current pelletized applications, and that difference in application rate would have understandably impacted selectively (regardless of the formulation). Text added to increase clarity. We believe the data suggest a dose-dependent response for some plants. The pelletized formulation allows a concentration profile different that avoids high peaks that are often associated with liquid fluridone applications.

- Fluridone is a slow-acting herbicide and mgmt. 'success' may not necessarily be seen by lake riparians during the 2019 open water season as plants are slowly dying back. There is little to no discussion on the length of time it takes fluridone to kill the plants - is lake association aware that this product may not provide relief from nuisance conditions until after the 2019 season (i.e. year 2020)? Text added for clarity; see Dr. Heilman's comments (Appendix D). EWM impacts from Onterra-monitored pelletized fluridone treatments was after approximately 5 weeks with plant mortality being more obvious after 2 months. Concerns are noted that delayed implementation could delay when impacts are observed.

- Will dissolved oxygen be monitored following treatment? Are there concerns about lakewide plant die-back leading to low DO conditions? Text added for clarity, but typically this is a concern more-often associated with quicker-acting control methods.

- At this time, the Department does not fund any use or purchase of drones through surface water grants. This statement applies to the specific monitoring methodologies included within the AIS Grant applications.

- The review team recommends that a quantitative trigger (i.e., >12% EWM frequency) should be utilized to initiate discussion/consideration of <u>all</u> available management options, and not just a direct pathway for considering large-scale herbicide treatment. Group needs to take a more integrated management approach and should not rely solely on herbicides without critically looking at the data periodically over time. Views on herbicides (and other available management tools) may very well change over time as more data and resources becomes available. Team also feels strongly that given the previous documented frequencies of EWM on S. Twin Lake that a trigger of 12% littoral frequency is too low and very likely not sustainable to maintain over the long-term. These standards were developed as part of the lake management planning project. Onterra and the district acknowledge the WDNR's opinion on the trigger.
- Need consistency between mgmt. triggers listed in approved mgmt. plan and what's included in this EWM control report. Clarity added to make sure reader understands the trigger in all instances where it is referenced.

- Integrated pest management is barely mentioned in this report, except for one sentence which says that these activities will 'preferably' occur. Based on a lack of critical discussion on IPM, it does not seem that this component of the management strategy has been critically discussed and planned for, but rather is added in as an after-thought behind a very lengthy discussion on chemical control. A critical discussion of IPM following the large-scale fluridone treatment needs to added to this report. Include a discussion on why previous IPM didn't meet the lake association goals (too little time and effort?), and work to identify an IPM strategy that may be more successful than what was previously implemented (i.e. maybe consider building a DASH boat and/or hiring a full-time crew instead of hiring a company for only a few days per season). Extensive discussion of IPM is included within the *Comprehensive Management Plan* and was part of the overall planning project. The purpose of this report was to flesh out the short-term control strategy, with the long-term strategy being included within the management plan. Subsequent annual reports will provide more specific technical discussion on IPM as applicable.

- If fluridone does not met the EWM mgmt. goals defined by the lake associations, what is next? It is almost certain that EWM will rebound again following this fluridone treatment, as it has rebounded in all the other case studies presented (i.e. Heath 2018a, 2018b). Are non-chemical controls being considered in the future? This is discussed in the *Comprehensive Management Plan* both as it relates to the possibility of the fluridone treatment not meeting success criteria as well as within a specific action titled: *Investigate and Study Alternative Management Methodologies*.

- The team does not necessarily agree with statements that prior large-scale 2,4-D treatments have been 'ineffective'. These past treatments have resulted in lakewide control of EWM, and in the case of the 2010 treatment multi-year lakewide control of EWM without the need for additional large-scale management. If the group believe that an 'effective' treatment is only one where EWM never comes back and re-establishes, then we fear that they will never achieve that goal with the strategies and current efforts which they are implementing. If EWM almost inevitably comes back again (likely at a lakewide scale) following the fluridone treatment, will this treatment also be deemed 'ineffective' by the lake group in the near future? Need some clarity on what the lake group believes *is* actually an effective treatment outcome rather than just pointing out (after the fact) all the scenarios that they deem non-effective. This comment relates to a statement made within the District's cover letter.

- The team does not believe that there is currently any hard evidence that the EWM out on the Twin Lakes is developing resistance, and this statement is speculative. Observed lakewide EWM control in South Twin Lakes follows what would have been expected based on other EWM large-scale 2,4-D lakes we've monitored to date. Figure 2.1-1 suggest that the longevity of EWM control on South Twin Lake is NOT consistent with what is observed on other pure-strain EWM populations in the NLF ecoregion. The data more closely align with HWM populations. It is also possible to develop resistance by using fluridone, and there are fluridone-resistant watermilfoil strains which have been documented in neighboring Midwest states (i.e., Michigan). Only a single case of fluridone tolerance by invasive milfoil populations have been documented. The mechanism for herbicide tolerance (diquat and fluridone) by the Townline Lake HWM biotype is unknown and some suspect it is due to a single somatic gene replacement (which is documented in dioecious hydrilla) versus natural tolerance shift in response to previous treatment history. Will fluridone not be considered again in any potential future treatments for this same concern about potential development of resistance? The *Comprehensive Management Plan* states that herbicide use patterns may require rotation to avoid population-level herbicide tolerance evolution from occurring.

- Based on the morphology and sediment of North Twin Lake, it seems that EWM may very well not establish in the same way like it has in South Twin. EWM has been present in North Twin for at least 17 years and is still contained to a very localized area. It is not to say that EWM couldn't ever 'take off' in North Twin, as we know invasives can respond to environmental triggers after long-periods of low population levels. There seems to be limited scientific evidence for the lake group's fear that the EWM in North Twin is 'on track' to look like S. Twin. The *Comprehensive Management Plan* outlines the recently expanded footprint of the North Twin EWM population.

- Statements that ProcellaCOR: "*is the best herbicide because of its rapid absorption as well as selective impact on EWM*" is not supported by actual data which the review team is aware of. This comment relates to a statement made within the District's cover letter.

- Need to better understand the mechanism of why EWM is re-establishing on a lakewide scale following large-scale chemical mgmt. Are plants re-establishing from seeds, root stocks, vegetative materials, being re-introduced? This is a big question that needs to be answered, as if we don't understand the mechanism of EWM recovery then we cannot tailor additional IPM to target this mechanism before it spreads to a lakewide scale again. Onterra supports research directed towards this hypothesis.

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- Of the lakes which have used fluridone (i.e. Bughs, George, Cloverleaf, Silver) how many of these also do other IPM? It is important to note here all the mgmt. activities which these groups are currently implementing to not give the impression that they only used fluridone and no follow-up or other integrated management as well. Additional discussion on level of known IPM is added.

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- Figure 3.2-3 is confusing. Are black bars the data from within the targeted treatment areas over time, or is this lakewide PI data?? "During 2007 to 2013… point-intercept sub-sampling took place…" "black error bars represent range of annual frequencies during the time period."

- North Twin: Mid-October 2018 pre-treatment sub-PI data is very late in the season to capture all natives, many of which are likely senscing….why was this data collected so late in the season?? When will the post-treatment sub-PI data be collected? May want to consider scheduling the 2019 survey timing to best match up with pre-treatment data collection, or consider two sub-PI surveys (i.e. mid-summer and then mid-October). The logistics of identifying potential future control strategies, creating a sampling grid, and returning to the lake to collect the data are challenging and it is acknowledged that these data were collected rather late in the season. Finding a way to best compare the data is important.

- Diquat treatment on N. Twin in 2013 resulted in two years of reduced milfoil…why is this approach not being considered again? Diquat was considered, but "Concern exists whether this herbicide has the capacity to kill the entire plant, or simply impacts the above ground biomass and the plant rebounds from unaffected root crowns"

- ProcellaCOR: states "short exposure time scenarios". Can this be quantified. i.e. X hrs, X days? Dr. Mark Heilman addressed in his comments. Non-acid-based herbicides that have high KOC are difficult to evaluate CETs for, which is why a surrogate (dye) is often used for similar herbicides (diquat).

