B

APPENDIX B

Stakeholder Survey Response Charts and Comments

Phillips Chain O'Lakes - Anonymous Stakeholder Survey

Surveys Distributed: 463
Surveys Returned: 161
Response Rate: 35%

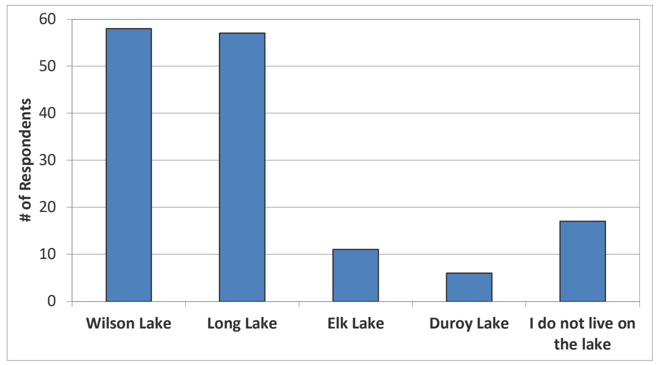
Phillips Chain O'Lakes Property

1. Do you own or rent your property on the Phillips Chain O'Lakes?

Answer Options	Response Percent	Response Count
Own	87.6%	127
Rent	12.4%	18
	answered question	**145**
	skipped question	**16**

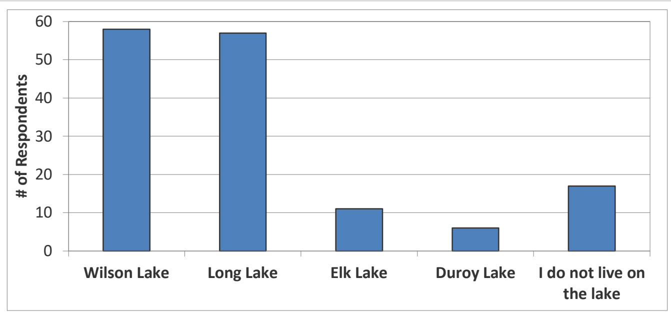
2. On which lake is your Phillips Chain O'Lakes property located?

Answer Options	Response Percent	Response Count
Wilson Lake	38.9%	58
Long Lake	38.3%	57
Elk Lake	7.4%	11
Duroy Lake	4.0%	6
I do not live on the lake	11.4%	17
	answered question	**149**
	skipped question	**12**



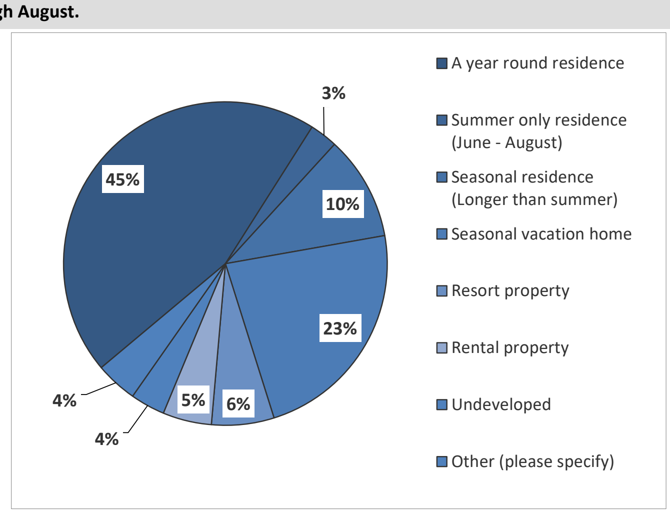
3. In which municipality is your Phillips Chain O'Lakes property located?

Answer Options	Response Percent	Response Count
City of Phillips	20.0%	29
Town of Worcester	6.2%	9
Town of Elk	73.8%	107
	answered question	**145**
	skipped question	**16**



4. Which category best describes how your property on the Phillips Chain O'Lakes is utilized? Consider a residence to be your primary home during that time and a vacation home to be used on weekends or occasional weeks. Summer is defined as June through August.

Answer Options	Response Percent	Response Count
A year round residence	45.1%	65
Summer only residence (June - August)	2.8%	4
Seasonal residence (Longer than summer)	10.4%	15
Seasonal vacation home	22.9%	33
Resort property	6.3%	9
Rental property	4.9%	7
Undeveloped	3.5%	5
Other (please specify)	4.2%	6
	answered question	**144**
	skipped question	**17**

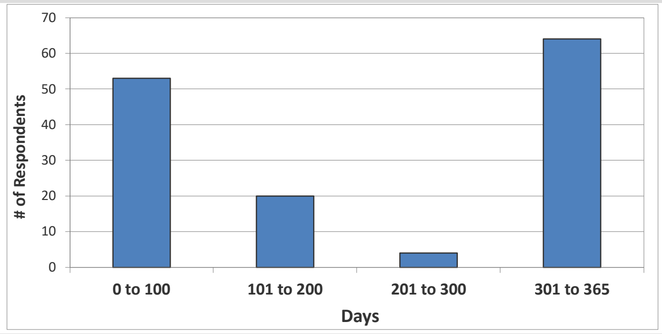


Number	Other (please specify)
1	365
2	Group camping sites
3	Parks and Boat Landings
4	Boat launch
5	year round vacation home
6	Lakefront Developed Lot

5. How many days each year is your property used by you or others?

Answer Options	Response Count
answered question	141
skipped question	20

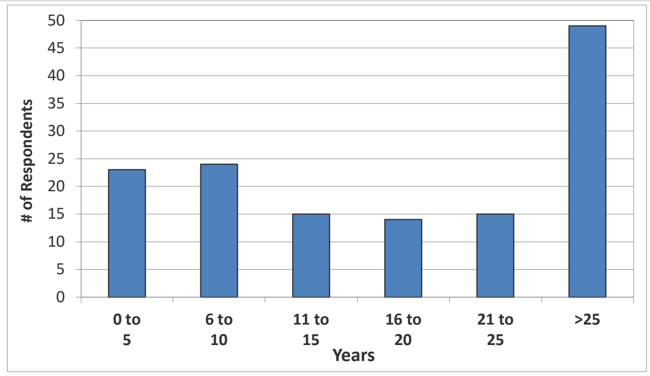
Category (# of days)	Responses	% Response
0 to 100	53	38%
101 to 200	20	14%
201 to 300	4	3%
301 to 365	64	45%



6. How long have you owned or rented your property on the Phillips Chain O'Lakes? If less than one year please respond with "1".

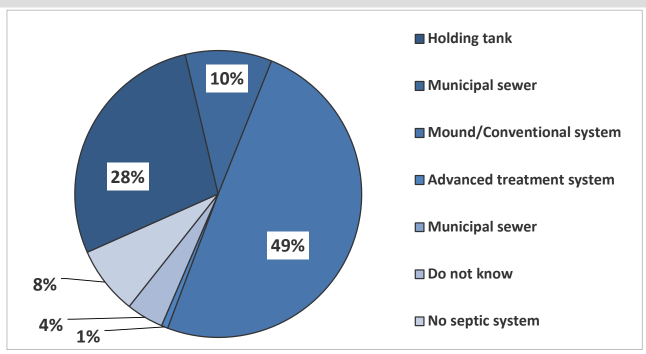
Answer Options	Response Count
	140
answered question	140
skipped question	21

Category (# of years)	Responses	% Response
0 to 5	23	16%
6 to 10	24	17%
11 to 15	15	11%
16 to 20	14	10%
21 to 25	15	11%
>25	49	35%



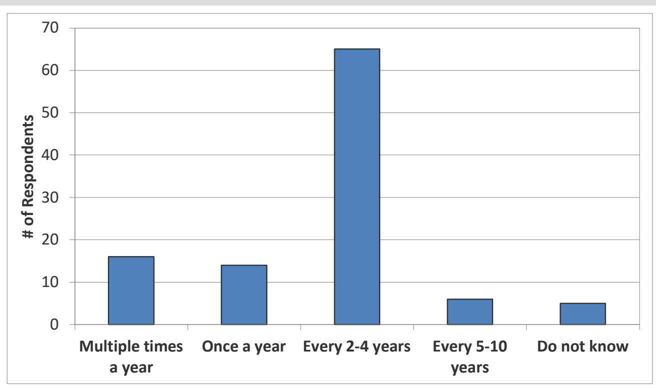
7. What type of sewer system does your property utilize?

Answer Options	Response Percent	Response Count
Holding tank	28.0%	40
Municipal sewer	9.8%	14
Mound/Conventional system	49.7%	71
Advanced treatment system	0.7%	1
Municipal sewer	0.0%	0
Do not know	4.2%	6
No septic system	7.7%	11
	answered question	143
	skipped question	18



8. How often is the sewer system on your property pumped?

Answer Options	Response Percent	Response Count
Multiple times a year	15.1%	16
Once a year	13.2%	14
Every 2-4 years	61.3%	65
Every 5-10 years	5.7%	6
Do not know	4.7%	5
	answered question	106
	skipped question	55

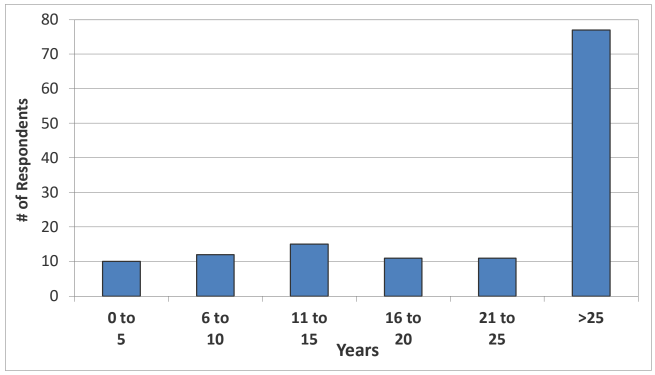


Recreational Activity on Phillips Chain O'Lakes

9. How many years ago did you first visit the Phillips Chain O'Lakes? Please answer in approximate number of years.

Answer Options	Response Count
	136
answered question	**136**
skipped question	**25**

Category (# of years)	Responses	% Response
0 to 5	**10**	**7%**
6 to 10	**12**	**9%**
11 to 15	**15**	**11%**
16 to 20	**11**	**8%**
21 to 25	**11**	**8%**
>25	**77**	**57%**

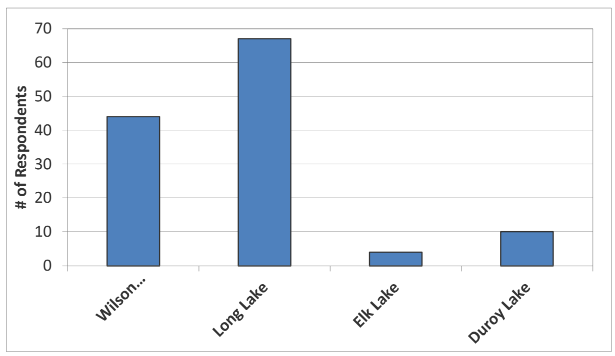


10. Have you personally fished on the Phillips Chain O'Lakes in the past three years?

Answer Options	Response Percent	Response Count
Yes	88.7%	126
No	11.3%	16
	answered question	**142**
	skipped question	**19**

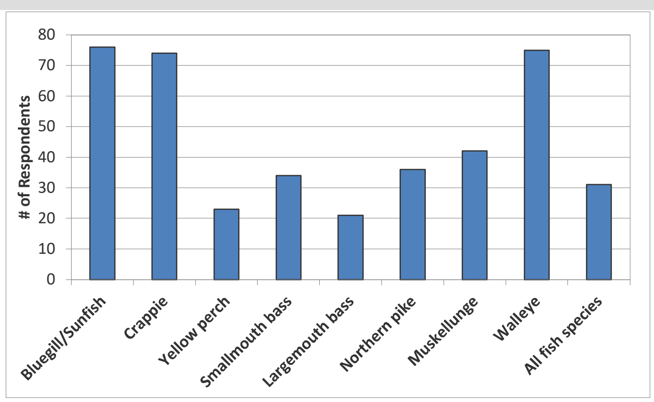
11. Which chain lake do you fish on the most?

Answer Options	Response Percent	Response Count
Wilson Lake	35.2%	44
Long Lake	53.6%	67
Elk Lake	3.2%	4
Duroy Lake	8.0%	10
	answered question	**125**
	skipped question	**36**



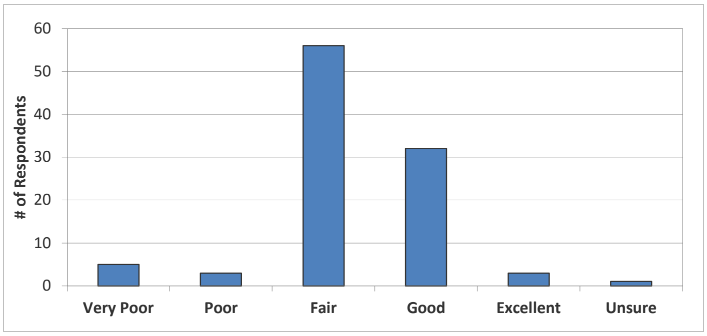
12. What species of fish do you like to catch on the Phillips Chain O'Lakes?

Answer Options	Response Percent	Response Count
Bluegill/Sunfish	59.8%	76
Crappie	58.3%	74
Yellow perch	18.1%	23
Smallmouth bass	26.8%	34
Largemouth bass	16.5%	21
Northern pike	28.4%	36
Muskellunge	33.1%	42
Walleye	59.1%	75
All fish species	24.4%	31
Other (please specify)	0.8%	1
	answered question	**127**
	skipped question	**34**



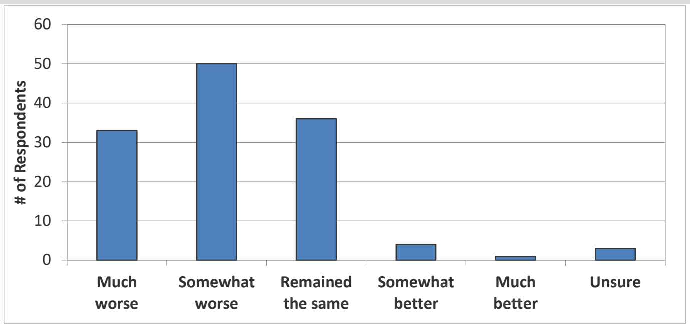
13. How would you describe the current quality of fishing on the Phillips Chain O'Lakes?

Answer Options	Very Poor	Poor	Fair	Good	Excellent	Unsure	Response Count
	5	3	56	32	3	1	127
						answered question	127
						skipped question	34



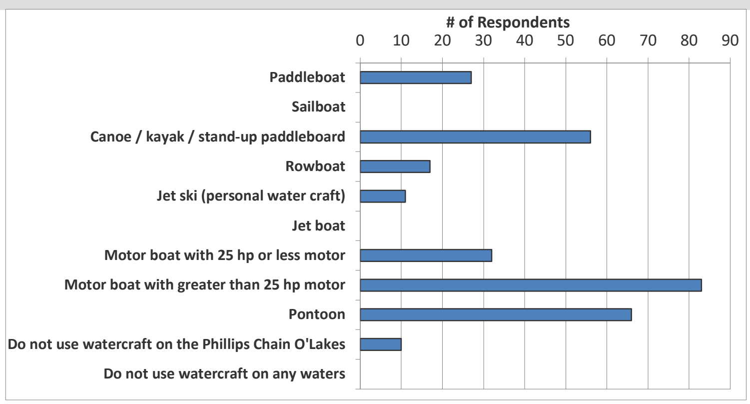
14. How has the quality of fishing changed on the Phillips Chain O'Lakes since you have started fishing the lake?