- On N. Twin, the main dominant native species within the treatment area is coontail, and NWM is also relatively dominant. Both of these natives are known to be susceptible to ProcellaCOR. Dr. Mark Heilman addressed in his comments. Will damaging these natives potentially create more room for EWM to eventually colonize within this area? Empty niche and disturbance principals addressed.

- What was PDU rate used on Silver Lake, Kenosha Co. How does this compare to the rate being recommended for N. Twin? Dr. Mark Heilman addressed in his comments.

SePRO Responses to North and South Twin DNR Tech Review and Great Lakes Indian Fish and Wildlife Commission comments on 2019 Proposed EWM Management

Responses are in blue and questions should be directed to Dr. Mark Heilman, Senior Aquatic Technology Leader, markh@sepro.com, 317-775-3309

North & South Twin Lakes, Vilas Co.

Lake Technical Review Team Comments

- Several team members noted that the report does not mention mechanical harvesting as a potential management option for South Twin Lake and wanted to know if this mgmt. option has been seriously considered and discussed amongst stakeholders. Regional staff indicated that some verbal discussions were had on mechanical harvesting, but that this technique was not supported by the lake group at this time. It would be good to document in this report when and with whom these discussions occurred and what the outcomes were (i.e., why was this technique not supported) to provide some additional information that this management technique was indeed considered and critically evaluated alongside other control options (such as large-scale herbicide) which are discussed within the report.
- Certain native species (e.g., NWM, water marigold, water stargrass, etc.) are still currently below 'pre-treatment' (i.e., 2008) frequencies and there are concerns that another whole-lake herbicide treatment may further suppress these native species ability to recover. The fluridone treatment may affect different natives than impacted by 2,4-D. Lakewide suppression of natives following repeated large-scale herbicide treatments may provide a better opportunity for EWM to expand once it begins to recover following treatment, potentially exasperating the problem.

Recent experiences with use of low-dose, pelletized Sonar® (fluridone) in both WI (Silver Lakes in Kenosha and Waushara County and several others) and MN (example: Crooked Lake MN – DNR presentation from Upper Midwest Invasive Species Conference attached) are documenting sustained selective control versus older, higher-rate fluridone uses (better selectivity with pellets) and many lake-wide 2,4-D treatments (longer control of invasive watermilfoil – particularly hybrid EWM). It is true that different native plants will show some sensitivity to this use pattern versus 2,4-D but the longer-term results show excellent restoration and longer-term maintenance of diverse aquatic plant communities with integrated follow-up strategies.

- The average number of natives per site was 4.2 species/site in 2008 and has ranged from 2.5-3.5 species/site since large-scale chemical management began to occur in South Twin Lake (a net reduction of 0.7-1.7 species/site).
- There are also concerns that repeated whole-lake herbicide treatments may impact other aquatic organisms, either directly (e.g., toxicity) or indirectly (e.g., habitat loss).

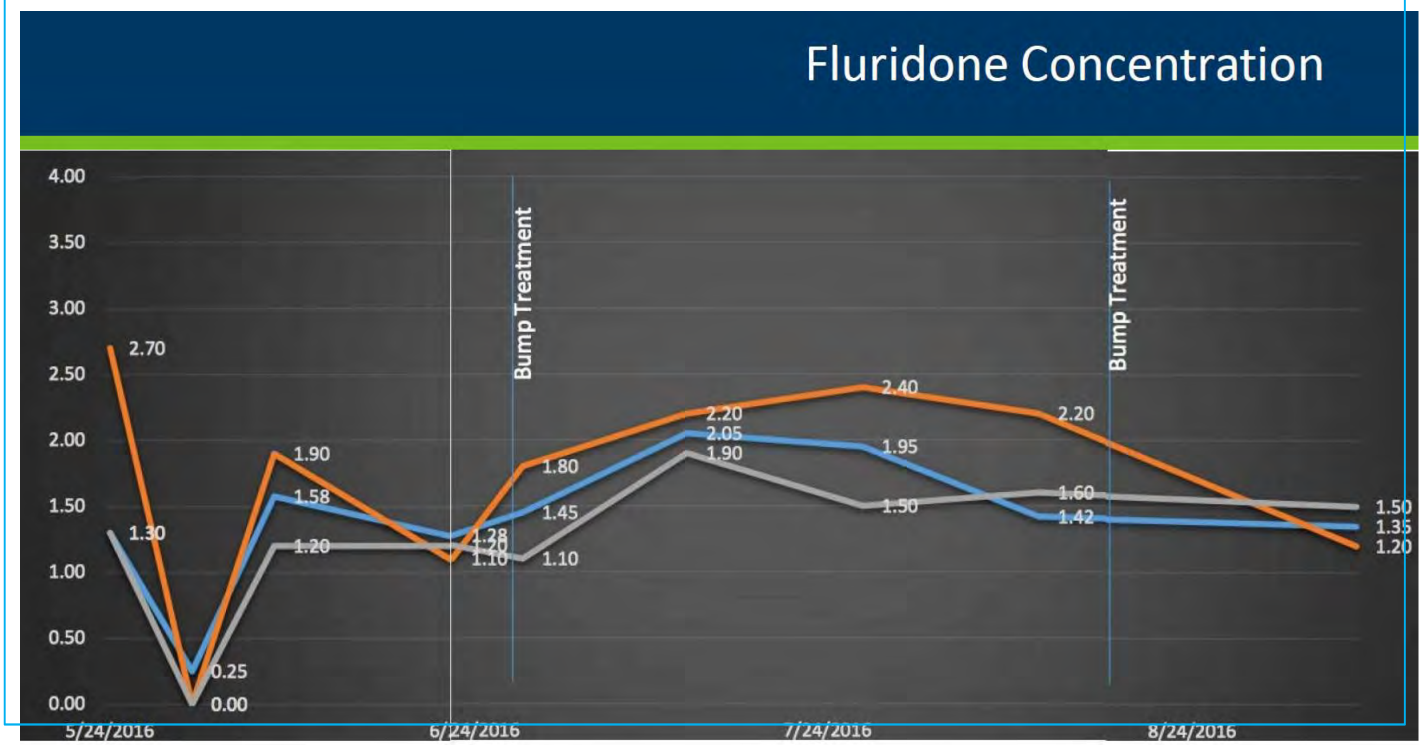
Different lake-wide management methods should be considered separately in terms of impacts to non-target species. There is little or no evidence that lake-wide management with Sonar has negative impacts to fisheries (example, longer-term outcome of 20,000 acre Houghton Lake treatment – Michigan DNR fishery assessment https://www.michigan.gov/documents/dnr/2012-141_388115_7.pdf) and all environmental assessments of Sonar use over many decades have concluded negligible risk of the herbicide to non-target aquatic fauna.

- Tribes have asked that herbicide application occur after spearfishing concludes. However, fluridone is anticipated to still be present at detectable levels into spring 2020 and may overlap with spearfishing and early season spawning activities.

It is technically accurate that fluridone detection in the early spring of 2020 is possible although a final management design will seek to minimize this carryover. All risk assessments of the herbicide support that this scenario has negligible human health and ecological risk and hopefully effective outreach on this subject will be successful in mitigating concern of tribal partners on this possibility.

- Include a column in Table 2.3-1 for plants known to be affected by fluridone as listed on the herbicide product label. Label indicates that 20 of 27 species (~75%) found in South Twin could be affected by this treatment.

Recent successful Sonar pellet treatments in the upper Midwest for Eurasian/hybrid Eurasian watermilfoil control are documenting much reduced responses of potentially sensitive aquatic plants noted on the product label. One Upper Midwest project outside of Wisconsin that was reviewed at the Upper Midwest Invasive Species Conference in October was Crooked Lake outside of Minneapolis. MN DNR reviewed results of the 2016 Sonar One pellet treatment that documented favorable efficacy and selectivity (two data snapshots below…Year of treatment, 1 and 2 year post vegetation data and the Sonar dissipation pattern in 2016). A full technical report from MN DNR is in development but relevant presentation from recent UMISC conference is attached.