Answer Options	Much worse	Somewhat worse	Remained the same	Somewhat better	Much better	Unsure	Response Count
	33	50	36	4	1	3	127
						answered question	127
						skipped question	34



15. What types of watercraft do you currently use on the Phillips Chain O'Lakes?

Answer Options	Response Percent	Response Count
Paddleboat	18.8%	27
Sailboat	0.0%	0
Canoe / kayak / stand-up paddleboard	38.9%	56
Rowboat	11.8%	17
Jet ski (personal water craft)	7.6%	11
Jet boat	0.0%	0
Motor boat with 25 hp or less motor	22.2%	32
Motor boat with greater than 25 hp motor	57.6%	83
Pontoon	45.8%	66
Do not use watercraft on the Phillips Chain O'Lakes	6.9%	10
Do not use watercraft on any waters	0.0%	0
	answered question	144
	skipped question	17



16. Do you use your watercraft on waters other than the Phillips Chain O'Lakes?

Answer Options	Response Percent	Response Count
Yes	46.0%	63
No	54.0%	74
	answered question	137
	skipped question	24

17. What is your typical cleaning routine after using your watercraft on waters other than the Phillips Chain O'Lakes?

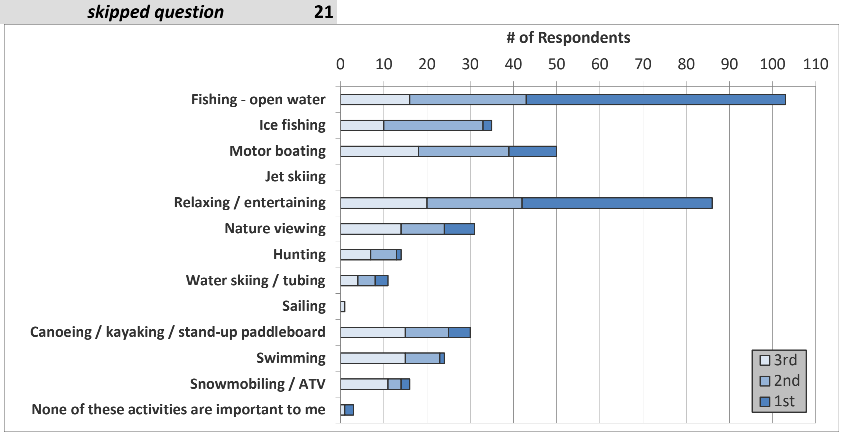
Answer Options	Response Percent	Response Count
Remove aquatic hitch-hikers (ex. - plant material, clams, mussels)	84.1%	58
Drain bilge	73.9%	51
Rinse boat	36.2%	25
Power wash boat	14.5%	10
Apply bleach	1.5%	1
Air dry boat for 5 or more days	42.0%	29
Do not clean boat	1.5%	1
Other (please specify)		4
	answered question	69
	skipped question	92

Number	Other (please specify)
1	Do not use it on any other lakes
2	Stays on Phillips chain
3	boat only used on phillips chain and no where else
4	boats stay on the chain

18. Please rank up to three activities that are important reasons for owning or renting your property on the Phillips Chain O'Lakes. Select your top activities below with 1st being the most important.

Answer Options	1st	2nd	3rd	Rating Average	Response Count
Fishing - open water	60	27	16	1.57	103
Ice fishing	2	23	10	2.23	35
Motor boating	11	21	18	2.14	50
Jet skiing	0	0	0	0	0
Relaxing / entertaining	44	22	20	1.72	86
Nature viewing	7	10	14	2.23	31
Hunting	1	6	7	2.43	14
Water skiing / tubing	3	4	4	2.09	11
Sailing	0	0	1	3	1
Canoeing / kayaking / stand-up paddleboard	5	10	15	2.33	30
Swimming	1	8	15	2.58	24
Snowmobiling / ATV	2	3	11	2.56	16
None of these activities are important to me	2	0	1	1.67	3
Other (please specify below)	2	0	0	1	2
				answered question	140
				skipped question	21

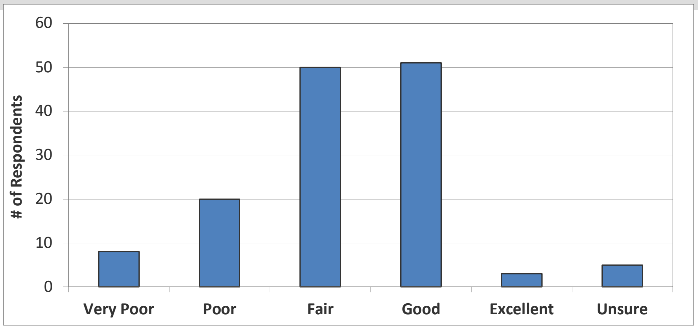
Number	"Other" responses
1	Enjoy where we are!
2	We also grew up in Price County
3	agriculture



Phillips Chain O'Lakes Current and Historic Condition, Health and Management

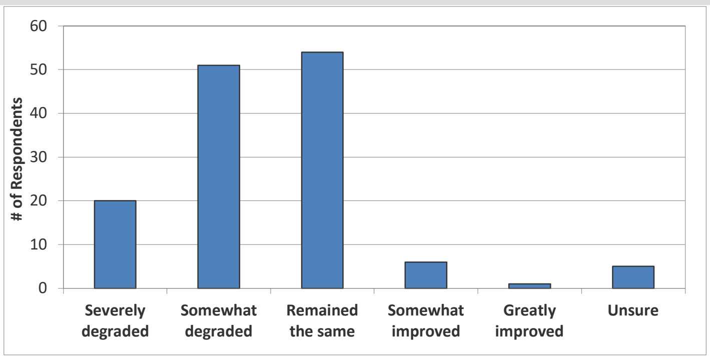
19. How would you describe the overall current water quality of the Phillips Chain O'Lakes?

Answer Options	Very Poor	Poor	Fair	Good	Excellent	Unsure	Response Count
	8	20	50	51	3	5	137
						answered question	**137**
						skipped question	**24**



20. How has the overall water quality changed in the Phillips Chain O'Lakes since you first visited the lake?

Answer Options	Severely degraded	Somewhat degraded	Remained the same	Somewhat improved	Greatly improved	Unsure	Response Count
	20	51	54	6	1	5	137
						answered question	**137**
						skipped question	**24**



21. Considering how you answered the questions above, what do you think of when describing water quality?

Answer Options	Response Percent	Response Count
Water clarity (clearness of water)	56.2%	77
Aquatic plant growth (not including algae blooms)	72.3%	99
Water color	32.9%	45
Algae blooms	51.1%	70
Smell	21.9%	30
Water level	25.6%	35
Fish kills	17.5%	24
Other (please specify)	5.1%	7
	answered question	**137**
	skipped question	**24**

22. Based on your answer above, which of the following would you say is the single most important aspect when considering water quality?

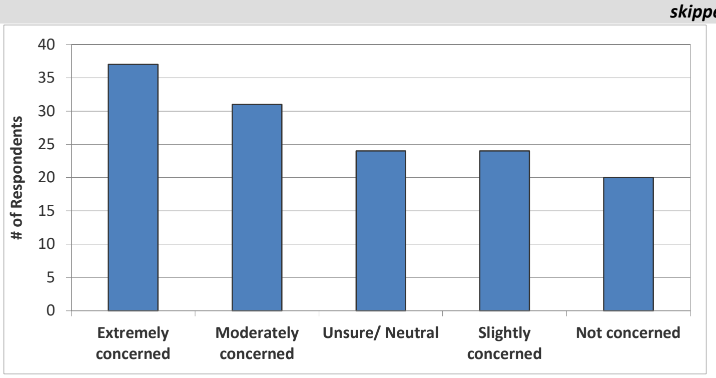
Answer Options	Response Percent	Response Count
Water clarity (clearness of water)	22.1%	30
Aquatic plant growth (not including algae blooms)	41.2%	56
Water color	0.7%	1
Algae blooms	15.4%	21
Smell	2.9%	4
Water level	7.4%	10
Fish kills	4.4%	6
Other (please specify)	5.9%	8
	answered question	**136**
	skipped question	**25**

23. Currently the State of Wisconsin requires all buildings and structures to be setback a minimum of 75-feet from the ordinary high-water mark of navigable lakes. Do you believe the current setback of 75-feet is too restrictive?