Post fluridone plant summary

12/19/2018

PLANT TAXA	COMMON NAME	% OCCURRENCE 2016	2017	2018
ALL TAXA (combined)		**72**	**73**	**75**
SUBMERSED TAXA				
*Myriophyllum spicatum**	Eurasian watermilfoil	60	1	6
Ceratophyllum demersum	Coontail	50	25	26
Chara sp.	Muskgrass	41	42	42
Potamogeton illinoensis	Illinois pondweed	29	38	28
Stuckenia pectinata	Sago pondweed	17	38	23
Najas guadalupensis	Southern naiad	6	3	18
Eleocharis acicularis	Needle spikerush	1	1	1
*Potamogeton crispus**	Curly-leaf pondweed	1	16	1
Potamogeton pusillus	Small pondweed	1	5	4
Heteranthera dubia	Water stargrass	1	5	14
Najas flexilis	Slender naiad	–	6	15
Utricularia vulgaris	Common bladderwort	–	3	5
Potamogeton gramineus	Variable pondweed	–	P	3
Potamogeton zosteriformis	Flat-stem pondweed	–	2	1
Utricularia minor	Small bladderwort	–	–	1
Potamogeton nodosus	Long-leaf pondweed	–	–	1

The following table projects response of South Twin plant community to low-dose application using Sonar One pellets (+ = increases, - or -- = decreases, 0 = neutral).

Scientific Name	Common Name	Sonar One progra response YOT	YAT
Myriophyllum spicatum	Eurasian watermilfoil	--	--
Myriophyllum sibiricum	Northern watermilfoil	--	-
Ceratophyllum demersum	Coontail	-	+
Bidens beckii	Water marigold	-	+/0
Myriophyllum alterniflorum	Alternate-flowered watermilfoil	-	+/0
Vallisneria americana	Wild celery	-/0	+
Potamogeton gramineus	Variable-leaf pondweed	+	+
Najas flexilis	Slender naiad	-	+
Potamogeton robbinsii	Fern-leaf pondweed	+/0	+
Chara spp.	Muskgrasses	+	+/0
Elodea canadensis	Common waterweed	-	+
Potamogeton zosteriformis	Flat-stem pondweed	0	0
Potamogeton richardsonii	Clasping-leaf pondweed	+	+
Heteranthera dubia	Water stargrass	0	+
Potamogeton praelongus	White-stem pondweed	0	+
Potamogeton pusillus, P. strictifolious, P. friesii	Thin-leaved pondweeds	-	+/0
Potamogeton pusillus	Small pondweed	-	+
Eleocharis acicularis	Needle spikerush	0	0
Isoetes spp.	Quillwort spp.	-/0	0
Nitella spp.	Stoneworts	+	0
Potamogeton amplifolius, P.amp x praelongus	Large-leaf pondweed & Hybrid pondweed	+/0	+
Potamogeton amplifolius	Large-leaf pondweed	+/0	+
Potamogeton hybrid 1	Pondweed Hybrid 1	+/0	+

- Much of the previous data on the use of pelletized fluridone is from presentations given at conferences (i.e., Heath et al. 2018a, 2018b). Are these available publicly or in another format? Much of the justification for use of fluridone in South Twin is based on the information from a few limited Wisconsin treatments within these reports and it would be beneficial to have wider access to this data.
- Native impacts based on previous studies (i.e., Heath et al. 2018a, 2018b) were stated, but it is not clear how these previously documented impacts were actually taken into consideration when deciding to pursue fluridone. There seems to be a disconnect between listing likely native impacts and then still recommending this particular treatment strategy. Need a more throughout analysis of pros and cons of this treatment approach; it seems that the lake group believes the pro of controlling EWM outweighs any potential cons to native plants that may be associated with this treatment. Perhaps consider incorporating native plant 'goals' in addition to EWM? [and more than just a generic 'strive to protect native communities' blanket statement; for hypothetical example: a whole-lake treatment will not occur if native plant populations [and/or species X] have been reduced to more than Y% of pre-treatment levels; this will allow for consideration of both EWM goals as well as native plant protection].
- Data from previous pelletized fluridone applications have been for HWM control. The efficacy of low dose pelletized fluridone on pure EWM is not known (but likely to be similar?).

Berger et al (2012) showed that fluridone response of multiple sensitive biotypes of invasive watermilfoils was similar. Field experiences in recent years have focused on HWM because of interest in alternate methods to HWM but response to fluridone treatment with one documented exception (Townline Lake, MI hybrid – study focus in Berger paper) is similar for EWM and HWM accessions.

- "*Overall the selectivity of the pelletized fluridone strategy appears to be better than the liquid case studies*." (p.12). This generalization is biased based upon the data presented. The liquid case studies (compiled by DNR SS) were at relatively higher application rates [i.e., 10+ ppb or 6-bump-6] than more current pelletized fluridone applications [i.e., 2-4 ppb] and so making a comparative statement on the selectivity of these two formulations is not an 'apples to apples' comparison since the liquid applications were at ~2-3x the rate of the current pelletized applications, and that difference in application rate would have understandably impacted selectively (regardless of the formulation).
- Fluridone is a slow-acting herbicide and mgmt. 'success' may not necessarily be seen by lake riparians during the 2019 open water season as plants are slowly dying back. There is little to no discussion on the length of time it takes fluridone to kill the plants - is lake association aware that this product may not provide relief from nuisance conditions until after the 2019 season (i.e. year 2020)?

Timing of 'relief from nuisance conditions' will be driven by timing of initial Sonar pellet application. If application occurs around the time of thermocline development, there will be some biomass up near the surface in early stages of application but less than under the same conditions without treatment. By mid-later summer—approximately two months into active management—EWM drop out will be more noticeable but there will be detectable injured EWM throughout the summer of 2019. Representative photos of milfoil condition at 3+ months post application are below (left – Crooked Lake, right – Silver Lake Kenosha County)



- Will dissolved oxygen be monitored following treatment? Are there concerns about lakewide plant die-back leading to low DO conditions?
- At this time, the Department does not fund any use or purchase of drones through surface water grants.
- The review team recommends that a quantitative trigger (i.e., >12% EWM frequency) should be utilized to initiate discussion/consideration of all available management options, and not just a direct pathway for considering large-scale herbicide treatment. Group needs to take a more integrated management approach and should not rely solely on herbicides without critically looking at the data periodically over time. Views on herbicides (and other available management tools) may very well change over time as more data and resources becomes available. Team also feels strongly that given the previous documented frequencies of EWM on S. Twin Lake that a trigger of 12% littoral frequency is too low and very likely not sustainable to maintain over the long-term.
- Need consistency between mgmt. triggers listed in approved mgmt. plan and what's included in this EWM control report.
- Integrated pest management is barely mentioned in this report, except for one sentence which says that these activities will 'preferably' occur. Based on a lack of critical discussion on IPM, it does not seem that this component of the management strategy has been critically discussed and planned for, but rather is added in as an after-thought behind a very lengthy discussion on chemical control. A critical discussion of IPM following the large-scale fluridone treatment needs to added to this report. Include a discussion on why previous IPM didn't meet the lake association goals (too little time and effort?), and work to identify an IPM strategy that may be more successful than what was previously implemented

(i.e. maybe consider building a DASH boat and/or hiring a full-time crew instead of hiring a company for only a few days per season).

- If fluridone does not met the EWM mgmt. goals defined by the lake associations, what is next? It is almost certain that EWM will rebound again following this fluridone treatment, as it has rebounded in all the other case studies presented (i.e. Heath 2018a, 2018b). Are non-chemical controls being considered in the future?
- The team does not necessarily agree with statements that prior large-scale 2,4-D treatments have been 'ineffective'. These past treatments have resulted in lakewide control of EWM, and in the case of the 2010 treatment multi-year lakewide control of EWM without the need for additional large-scale management. If the group believe that an 'effective' treatment is only one where EWM never comes back and re-establishes, then we fear that they will never achieve that goal with the strategies and current efforts which they are implementing. If EWM almost inevitably comes back again (likely at a lakewide scale) following the fluridone treatment, will this treatment also be deemed 'ineffective' by the lake group in the near future? Need some clarity on what the lake group believes *is* actually an effective treatment outcome rather than just pointing out (after the fact) all the scenarios that they deem non-effective.
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- Statements that ProcellaCOR: "*is the best herbicide because of its rapid absorption as well as selective impact on EWM*" is not supported by actual data which the review team is aware of.