Answer Options	Response Percent	Response Count
Yes	35.3%	48
No	54.4%	74
Unsure	10.3%	14
	answered question	**136**
	skipped question	**25**

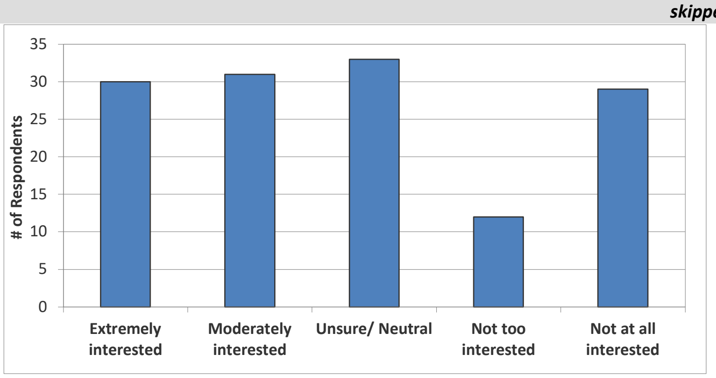
24. How concerned, if at all, are you about shoreline erosion?

Answer Options	Extremely concerned	Moderately concerned	Unsure/ Neutral	Slightly concerned	Not concerned	Response Count
	37	31	24	24	20	136
					answered question	**136**
					skipped question	**25**



25. Would you be interested in receiving assistance with installing rock rip-rap to protect private landowner shoreline?

Answer Options	Extremely interested	Moderately interested	Unsure/ Neutral	Not too interested	Not at all interested	Response Count
	30	31	33	12	29	135
					answered question	**135**
					skipped question	**26**



26. Before reading the statement above, had you ever heard of aquatic invasive species?

Answer Options	Response Percent	Response Count
Yes	97.8%	132
No	2.2%	3
	answered question	**135**
	skipped question	**26**

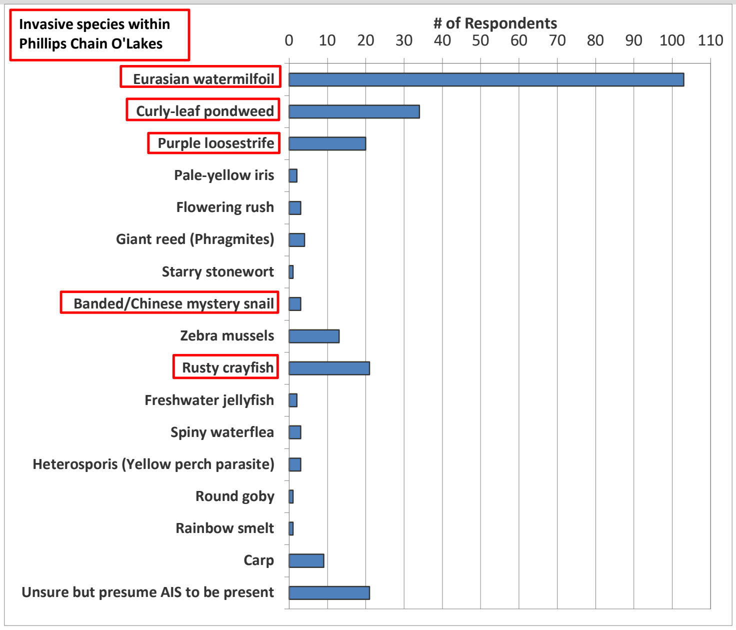
27. Do you believe aquatic invasive species are present within the Phillips Chain O'Lakes?

Answer Options	Response Percent	Response Count
Yes	72.7%	96
I think so but am not certain	21.2%	28
No	6.1%	8
	answered question	**132**
	skipped question	**29**

28. Which aquatic invasive species do you believe are in the Phillips Chain O'Lakes?

Answer Options	Response Percent	Response Count
Eurasian watermilfoil	84.4%	103
Curly-leaf pondweed	27.9%	34
Purple loosestrife	16.4%	20
Pale-yellow iris	1.6%	2
Flowering rush	2.5%	3
Giant reed (Phragmites)	3.3%	4
Starry stonewort	0.8%	1
Banded/Chinese mystery snail	2.5%	3
Zebra mussels	10.7%	13
Rusty crayfish	17.2%	21
Freshwater jellyfish	1.6%	2
Spiny waterflea	2.5%	3
Heterosporis (Yellow perch parasite)	2.5%	3
Round goby	0.8%	1
Rainbow smelt	0.8%	1
Carp	7.4%	9
Unsure but presume AIS to be present	17.2%	21
Other (please specify)	1.6%	2
	answered question	122
	skipped question	39

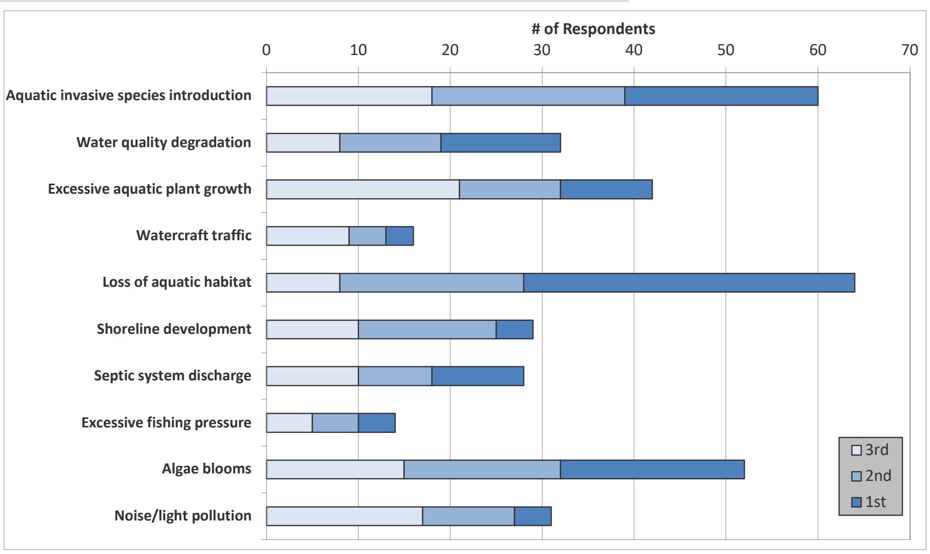
Number	"Other" responses
1	Wild Rice on north end of Duroy
2	Lily pads



29. From the list below, please rank your top three concerns regarding the Phillips Chain O'Lakes, with the 1st being your greatest concern.

Answer Options	1st	2nd	3rd	Response Count
Water quality degradation	21	21	18	60
Loss of aquatic habitat	13	11	8	32
Shoreline erosion	10	11	21	42
Shoreline development	3	4	9	16
Aquatic invasive species introduction	36	20	8	64
Excessive watercraft traffic	4	15	10	29
Unsafe watercraft practices	10	8	10	28
Excessive fishing pressure	4	5	5	14
Excessive aquatic plant growth (excluding algae)	20	17	15	52
Algae blooms	4	10	17	31
Septic system discharge	3	6	2	11
Noise/light pollution	0	1	0	1
Other (please specify)	4	0	2	6
			answered question	132
			skipped question	29

Number	"Other" responses
1	Not sure
2	Spearing
3	jet ski traffic
4	The boats designed to make big waves for jumping
5	Wakeboard boats that are makeing huge artifical waves that endanger boaters and are causing major shoreline changes
6	Water level
7	Quality of fishing (including the quantity of fish)
8	shallow spots that impede traffic flow
9	silt is preventing me from lake access
10	unable to use the water related to the weeds
11	Waterlevels



30. Do you believe you are able to identify Eurasian watermilfoil?

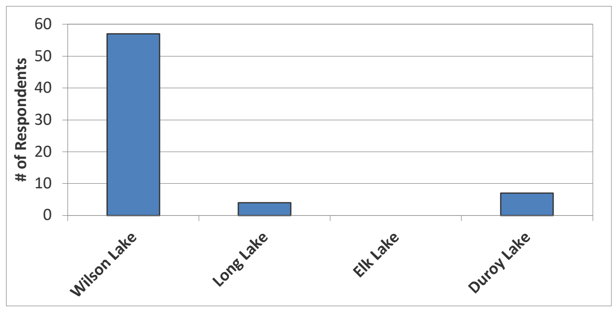
Answer Options	Response Percent	Response Count
Yes	57.1%	76
I think so but am not certain	28.6%	38
No	14.3%	19
	answered question	133
	skipped question	28

31. Eurasian watermilfoil, a non-native aquatic plant, has been found in all four lakes on the Phillips Chain. Are there any lakes in the Phillips Chain O'Lakes where you have decreased your recreation time on the lake because of Eurasian watermilfoil?