Recent herbicide uptake work by Haug at NC State University showed ProcellaCOR reaching 90% of maximum uptake (t_{90}) by EWM or HWM in 6-24 hours.

https://repository.lib.ncsu.edu/bitstream/handle/1840.20/35124/etd.pdf?sequence=1

Vassios at Colorado State University using nearly identical methods showed triclopyr t_{90} uptake to take a calculated 73 hours.

https://mountainscholar.org/bitstream/handle/10217/67656/Vassios_colostate_0053A_10996.pdf

On selectivity and short exposure efficacy, past similar large-scale work by Netherland and Glomski (2014) (link provided below) can be compared to results for ProcellaCOR by Beets et al (2019 JAPM in press…key figures 1 – 4 provided below). The Beets data has been presented at multiple meetings and conferences involving DNR staff.

http://www.apms.org/wp/wp-content/uploads/japm-52-02-57.pdf

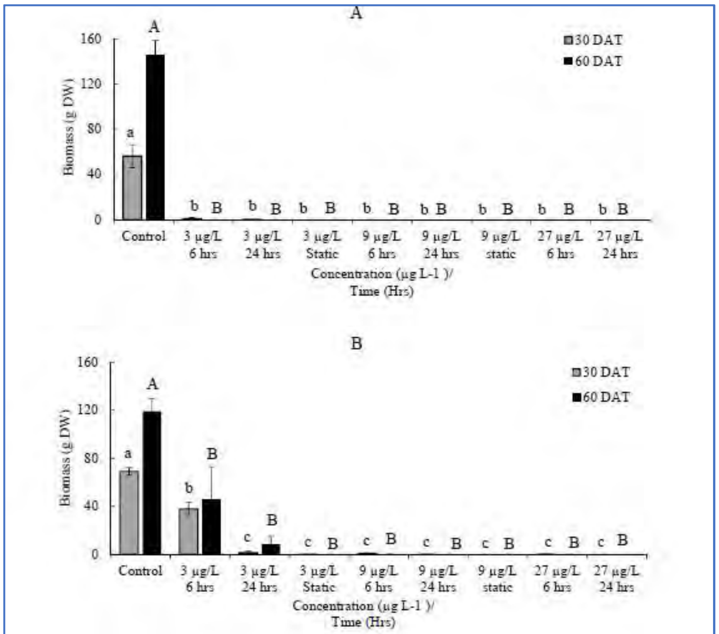

Figure 1. Mean (± SE) dry aboveground biomass at 30 and 60 days after treatment (DAT) with florpyrauxifen-benzyl at 3 µg L^{-1} for 6 hr, 24 hr and static water-exchange half-lives, 9 µg L^{-1} for 6 hr, 24 hr and static water-exchange half-lives, and 27 µg L^{-1} for 6 and 24 hr water-exchange half-lives on (a) EWM and (b) HWM (n=3). Letters above bars represent differences between treatments according to Tukey's test (α=0.05). Uppercase letters indicate 60 day harvest dates that were analyzed separately.

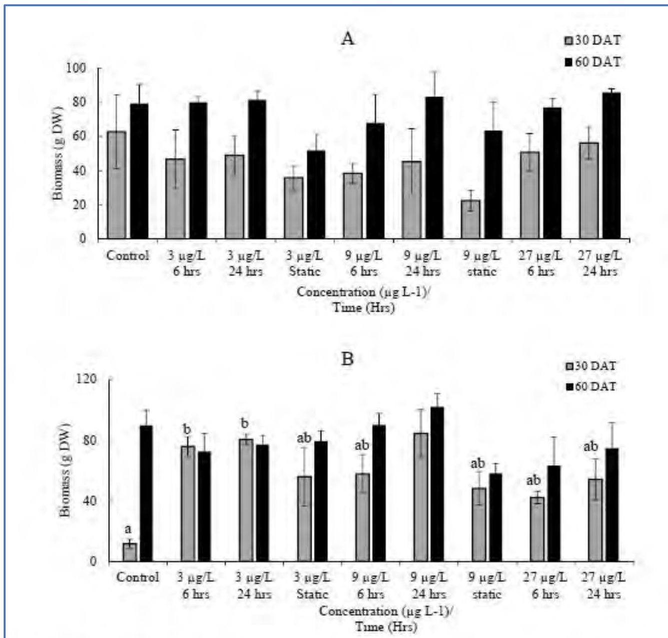

Figure 2. Mean (± SE) dry aboveground biomass at 30 and 60 days after treatment (DAT) with florpyrauxifen-benzyl at 3 µg L^{-1} for 6 hr, 24 hr and static water-exchange half-lives, 9 µg L^{-1} for 6 hr, 24 hr and static water-exchange half-lives, and 27 µg L^{-1} for 6 and 24 hr water-exchange half-lives on (a) American pondweed and (b) Illinois pondweed (n=3). Letters above bars represent differences between treatments according to Tukey's test (α=0.05). Differences in mean biomass between 60 day treatments were not observed.

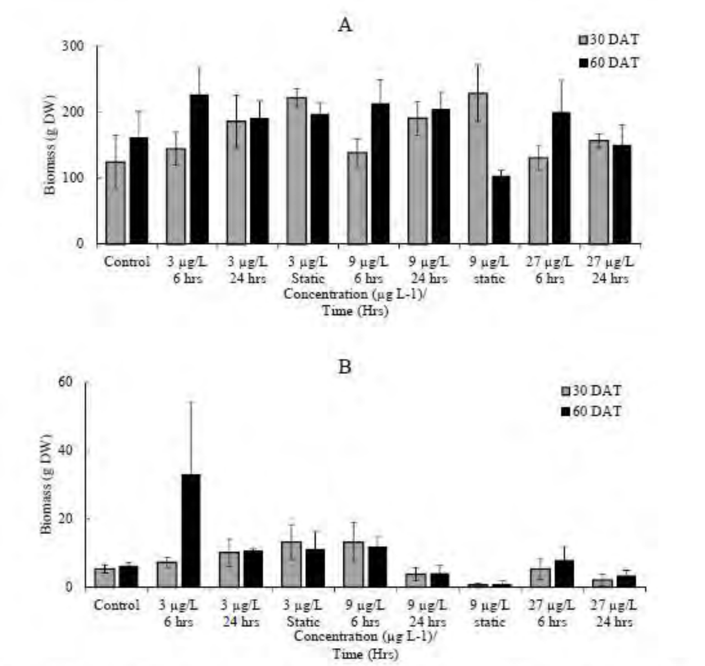

Figure 3. Mean (± SE) dry aboveground biomass at 30 and 60 days after treatment (DAT) with florpyrauxifen-benzyl at 3 µg L^{-1} for 6 hr, 24 hr and static water-exchange half-lives, 9 µg L^{-1} for 6 hr, 24 hr and static water-exchange half-lives, and 27 µg L^{-1} for 6 and 24 hr water-exchange half-lives on (a) elodea and (b) *Heteranthera* (n=3). Differences in mean biomass were not observed between treatments at 30 and 60 DAT.