Answer Options	Response Percent	Response Count
Yes	59.7%	68
No	40.4%	46
	answered question	114
	skipped question	47

32. If you answered yes to Question 31, on which lakes have you decreased your recreation time due to Eurasian watermilfoil?

Answer Options	Response Percent	Response Count
Wilson Lake	83.8%	57
Long Lake	5.9%	4
Elk Lake	0.0%	0
Duroy Lake	10.3%	7
	answered question	125
	skipped question	36

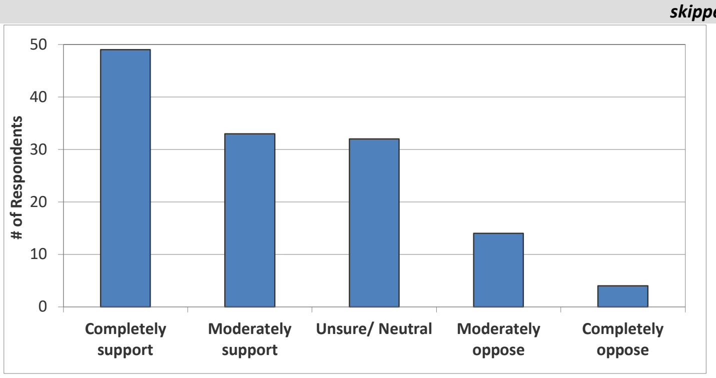


33. Before the present year, aquatic herbicides have been used to manage Eurasian watermilfoil on the Phillips Chain O'Lakes. Professional monitoring of the aquatic plant community has also occurred during this time. Prior to reading this information, did you know that aquatic herbicides were being applied in the Phillips Chain O'Lakes to help manage Eurasian watermilfoil?

Answer Options	Response Percent	Response Count
Yes	68.9%	91
I think so but can't say for certain	9.1%	12
No	22.0%	29
	answered question	132
	skipped question	29

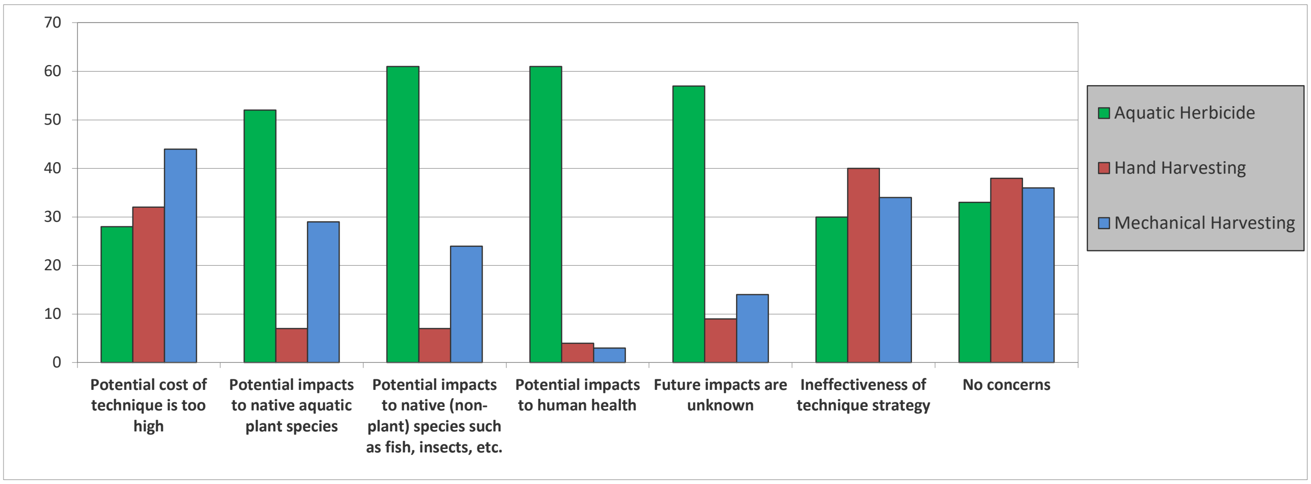
34. How do you feel about the past use of herbicides to treat Eurasian watermilfoil in previous years?

Answer Options	Completely support	Moderately support	Unsure/ Neutral	Moderately oppose	Completely oppose	Response Count
	49	33	32	14	4	132
					answered question	132
					skipped question	29



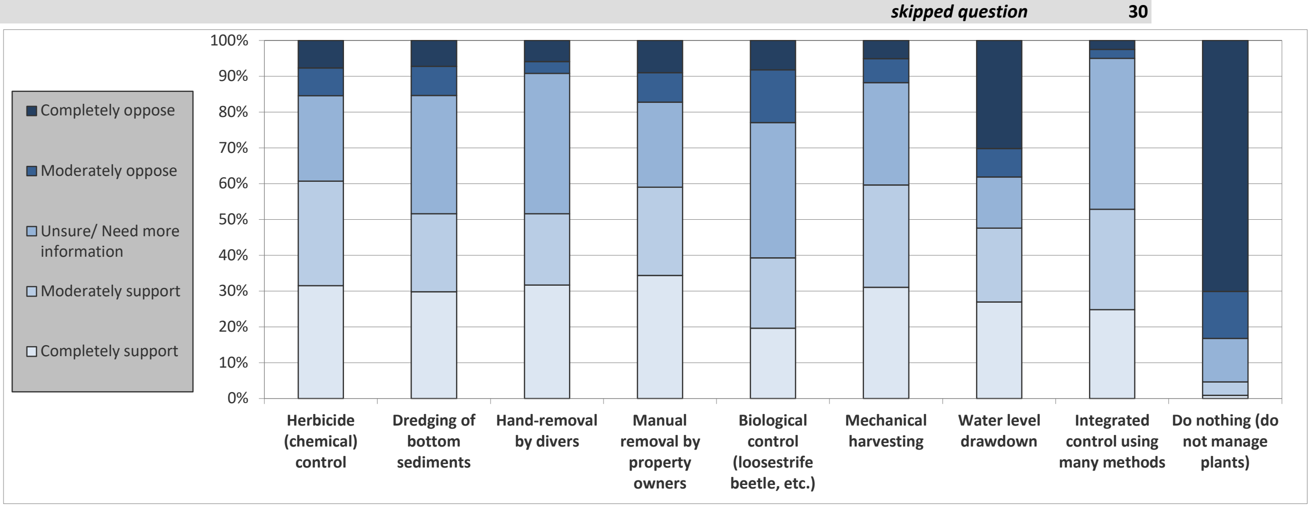
35. What concerns, if any, do you have for the future use of aquatic herbicides, hand harvesting and/or mechanical harvesting to target Eurasian watermilfoil in the Phillips Chain O'Lakes?

Answer Options	Aquatic Herbicide	Hand Harvesting	Mechanical Harvesting	Response Count
Potential cost of technique is too high	28	32	44	70
Potential impacts to native aquatic plant species	52	7	29	69
Potential impacts to native (non-plant) species such as fish,	61	7	24	75
Potential impacts to human health	61	4	3	67
Future impacts are unknown	57	9	14	65
Ineffectiveness of technique strategy	30	40	34	67
No concerns	33	38	36	59
			answered question	122
			skipped question	39



36. Aquatic invasive plants can be controlled using many techniques. What is your level of support or opposition for the responsible use of the following aquatic invasive plant management techniques on the Phillips Chain O'Lakes?

Answer Options	Completely support	Moderately support	Unsure/ Need more information	Moderately oppose	Completely oppose	Rating Average	Response Count
Herbicide (chemical) control	41	38	31	10	10	2.31	130
Dredging of bottom sediments	37	27	41	10	9	2.41	124
Hand-removal by divers	38	24	47	4	7	2.32	120
Manual removal by property owners	42	30	29	10	11	2.33	122
Biological control (loosestrife beetle, etc.)	24	24	46	18	10	2.72	122
Mechanical harvesting	37	34	34	8	6	2.26	119
Water level drawdown	34	26	18	10	38	2.94	126
Integrated control using many methods	30	34	51	3	3	2.3	121
Do nothing (do not manage plants)	1	4	13	14	75	4.48	107
						answered question	131
						skipped question	30

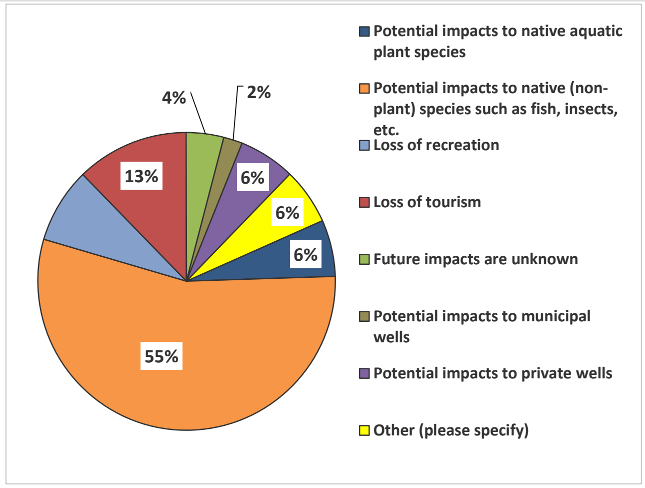


37. The Phillips Chain of Lakes is considering a water level drawdown. This means the water level would be lowered 6ft below normal water levels. The drawdown would take place during the winter months beginning approximately Labor Day and would refill prior to Memorial Day. The goal of the drawdown is to manage Eurasian watermilfoil. Would you support a water level drawdown on the Phillips Chain O'Lakes?

Answer Options	Response Percent	Response Count
Yes	42.9%	57
I think so but can't say for certain	19.6%	26
No	37.6%	50
	answered question	133
	skipped question	28

38. If you selected "No" in Question #37, what is the reason or reasons you would oppose a water level drawdown to the Phillips Chain O'Lakes?

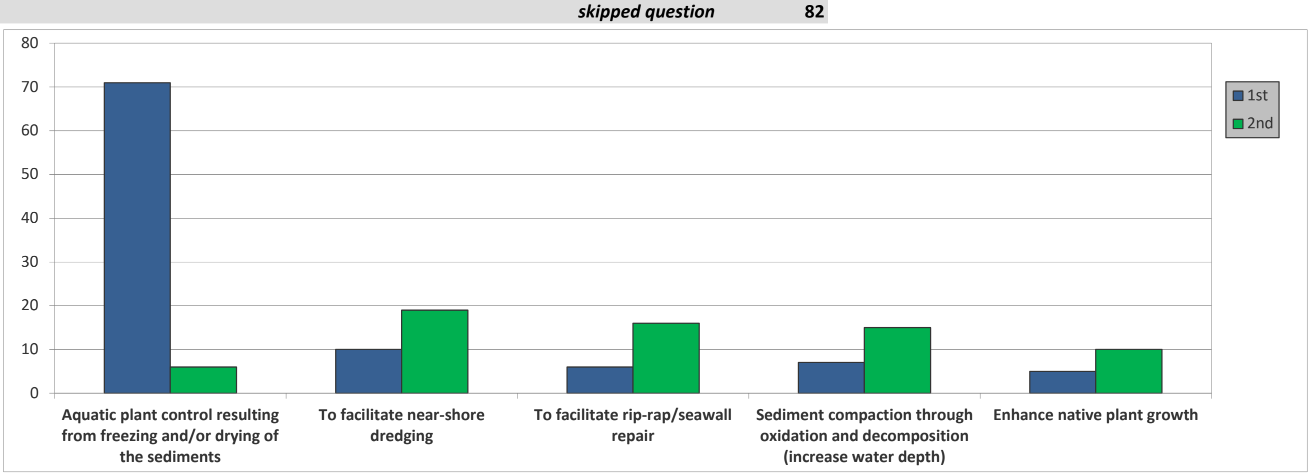
Answer Options	Response Percent	Response Count
Potential impacts to native aquatic plant species	6.1%	3
Potential increases in non-native aquatic plant species	0.0%	0
Potential impacts to native (non-plant) species such as fish, insects, etc.	55.1%	27
Navigation issues	0.0%	0
Loss of recreation	8.2%	4
Loss of tourism	12.2%	6
Future impacts are unknown	4.1%	2
Potential impacts to municipal wells	2.0%	1
Potential impacts to private wells	6.1%	3
Other (please specify)	6.1%	3
	answered question	49
	skipped question	112



Number	"Other" responses
1	6 feet is too much. Last draw down moved/removed several structures when water levels were restored.
2	All of the above!
3	Nav issue, loss if rec,potential imp to both above, future impacts or results are unkniwn

39. If you selected "Yes" or "I think so but would need more information" in Question #37, what are your top two reasons for wanting to complete a drawdown?

Answer Options	1st	2nd	Rating Average	Response Count
Aquatic plant control resulting from freezing and/or drying o	71	6	1.08	77
To facilitate near-shore dredging	10	19	1.66	29
To facilitate rip-rap/seawall repair	6	16	1.73	22
Sediment compaction through oxidation and decompositior	7	15	1.68	22
Enhance native plant growth	5	10	1.67	15
Other (please specify)				2
			answered question	79
			skipped question	82



Phillips Chain O'Lakes Association

40. Before receiving this mailing, have you ever heard of the Phillips Chain O'Lakes Associaiton?

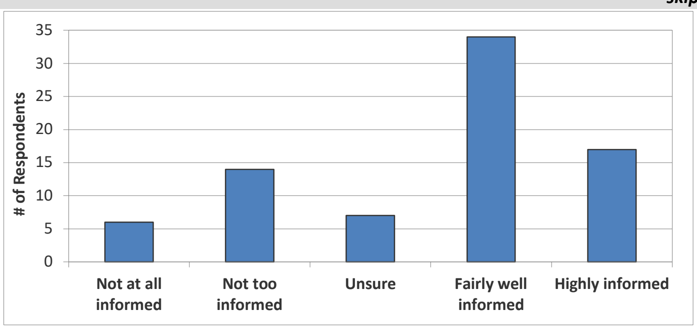
Answer Options	Response Percent	Response Count
Yes	98.5%	127
No	1.6%	2
	answered question	**129**
	skipped question	**32**

41. What is your membership status with the Phillips Chain O'Lakes Association?

Answer Options	Response Percent	Response Count
Current member	50.0%	64
Former member	10.9%	14
Never been a member	39.1%	50
	answered question	**128**
	skipped question	**33**

42. How informed has (or had) the Phillips Chain O'Lakes Association kept you regarding issues with the Phillips Chain O'Lakes and its management?

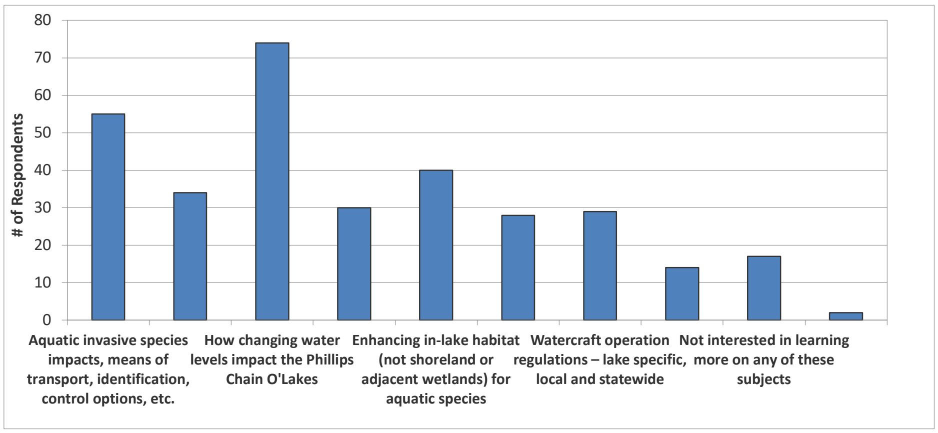
Answer Options	Not at all informed	Not too informed	Unsure	Fairly well informed	Highly informed	Response Count
	6	14	7	34	17	78
					answered question	**78**
					skipped question	**83**



43. Stakeholder education is an important component of every lake management planning effort. Which of these subjects would you like to learn more about?