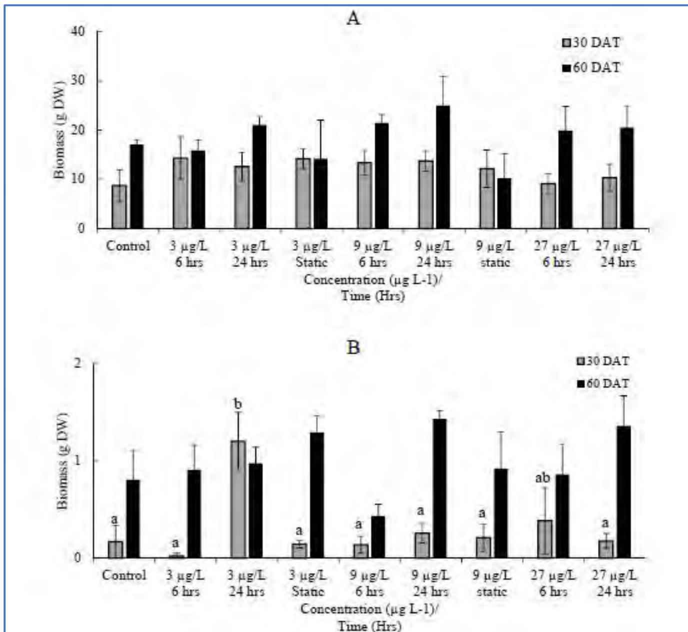

Figure 4. Mean (± SE) dry aboveground biomass at 30 and 60 days after treatment (DAT) with florpyrauxifen-benzyl at 3 µg L^{-1} for 6 hr, 24 hr and static water-exchange half-lives, 9 µg L^{-1} for 6 hr, 24 hr and static water-exchange half-lives, and 27 µg L^{-1} for 6 and 24 hr water-exchange half-lives on (a) Southern vallisneria and (b) Northern vallisneria (n=3). Differences in mean biomass were not observed between treatments at 30 and 60 DAT for S. vallisneria or 60 DAT for N. vallisneria.

- Need to better understand the mechanism of why EWM is re-establishing on a lakewide scale following large-scale chemical mgmt. Are plants re-establishing from seeds, root stocks, vegetative materials, being re-introduced? This is a big question that needs to be answered, as if we don't understand the mechanism of EWM recovery then we cannot tailor additional IPM to target this mechanism before it spreads to a lakewide scale again.
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The 6h exposure scenario in Beets et al 2019 is basically an identical planned exposure scenario as one of two simulated by Skogerboe et al (2006 – link provided) using diquat on EWM and several other species in the same large mesocosm system at the Corps facility in Lewisville TX in the early 2000s. ProcellaCOR exposure time requirements may not be quite as short as diquat but the systemic activity provides better control with short exposure along with much greater selectivity.

http://www.apms.org/japm/vol44/v44p122.pdf

- Figure 3.2-3 is confusing. Are black bars the data from within the targeted treatment areas over time, or is this lakewide PI data??
- North Twin: Mid-October 2018 pre-treatment sub-PI data is very late in the season to capture all natives, many of which are likely senscing….why was this data collected so late in the season?? When will the post-treatment sub-PI data be collected? May want to consider scheduling the 2019 survey timing to best match up with pre-treatment data collection, or consider two sub-PI surveys (i.e. mid-summer and then mid-October).
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- ProcellaCOR: states "short exposure time scenarios". Can this be quantified. i.e. X hrs, X days?

See data above from Beets et al 2019 (in press). Effective exposure times are on the order of 6 – 12 hours. Table 1 of representative dissipation monitoring results for the Beets study are below.

Table 1. Mean (SE) florpyrauxifen-benzyl concentration (μg L^{-1}) collected at hours after treatment (HAT) and days after treatment (DAT) intervals following treatment (n=3). Dashes indicate time periods where no sample was collected.

CET scenario	1 HAT	6 HAT	24 HAT	48 HAT	72 HAT	7 DAT	10 DAT	14 DAT
27 μg L^{-1} 6 hr	16.2 (2.8)	8.1 (0.66)	3.9 (2.9)	1.3 (0.21)	-	-	-	-
27 μg L^{-1} 24 hr	14.3 (2.1)	8.9 (0.72)	9.0 (0.31)	8.6 (2.48)	2.0 (0.42)	0.84 (0.21)	-	-
3 μg L^{-1} static	2.2 (0.2)	-	-	-	1.6 (0.67)	0.77 (0.43)	0.10 (0.03)	0.07 (0.03)
9 μg L^{-1} static	6.9 (0.5)	-	-	-	2.6 (0.04)	1.2 (0.27)	0.25 (0.04)	0.08 (0.04)

- On N. Twin, the main dominant native species within the treatment area is coontail, and NWM is also relatively dominant. Both of these natives are known to be susceptible to ProcellaCOR. Will damaging these natives potentially create more room for EWM to eventually colonize within this area?

2018 field efficacy results from WI (Silver Kenosha) and multiple other US projects have been provided to DNR showing lack of control of coontail. NWM is sensitive and will be controlled by a ProcellaCOR treatment of EWM.

- What was PDU rate used on Silver Lake, Kenosha Co. How does this compare to the rate being recommended for N. Twin?

PDU rate on Silver Kenosha was 5 PDU / Ac-ft. N. Twin application rate is planned for 8 PDU / Ac-ft based on known dilution characteristics of this location and past challenges with spot treatments with many herbicides.

GLIFWC comments

In the report the authors acknowledge “that management activities such as herbicidal treatments can impact some native species in the Twin Lakes” (which is highlighted in the various species abundance graphs), however, the link seems missing that the numerous APM treatments over the years at North and South Twin may actually be intensifying the issue. While it may seem “logical to consider alternative treatment methodologies to improve probability of success” – this may ultimately lead to further damage to the native plant community, increase the probability of treatment-resistant invasives, and still not be deemed successful to lake users. In addition, the reliance on DASH in subsequent years after the next 2019 chemical treatment seems laudable but doesn’t seem to reflect the past course of management strategies at North and South Twin; meaning will a different chemical treatment be proposed again in a year or two?

Unless an eradication approach is officially implemented and resourced, there will be variance in levels of invasives in a longer-term containment and maintenance strategy driven by a variety of factors. These factors include types/intensity of management, the availability of preferred niches in infested systems (only a certain percentage of the lake littoral is most favorable for invasive establishment and growth), and climatic variance (some years may favor invasive pressure more than others). Successful selective and sustained maintenance control requires continuous assessment and improvement of strategies. It is rare that such well-implemented efforts will see a continuous decline in desirable species and selection for the invasive. However, there will be variance in levels of infestation that are often predictable based on the intensity and quality of management that can be implemented based on available resources and sometimes the basic strength of diverse partnerships established to manage public aquatic resources.

- Have the native plant communities in North and South Twin significantly recovered to withstand another chemical treatment in 2019?
- Have the authors considered non-chemical management strategies for 2019 instead? (Using a mechanical harvester to help create navigation channels could thin out the sub-surface EWM, thereby lessening boating challenges, and potentially allow for more light penetration, possibly aiding in native plant recovery.)
- Who determines when stratification has occurred? (Is there information of prior years’ stratification timing?)

2016 data confirmed thermocline development in late May. Sonar One pellet treatment should begin around that same time (late May – early June) when surface water temperatures should more consistently be 65F and above. More aggressive EWM growth should begin around that time as well, which will enhance response of the invasive milfoil to Sonar herbicide. It is also important to be timely with initial treatment because delayed application will increase carbohydrate reserves of EWM and make its control slower over the summer.

- Are tribal spring harvest activities taken into account regarding the proposed or ideal treatment timeline? (A recommendation that no treatment occur within 2 weeks of the spring spearing season might be in order.)
- Could a possible treatment occur around mid-late June or would they want it earlier? (The later dates may allow for the conclusion of the tribal harvest season and for walleye eggs to develop and larval fish to move offshore.)

See above – late May / early June appears more optimal. ProcellaCOR shows no toxicity to fish regardless of life stage (EPA MRIDs 49677735, 49677736, 49677737, 49677742, 49677747, 49677910 plus Grue & Crosson 2018 in prep) so risk to walleye at any life stage appears negligible.

Sonar (fluridone) has a several decades-long history of regulatory and published studies, environmental risk assessments, and operational use without predicted or measured negative impacts to fish. In specific toxicity studies relative to walleye, Paul et al 1994 also showed no effect on juvenile walleye at 780 ppb, which is ~250X greater than concentrations projected for a South Twin Lake treatment.