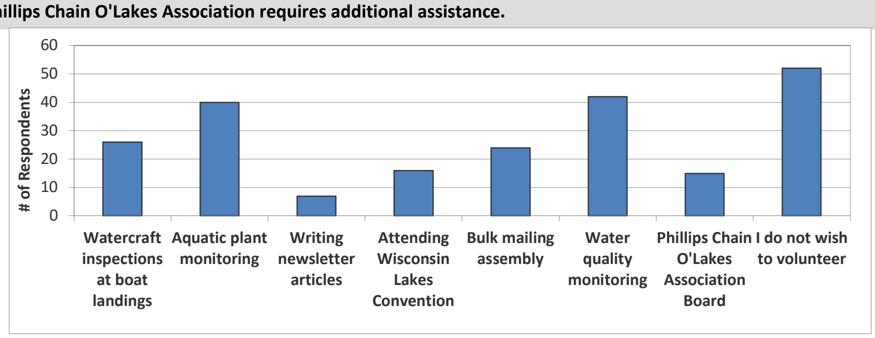
Answer Options	Response Percent	Response Count
Aquatic invasive species impacts, means of transport, identification, control options, etc.	45.5%	55
How to be a good lake steward	28.1%	34
How changing water levels impact the Phillips Chain O'Lakes	61.2%	74
Social events occurring around the Phillips Chain O'Lakes	24.8%	30
Enhancing in-lake habitat (not shoreland or adjacent wetlands) for aquatic species	33.1%	40
Ecological benefits of shoreland restoration and preservation	23.1%	28
Watercraft operation regulations – lake specific, local and statewide	24.0%	29
Volunteer lake monitoring opportunities (Clean Boats Clean Waters, Citizens Lake Monitoring Network, Loon Watch, the Phillips Chain O'Lakes Association programs, etc.)	11.6%	14
Not interested in learning more on any of these subjects	14.1%	17
Other (please specify)	1.7%	2
	answered question	121
	skipped question	40

Number	Other (please specify)
1	I haven't been on the lake in 12 years
2	How to decrease use of overly large boats on the lake, creating 3+ foot wakes and boaters not abiding by the no wake law when driving both too close to shore and/or through narrow passage areas such as Duroy to Long lake, while maintaining high speeds and significant wakes, without being mindful of other lake users. Being a good lake steward starts at the top. Permanent residing lake shore owners should lead by example to promote proper boating techniques and assist learning by people not residing in Price County full-time.
3	



44. Please note that because this survey is anonymous, your answer to this question will not be regarded as a commitment to participate, but instead will be used to gauge potential participation of stakeholders in the Phillips Chain O'Lakes Association.The effective management of your lake will require the cooperative efforts of numerous volunteers. Please circle the activities you would be willing to participate in if the Phillips Chain O'Lakes Association requires additional assistance.

Answer Options	Response Percent	Response Count
Watercraft inspections at boat landings	21.3%	26
Aquatic plant monitoring	32.8%	40
Writing newsletter articles	5.7%	7
Attending Wisconsin Lakes Convention	13.1%	16
Bulk mailing assembly	19.7%	24
Water quality monitoring	34.4%	42
Phillips Chain O'Lakes Association Board	12.3%	15
I do not wish to volunteer	42.6%	52
	answered question	122
	skipped question	39



45. Please feel free to provide written comments concerning the Phillips Chain O'Lakes, its current and/or historic condition and its management.

Answer Options	Response Count
	39
answered question	39
skipped question	122

Number	Response Text
1	Would like the lakes monitored for speed, jet skis to close to shore and docks. Do not like the idea of a bridge on highway W! All we need and larger boats. I am using shoreline as it is. The lake needs a "No Wake" at places.
2	Question 14 - More panfish less walleye. Question 22 - Invasive species Question 28 - Eurasian watermilfoil Wilson only Question 35 - Concern of effectivness Question 38 - Concern of the fishery in Wilson shallow drawdown is questionable depending on weather conditions.
3	Own property but not full time resident. Plan on joining PCOLA this year/ Note- have a renter who is full time.
4	I hope to hear the outcome of this survey.
5	Property has been in the family before the chain-of-lakes was formed. We try to pull as many invasive plants as we can every summer. I am concerned of the impact on the crayfish and clams of the lake drawdown, since they don't appear to have a good place to go, especially during the winter
6	High powered boats (over 50 hp) tear up the lake bottom, most lakes have a large amount of shallow (6' or less) areas. Fishing competitions bring a large number of high powered boats that tear up our lakes and bring with them invasive species and should be banned. Jet ski's are noisy, operate dangerously close to shore & each other and have a deep jet thrust that tears up lake bottoms. We have lost around 3' of shoreline in the last decade due to watercraft operating too close to shore and piers at high speeds. DNR does not seem to consider our lake a priority to monitor & catcl violators.
7	need more meetings to keep people informed
8	before chemical treatment on wilson lake, fishing was excellent. since fishing is poor. we now only fish long lake. this has put heavier fishing pressure on long lake.
9	Some of my responses were based on my kids usage of the lake as well as mine.
10	I am concerned with boat wake erosion of shoreline, especially when boats proceed at high speed near eroding shoreline. Perhaps there should be selected no wake zones or minimum distances from the shore.
11	A draw down will greatly harm the lake as far as fishing. It will impact the municipal wells. A drawdown because someone doesn't like weeds in front of their house or personal watercraft is foolish. The association needs to think about all wildlife species and not just themselves
12	Keep up the good work
13	See previous comment.
14	The decreased flows of the wilson creek from the sediments have greatly reduced the quality of fishing in the remaining lake.
15	get the entire town and area involved...these lakes are not your personal playground and are here for everyone not just lake shore property owners.
16	no reproduction on walleyes on willson lake.
17	It appears that Blake has his own initiatives and that is all he is concerned with. To many secretive agendas
18	Please don't drain the lake, I fish year round up there and spend a good about of money at local bait shops and restaurants, if the fishing declines I will go other places
19	Do something to restrict oversized ski boats, they are tearing up the shorelion
20	You are worried way to much about Weeds that have been here and are not going anywhere, some years the weeds cycle makes them grow more some years less. You need to focus more on Shoreline Damage from the Wake boats that are pushing 3 and 4 foot waves. Small resort boats have been swamped by these people in wake boats and if you can't see the damage you are doing you are blind.
21	It is a good group. We must take action to get rid of the all the invasive plants in the chain of lakes
22	We also have concerns about the water level and how it is maintained on the chain of lakes.
23	Weeds need to be dealt with before we loose something we can never get back
24	At one time we were sent a membership form and/or lake asso. fee/donation. I would not be opposed to a yearly membership fee to improve the chains fishing/water condition.
25	The property and cottage on the Wilson flowage has been in our family for over 40 years. Back in the seventies and eighties the fishing was spectacular and the milfoil was nonexistent. It seemed like after the first lake draw down to (freeze out the milfoil after it became a problem) helped for a couple years but decreased natural vegetation and the milfoil came back more aggressive and the fishing declined. The treatment of herbicides the few times that we are aware of did help the situation. The past several years the milfoil on the Wilson has been preventing its land owners from enjoying most water activities.
26	A concerted effort by property owners on the Phillips Chain of Lakes is long overdue. The lakes need our help they are dying a slow death as it relates to water quality and the reproduciton of game fish. Thank you for you efforts in creating this survey. Dave Botz
27	Only one or 2 of the lakes have invasive species that affect them. I suggest using the same methods to help control them as thats about all that can be done. Lowering the water level hurts the fish population and fishing has been terrible since the water levels have been adjusted. We see less fry on the shoreline as this pushes the small fish into deeper water which then become prey. I would say that the chain is the same as anywhere else and only need moderate control every few years.
28	Please get rid of the weeds.
29	Would like information on enlarging the tunnel between Wilson & Long Lake.
30	We have been past members and our membership has lapsed not due to lack of support but rather not receiving a renewal notice!
31	I feel dredging the south end of Wilson Lake during a draw down will help the fish habitat and the control of invasive species. The lake is so silted in I cannot use a boat to get to my property and dock by boat
32	To me, the Chain is going downhill...
33	Eurasian millfoil is returning after a successful use of treatment greatly reduced this invasive species. I am in favor of doing what seemed to work well in the past to combat this threat to the viability of our lake system.
34	would like to see signage installed to remind boaters to slow down their speed in the channel in front of Reese’s Resort. signs at the boat landing and by the tunnel are ineffective. People don’t know the speed limit is lower when you are less than 100 feet from shore. I feel there will have to be a severe accident before this is addressed. Law enforcement has expressed frustration monitoring this area.
35	we are trying to sell our property because the kids don't want to come fishing anymore because of the weeds, and no one wants to buy because of the lily pads and milfoil, we won't have water on our end very soon if nothing is done, we can't get a boat out, its much worse than when we bought.
36	Am all in favor of the association, but would like to be more informed of lake management. Periodically received emails and news letters, maybe that was all that was available.
37	I volunteered. You stated you didn't need my expertise. I left never to return.
38	When we first bought our place the lake was clear, A few weeds started to show up and as the PWC rammed around the lake and tore up weeds they floated to our shore line and started to grow and spread. Driving in this manner starts to stop!
39	too many bays choked with weeds hampering fishing and boating