- Do the Twin Lakes APM applicants plan to consider the spawning, hatching and early growth phases of fish in order to avoid treating during critical life stages? (This is especially important since some of the proposed treatment area overlaps with potential spawning habitat.)

- Have toxicity studies fully examined the effect of fluridone on early life stages of fish? (Early life stages of fish are generally most susceptible to toxic effects of treatment chemicals.)

See above

- It appears as though native aquatic plants found all through the entire water column are potentially affected by ProcellaCOR. What can be done to mitigate the potential losses?

All university and government scientists who have worked with the herbicide can confirm the high selective activity of the herbicide on invasive watermilfoils and the comparative tolerance of native, desirable aquatic plants. Technical findings can be found in a number of recent journal publications (Netherland and Richardson 2016, Richardson et al 2016, Haug 2018, Beets et al 2019 in press). A recent field project in Minnesota monitored by MNDNR (Lake Jane) also documents successful selective control of HWM and a draft report on the project from DNR is attached.

- Are the current or recent invasive levels in N&S Twin seen as a product of repeated chemical treatment applications?

Onterra can comment further but invasive levels likely reflect different levels of integrated management pressure (with overall less management in years with increasing densities of invasives) and also year-to-year variance in environmental conditions promoting different levels of weed growth.

- Also, it is unclear if the treatment strategies/recommendations in the email attachment are those of the two authors or are those of Onterra as well.

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Great Lakes Indian Fish & Wildlife Commission Comments to North and South Twin Lake 2018 EWM Monitoring & Control Strategy Development Report

November 2018

Lisa David, Manoomin Biologist

Response Comments by Carol Warden, WDNR Project Team Leader, UW Trout Lake Station Center for Limnology Aquatic Invasive Species Specialist
Response Comments by Jay Wittman, NSTLPRD

In the report the authors acknowledge “that management activities such as herbicidal treatments can impact some native species in the Twin Lakes” (which is highlighted in the various species abundance graphs), however, the link seems missing that the numerous APM treatments over the years at North and South Twin may actually be intensifying the issue. While it may seem “logical to consider alternative treatment methodologies to improve probability of success” – this may ultimately lead to further damage to the native plant community, increase the probability of treatment-resistant invasives, and still not be deemed successful to lake users. In addition, the reliance on DASH in subsequent years after the next 2019 chemical treatment seems laudable but doesn’t seem to reflect the past course of management strategies at North and South Twin; meaning will a different chemical treatment be proposed again in a year or two? We discussed this long-term, larger framework and I was explicit in our conversation with the lake group that if a whole-lake fluridone treatment is approved for 2019, it in no way means another chemical treatment (of any kind) would be guaranteed again in the future based largely on what you lay out here. Our management plan is outlined in our narrative. We desire to return the lake to as "natural" status as possible giving consideration to all stakeholders as well as the resource of water, plants and fishes. Given the current impaired status of S Twin resulting from EWM population, we believe it requires a new herbicide followed by aggressive DASH and hand pulling and possibly spot treatments down the road to get S Twin "under control"! Subsequent data collection will be critical to assist with future strategies.

Further,

- Have the native plant communities in North and South Twin significantly recovered to withstand another chemical treatment in 2019? This is also something we discussed with the lake group. The downward trajectory of their native species along with the fact that fluridone may impact 20 of their 27 species is a very real concern from the DNR's perspective. Though many of the plants have fluctuated through time, some have been moving toward lower and lower numbers. This has not been overlooked and is considered in the process. See narrative, we believe yes
- Have the authors considered non-chemical management strategies for 2019 instead? (Using a mechanical harvester to help create navigation channels could thin out the sub-surface EWM, thereby lessening boating challenges, and potentially allow for more light penetration, possibly aiding in native plant recovery.) They have. The lake group feels strongly that mechanical harvesting is equivalent to throwing in the towel. They want to

get a hold on the population with a "1,2,3 punch" largescale treatment, then implement hand pulling and DASH. We were not shy in saying that we would be thrilled if the lake group decided to go with a mechanical harvesting route and that it wouldn't mean we could never discuss alternatives again. As of now, the lake group is resolved in a whole-lake fluridone treatment as their best/most effective mode of action. Jay and Joe, please feel free to add or correct me on this. See final report which has a segment on mechanical harvesting. Below is the Lake District's perspective on Mechanical Harvesting:

> Mechanical harvesting is a form of management that the former Lake Association (NSTLRA) and current Lake District (NSTLPRD) have discussed with Onterra, WDNR and other associations, applicators and our lake riparians. Our consistent concerns are we do not believe we have explored all alternative management options, which we and the riparians believe should be the priority. It is generally viewed by the Lake District, et al, that Mechanical Harvesting is a last resort form of Management to allow for riparian use of the lake resource. At this time a herbicide such as Fluridone (low dose, long CET) has never been used and based on research it felt to be a preferred management option. Additionally, this herbicide treatment is to be followed by several years of monitoring and use of less aggressive actions such as DASH and hand pulling initially and possibly herbicidal spot treatment to achieve extended efficacy from EWM. Unless that management approach is utilized, we will never know it's wholistic success in combating EWM. Should that not be successful, other management approaches including mechanical harvesting can be explored. We have a strong concern that mechanical harvesting will not provide desired efficacy and will contribute via fragmentation to the expansion of EWM into N Twin. Thus, mechanical harvesting is not a management practice for the Twin Lakes at this point in time.

· Who determines when stratification has occurred? (Is there information of prior years' stratification timing?) The consultant will either take these readings themselves or enlist someone from the District to do so.

· Are tribal spring harvest activities taken into account regarding the proposed or ideal treatment timeline? (A recommendation that no treatment occur within 2 weeks of the spring spearing season might be in order.) Yes, we can work with recommendations such as these to determine application time. Something else that came up in the meeting was that fluridone persists throughout the year so it may be a concern for 2020 harvest. Eddie of Onterra (consultant) said we could work with adjusting bump concentrations, the lake group voiced that they still want to be sure that treatment is done in a way that will give the treatment its best chance at being a success. As was the case when we treated N Twin in 2017, we intend to allow tribal spearing to be completed in 2019 prior to our treatment.

· Could a possible treatment occur around mid-late June or would they want it earlier? (The later dates may allow for the conclusion of the tribal harvest season and for walleye eggs to develop and larval fish to move offshore.) We could discuss this. Mid to late

June would not allow adequate exposure time for Fluridone to maximize efficacy of this treatment.

· Do the Twin Lakes APM applicants plan to consider the spawning, hatching and early growth phases of fish in order to avoid treating during critical life stages? (This is especially important since some of the proposed treatment area overlaps with potential spawning habitat.) I will speak for DNR and Twin Lakes District that there is an understanding that this is important to consider. We are open to discussion on this.We have had conversation with the Mole Lake tribe who expressed possible interest in monitoring fish tissues pre, during and post treatment as an option. Suggestions as to how GLIFWC can partner on this?

· Have toxicity studies fully examined the effect of fluridone on early life stages of fish? (Early life stages of fish are generally most susceptible to toxic effects of treatment chemicals.) According the the EPA and DNR fact sheet, there are not acute or long-term effects on fish. Agree

· It appears as though native aquatic plants found all through the entire water column are potentially affected by ProcellaCOR. What can be done to mitigate the potential losses? The consultant proposes a low dose with a bump treatment in hopes to mitigate potential loss of native plants. The truth is, we are uncertain what the total effect will be on the native species in S. Twin. Carol errored in her response here. ProcellaCOR has had very good selectivity regarding aquatic plants with focus on HWM/EWM. See Dr Heilman commentary. This is specific only to 14.3 acres on N Twin.

· Are the current or recent invasive levels in N&S Twin seen as a product of repeated chemical treatment applications? DNR has discussed this hypothesis with the lake group. It is indeed a possibility, however we do not have conclusive evidence to say unequivocally. There is a real concern that a whole-lake fluridone treatment would knock native plants down and leave even more real-estate on the lake bed for more milfoil to take hold. That said, the milfoil has a very large footprint already and it's hard to say how much/if that footprint would advance. See our native plant comments and Dr Mark information.

· Also, it is unclear if the treatment strategies/recommendations in the email attachment are those of the two authors or are those of Onterra as well. From what I understand, Onterra lists strategies that the lake group is interested in after working together one what may be the best course of action. Hopefully Jay and Joe can provide a better understanding as to whether Onterra proposes a strategy and the lake group adopts it, or whether the lake group takes ideas to their consultant and then the consultant writes it up in a comprehensive way? Our approach to Lake Management is a collaborative one. We have reached out to Pioneer Lake, GLIFWC, Mole and Lac du Flambeau tribes, multiple applicators, MI and MN DNR equivalents, lake associations and districts, a survey of our lake riparians, experts at UWSP and Onterra, Dr Mark Heilman and read multiple studies on a variety of lake management activities in order to make the best possible recommendations for management of The Twins. We do not take this responsibility lightly. We welcome ideas, comments, information etc so our knowledge continues to grow so we can effectively manage the resource in an appropriate manner.

South Twin Lake Aquatic Analysis
2008 - 2018

This will be Appendix A data in the 2018 Annual Monitoring Report for the Twin Lakes as gathered by Onterra and evaluated by NSTLPRD. The analysis comments, assumptions and conclusions area as follows;

ASSUMPTIONS

- While Northern Milfoil is not technically an AIS, for purposes of this analysis it shall be excluded as the native plant community is evaluated over time.

- We acknowledge that the concentration of 2,4-D whole lake treatment in 2010 had an impact on native plants as it was a stronger concentration of 2,4-D.

- We have excluded any plants which over the period of 2008 - 2018 having an average LFOO% of < 1% as we feel they are not a material element of the plant community and thus should be excluded from the analysis.

- The analysis thus focuses on the 22 native species found in S Twin which have a material presence exclusive of Eurasian and Northern Milfoil.

FACTS

- DNR noted in their technical review that the # of natives per site was 4.2 species/site in 2008 and has ranged from 2.5 - 3.5 species/site since large-scale began. While true, we do not know if 2008 was an abnormally high year for native plants in S Twin. We unfortunately do not have site data prior to 2008 which would provide a more historical trend over time vs one data point. However, we do know that 2017 had the highest # of natives per site since 2008 at 3.5 and we saw a precipitous decline in 2018 to 2.5. Might that suggest that the significance in density of EWM in 2018 (Madsen 1991, 1999,Boylen 1999) at 71 of 145 acres of EWM with highly dominant or matted may be responsible for such decline?

- Using the period 2008 as a baseline for measurement compared to 2018 there were 15 species who declined and 7 species who increased in LFOO%.

- Using the period 2011 as a baseline for measurement compared to 2018 there were 10 species who declined and 12 species who increased in LFOO%.

- Using the period 2017 as a baseline for measurement compared to 2018 there were 14 species who declined and 8 species who increased in LFOO%. Impact of EWM?

- Of the 22 species evaluated, there were 13 species which had their highest LFOO% in years 2008 or 2009 prior to the 2010 2,4-D treatment.

- Of the 22 species evaluated, there were 9 species which had their highest LFOO% in years 2015 - 2018.

OBSERVATIONS

- Coontail reached it highest LFOO% post 2010 treatment in 2017 at 18.6%. It had it's low point of 9.18% in 2015 when EWM increased to 37%. Coontail actually increased in 2016, a year in which we performed a whole lake treatment of 2,4-D, from the 9.18% to 11.6%. We again saw a decrease in coontail from 18.6% in 2017 to 15.1% in 2018 when coincidentally, EWM increased to 40%. Coontail stayed relatively flat (21 - 23.4) in the period 2008-2011 which included the 2010 treatment. The data suggests that coontail growth has declined each time there has been significant EWM population in S Twin and herbicidal treatments do not appear to have impacted it's LFOO% in S Twin.

- Water Marigold has rebounded to a level of 7.24% which is higher than the 6.27% experienced pre 2010 treatment and is at it's highest level since 2008. WM decreased from 14.1% in 2008 to 6.27% in 2009, a decrease of over 50% yet no herbicide impact in those years.

- Wild Celery was at levels of 58.2% in 2011, post 2010 treatment, and was trending up from 45.6% in 2009. In 2018 it showed a significant decline to 33.2% from 45% in 2017. There does not seem to be conclusive data to suggest that past treatments or EWM concentration are contributors to Wild Celery trends. Rather, it appears that Wild Celery population data suggests it will have cycles ranging between 30% - 60% LFOO.

- Variable Leaf Pondweed was at 44.3% in 2017 compared to 46.7% in 2008. It has a similar pattern to Coontail in that is has suffered significant declines when EWM population is significant as exhibited by the 38.4% in 2015 and 29.6% in 2018. The 29.6% in 2018 is the lowest % for VLP in the entire data set. VLP actually increased after the 2016 treatment so difficult to suggest it's life trends are impacted by application of herbicides.

- Slender Niad has been at it's highest level in the data set in 2017 when it reached 40.7%.

- Fern-leaf Pondweed has shown similar trends to VLP and Coontail in that it's declines correlate when EWM presence is significant.

- Muskgrasses peaked in 2017 at 40.7% and had significant decline in 2018 to 13.8%. The 40.7% was the highest LFOO% in the data set. Muskgrass does not seem to trend either with herbicides or EWM.

- Common Waterweed has experienced a large decline from 34.5% in 2017 to 10.5% in 2018 and also strong growth in 2011 at 31.3% following treatment vs 2008 level of 24.7%. No correlation between treatments and EWM can be concluded other than it appears to cycle in a range of 10% - 30%.

- Flat Stem Pondweed data does not suggest any conclusion regarding EWM presence or treatments other than it's cycle appears to range between 10% - 30%.

- Clasping Leaf Pondweed, Needle Spikerush, Quillwort, Stonewort, and Large Leaf Pondweed all show highest LFOO% in 2015 or 2017 and show some decline in years when EWM presence is high.

- Water Stargrass has been reasonably stable each year from 2009 - 2018 at +/- 10%.

- White Stem Pondweed has been stable at +/- 10% during the data set.

- Thin Leaved Pondweeds were at highest levels (16%-20%) in 2008 and 2009 prior to 2010 treatment and has only rebounded to a level of 10.9% in the period 2011 - 2018.

- Small Pondweed was at highest levels in 2008 and 2009 and has only reached 10.3% in 2013.

CONCLUSION

The observations of the most significant natives in S Twin have varying trends. Thin Leaved and Small Pondweed appear to have been impacted by the 2010 treatment as they have not rebounded to pre 2010 treatment levels. They have been operating in a range of 5%-10% for the period post 2010 treatment which may suggest a possible new baseline. Eight of the twenty-two natives were at their highest level in the data set in either 2017 and 2018 which suggests they are strong relative to the period for which we have data. Coontail, Variable Leaf Pondweed and Fern Leaf Pondweed all have exhibited trends for decline when EWM presence is high in two recent cycles. Other species discussed above appear to have cyclical trends which we are not able to conclude if EWM, climate, or herbicide are impacting their population.

The District believes that the lowered use rate for fluridone being proposed for S Twin Lake will have less impact on natives than previously used higher dosage strategies of 2,4-D used in 2010 specifically. Additionally, The District believes that EWM itself is impacting the decline of some natives as evidenced in trends for years following EWM presence > 25%. The District, a public entity, is elected in part, to represent the over 500 riparian parcels in the district who have clearly voiced their support for herbicidal treatment of the Lakes in order to permit them to use the resources for recreation which the WDNR clearly witnessed when they visited this summer. This was documented in the 2016 riparian survey where 134/164 (82%) respondents either strongly or moderately supported herbicidal treatment while 8/164 (5%) moderately or completely opposed herbicidal treatment. We have been very transparent with the other stakeholders (Pioneer Lake Assoc, Lac du Flambeau and Mole Lake tribes and GLIFWC) regarding our planned management activities. Our biggest concern today is the migration of EWM into N Twin which has clearly expanded in size and density over the past 5 years. Should this expanse continue, the impact to riparians, the lake eco system, economic value of property and the impact to area businesses will undoubtedly be in jeopardy.

While we understand that mechanical harvesting is a management option, we do not believe it should be an option today until a multi year strategy (which has never been done) is conducted. This includes;

Year 1 - Herbicidal treatment of S Twin Lake with Fluridone, N Twin large scale spot with ProcellaCOR in an expanding colony, DASH for 15 days in N Twin in areas treated herbicidally in 2017 in order to maximize length of efficacy for that treatment.

Year 2 - Active monitoring of S Twin analyzing impact on natives and water quality, DASH in areas to-be determined and DASH in the 14 acre area on N Twin treated in 2019. Fall PI.

Year 3 - Continued monitoring and aggressive use of hand pulling/DASH to maximize efficacy of herbicidal treatments. Possible spot treatments to manage control in targeted areas. Fall PI. Riparian survey.

Year 4 - Depending on fall PI from Year 3, trends on natives, impact on riparian recreational use of the resource, feedback of stakeholders, we will evaluate the state of the lake and what management options should be considered, if any updates to our Lake Management Plan are warranted in order to continue to manage the lake in a responsible manner.

Today, the ability to recreate on the Twin Lakes is significantly impaired, it is unknown as to the future impact EWM will have on the balance of the fisheries in the lakes, there are property value concerns as well as area economic impacts to the greater Phelps/Conover communities and we believe the native plant community is sufficiently strong to be able to withstand the impact of the proposed treatments. The District believes the proposed management activities are in the best interests of the resource and the stakeholders given the known facts we have available to us. Future data gathering will be critical to determining what future management activities are most appropriate to balance concerns over EWM, native plants, economic and recreational desires, fish impact and other interests of stakeholders.

South Twin Lake Historical Aquatic Analysis

Scientific Name	Common Name	South Twin LFOO (%) 2008	2009	2010	2011	2012	2013	2014	2015	2016	2017	2018	Change 2018 v 2008	Change 2018 v 2011	Change 2018 v 2017
Myriophyllum spicatum	Eurasian watermilfoil	20.7	10.8	0	0.33		3.22	11.9	37.7	4.41	14.3	40.1	19.41	39.80	25.80
Myriophyllum sibiricum	Northern watermilfoil	28.3	12.9	0	0		1.93	8.61	11.8	5.08	4.23	7.24	-21.05	7.24	3.00
Ceratophyllum demersum	Coontail	23.4	22	21	22		14.5	14.2	9.18	11.9	18.6	15.1	-8.22	-6.91	-3.44
Bidens beckii	Water marigold	14.1	6.27	0.65	2.3		3.86	6.62	6.56	2.71	1.95	7.24	-6.91	4.93	5.28
Myriophyllum alterniflorum	Alternate-flowered watermilfoil	3.62	4.88	0.65	0.99		0.96	0.99	5.25	3.39	5.86	8.88	5.26	7.89	3.02
Vallisneria americana	Wild celery	60.9	45.6	53.7	58.2		37.3	36.8	43.6	47.8	45	33.2	-27.63	-25.00	-11.73
Potamogeton gramineus	Variable-leaf pondweed	46.7	49.1	37.5	40.5		46.6	52.6	38.4	46.1	44.3	29.6	-17.11	-10.86	-14.69
Najas flexilis	Slender naiad	33.2	33.1	17.2	43.4		14.8	25.8	33.4	27.8	40.7	35.2	1.97	-8.22	-5.52
Potamogeton robbinsii	Fern-leaf pondweed	31.3	34.5	33.7	31.6		30.9	30.8	24.9	26.1	25.7	16.4	-14.80	-15.13	-9.29
Chara spp.	Muskgrasses	29.3	18.1	22.3	25.3		14.5	16.9	32.1	40.3	40.7	13.8	-15.46	-11.51	-26.90
Elodea canadensis	Common waterweed	24.7	27.9	15.5	31.3		15.1	12.9	14.1	24.4	34.5	10.5	-14.14	-20.72	-24.00
Potamogeton zosteriformis	Flat-stem pondweed	31.3	27.5	5.18	16.1		26	20.9	12.5	7.46	10.4	10.5	-20.72	-5.59	0.10
Potamogeton richardsonii	Clasping-leaf pondweed	11.5	18.5	16.8	18.1		10.9	8.94	7.87	13.2	21.5	11.2	-0.33	-6.91	-10.31
Heteranthera dubia	Water stargrass	22.4	9.41	4.21	8.55		9.65	13.6	16.1	10.2	11.7	9.21	-13.16	0.66	-2.52
Potamogeton praelongus	White-stem pondweed	10.5	10.5	10.4	7.57		10.3	5.63	7.87	10.2	9.77	8.22	-2.30	0.66	-1.55
Potamogeton pusillus, P. strictifolious, P. freissi	Thin-leaved pondweeds	21.4	16	3.24	4.61		10.9	8.94	4.59	3.05	2.61	7.24	-14.14	2.63	4.63
Potamogeton pusillus	Small pondweed	18.4	13.2	2.59	1.64		10.3	8.94	4.26	2.03	1.95	6.58	-11.84	4.93	4.62
Eleocharis acicularis	Needle spikerush	5.92	4.18	5.5	1.64		3.22	6.95	9.84	4.41	5.21	6.25	0.33	4.61	1.04
Isoetes spp.	Quillwort spp.	3.95	2.79	5.5	3.29		0.64	1.32	6.89	6.1	10.4	8.55	4.61	5.26	-1.87
Nitella spp.	Stoneworts	2.3	2.79	0.65	2.63		0	0.66	0	2.03	7.49	5.26	2.96	2.63	-2.23
Potamogeton amplifolius, P.amp x praelongus	Large-leaf pondweed & Hybrid pondw	0.66	0.35	1.62	0.66		1.93	2.98	0	2.37	4.23	3.62	2.96	2.96	-0.62
Potamogeton amplifolius	Large-leaf pondweed	3.62	3.83	0.65	0.66		0.96	0.33	0	2.37	2.28	0.33	-3.29	-0.33	-1.95
Potamogeton hybrid 1	Pondweed Hybrid 1	0.66	0.35	0.97	0		1.93	2.65	0	0	1.95	3.29	2.63	3.29	1.34
Potamogeton friesii	Fries' pondweed	9.21	3.48	0.65	0.33		0	0	0	0	0	0.66	-8.55	0.33	0.66
													-157.89	-70.39	-95.92
Ranunculus aquatilis	White water crowfoot	0.66	0.35	0	0		0	0	0	0	0.65	0			
Myriophyllum tenellum	Dwarf watermilfoil	0.99	0	0	0		0	0	0	0	0	0.33			
Potamogeton strictifolius	Stiff pondweed	0	0.35	0.32	2.63		0.64	0	0.66	1.02	0.65	0			
Potamogeton hybrid 2	Pondweed Hybrid 2	0	0	0	0		0	3.64	0	0	0	0			
Sagittaria sp. (rosette)	Arrowhead sp. (rosette)	1.97	0.7	0	0		0.32	0.33	0	0	0	0			
Juncus pelocarpus	Brown-fruited rush	0	0	0.32	1.64		0	0	0.66	0.34	0	0			
Sagittaria graminea	Grass-leaved arrowhead	1.32	0.35	0	0		0	0	0	0	0	0			
Stuckenia pectinata	Sago pondweed	0	0	0.32	0		0	0	0.33	0	0	0			
Potamogeton illinoensis	Illinois pondweed	0	0	0	0		0	0	0	0.34	0	0			
Eriocaulon aquaticum	Pipewort	0	0.35	0	0		0	0	0	0	0	0			
Elatine minima	Waterwort	0	0.35	0	0		0	0	0	0	0	0			

Indicates Plants which average less than 1% LFOO and thus are excluded from analysis as their relevance to the entire plant population is not material

Indicates the year in which the native plant species exhibits the highest LFOO% for the period 2008 - 2018.

Indicates positive growth trends for native plants for the period evaluated.