C

APPENDIX C

Water Quality Data

	Secchi (feet)				Chlorophyll-*a* (µg/L)				Total Phosphorus (µg/L)			
	Growing Season		Summer		Growing Season		Summer		Growing Season		Summer	
Year	**Count**	**Mean**	**Count**	**Mean**	**Count**	**Mean**	**Count**	**Mean**	**Count**	**Mean**	**Count**	**Mean**
1996	2	4.0	1	4.0	2	12.6	1	12.4	2	37.0	1.0	41.0
1997	0		0		0		0		0		0.0	
1998	0		0		0		0		0		0.0	
1999	0		0		0		0		0		0.0	
2000	4	3.6	3	3.4	3	8.0	3	8.0	4	58.8	3.0	67.3
2019	4	2.9	3	2.6	4	12.6	3	16.0	4	48.8	3.0	54.1
All Years (Weighted)		3.4		3.1		11.0		12.0		50.4		57.9
SLDL Median				5.6				9.4				33.0
NLF Ecoregion Median				8.9				5.6				21.0

	Secchi (feet)				Chlorophyll-*a* (µg/L)				Total Phosphorus (µg/L)			
	Growing Season		Summer		Growing Season		Summer		Growing Season		Summer	
Year	**Count**	**Mean**	**Count**	**Mean**	**Count**	**Mean**	**Count**	**Mean**	**Count**	**Mean**	**Count**	**Mean**
1996	2	4.0	1	4.0	2	12.9	1	13.4	2	37.0	1.0	40.0
1997	0		0		0		0		0		0.0	
1998	0		0		0		0		0		0.0	
1999	0		0		0		0		0		0.0	
2000	9	3.8	4	3.5	3	8.7	3	8.7	4	63.5	3.0	74.7
2001	5	3.0	4	2.5	1	33.0	1	33.0	1	97.0	1.0	97.0
2002	5	3.6	4	3.6	0		0		0		0.0	
2003	2	3.4	1	3.3	0		0		0		0.0	
2004	4	3.9	1	3.5	0		0		0		0.0	
2005	3	3.7	2	3.6	0		0		0		0.0	
2019	4	3.0	2	2.4	5	12.3	3	18.8	5	54.0	3.0	59.1
2020									0		0.0	
All Years (Weighted)		3.5		3.2		13.3		16.1		57.9		67.3
DLDL Median				8.5				7.0				23.0
NLF Ecoregion Median				8.9				5.6				21.0

Year	Secchi (feet)				Chlorophyll-*a* (µg/L)				Total Phosphorus (µg/L)			
	Growing Season		Summer		Growing Season		Summer		Growing Season		Summer	
	Count	Mean	Count	Mean	Count	Mean	Count	Mean	Count	Mean	Count	Mean
1990	8	16.2	8	16.2	0		0		0		0.0	
1991	13	11.3	8	11.9	0		0		0		0.0	
1992	11	16.1	9	14.9	0		0		0		0.0	
1993	2	18.6	0		0		0		0		0.0	
1994	0		0		0		0		0		0.0	
1995	0		0		0		0		0		0.0	
1996	2	4.1	1	4.0	2	21.7	1	25.5	2	39.5	1.0	42.0
1997	0		0		0		0		0		0.0	
1998	0		0		0		0		0		0.0	
1999	3	3.6	0		0		0		0		0.0	
2000	4	4.8	3	4.8	3	2.7	3	2.7	4	62.8	3.0	72.0
2019	5	3.7	3	3.6	5	14.0	3	21.7	5	51.6	3.0	47.8
All Years (Weighted)		11.4		12.1		12.1		14.1		53.5		57.3
DLDL Median				8.5				7.0				23.0
NLF Ecoregion Median				8.9				5.6				21.0

	Secchi (feet)				Chlorophyll-*a* (µg/L)				Total Phosphorus (µg/L)			
	Growing Season		Summer		Growing Season		Summer		Growing Season		Summer	
Year	Count	Mean	Count	Mean	Count	Mean	Count	Mean	Count	Mean	Count	Mean
1996												
1997												
1998	9	2.6	5	2.0	0		0		0	0.0	0.0	0.0
1999	7	2.3	4	1.7	0		0		0		0.0	
2000	14	3.4	9	3.1	2	20.0	2	20.0	4	57.3	3.0	66.3
2001	16	3.1	11	2.6	5	23.5	3	29.7	5	58.2	3.0	61.7
2002	8	3.4	5	3.3	4	26.5	3	31.2	6	60.5	5.0	63.2
2003	5	3.5	2	2.5	3	30.2	2	39.1	4	53.8	2.0	74.0
2004	5	3.3	3	3.2	4	22.6	3	23.5	5	39.1	3.0	51.3
2005	6	3.5	3	3.2	4	24.9	3	25.4	4	63.5	3.0	67.0
2006	6	3.3	3	3.0	4	17.2	3	18.0	5	52.2	3.0	57.3
2007	4	4.0	2	3.8	3	11.7	2	12.3	4	38.5	2.0	42.5
2008	5	3.9	2	2.9	3	20.2	2	19.0	4	54.0	2.0	63.5
2019	5	3.9	3	3.2	5	13.3	3	16.8	5	43.2	3.0	47.3
All Years (Weighted)		3.3		2.8		20.9		23.6		52.1		59.6
SLDL Median				5.6				9.4				33.0
NLF Ecoregion Median				8.9				5.6				21.0

D

APPENDIX D

Point-Intercept Aquatic Macrophyte Survey Data

Duroy Lake

	Scientific Name	Common Name	LFOO (%) 2009	LFOO (%) 2019
Dicots	*Myriophyllum spicatum*	Eurasian water milfoil	19.3	16.7
	Ceratophyllum demersum	Coontail	14.9	24.2
	Myriophyllum heterophyllum	Various-leaved water milfoil	9.3	6.8
	Nuphar variegata	Spatterdock	3.1	5.3
	Nymphaea odorata	White water lily	3.7	3.8
	Bidens beckii	Water marigold	1.9	2.3
	Ceratophyllum echinatum	Spiny hornwort	0.0	3.0
	Utricularia vulgaris	Common bladderwort	0.0	0.8
	Callitriche hermaphroditica	Atumnal water starwort	0.0	0.8
Non-dicots	*Potamogeton crispus*	Curly-leaf pondweed	0.0	1.5
	Elodea canadensis	Common waterweed	24.8	9.1
	Sparganium fluctuans	Floating-leaf bur-reed	9.3	21.2
	Zizania spp.	Wild rice sp.	0.0	28.8
	Chara spp.	Muskgrasses	0.6	20.5
	Potamogeton zosteriformis	Flat-stem pondweed	9.9	6.8
	Najas flexilis	Slender naiad	3.1	12.1
	Potamogeton spirillus	Spiral-fruited pondweed	5.0	7.6
	Potamogeton natans	Floating-leaf pondweed	4.3	6.1
	Potamogeton epihydrus	Ribbon-leaf pondweed	3.1	5.3
	Potamogeton berchtoldii	Slender pondweed	0.0	7.6
	Nitella spp.	Stoneworts	1.2	5.3
	Elodea nuttallii	Slender waterweed	0.0	6.8
	Potamogeton pusillus	Small pondweed	2.5	2.3
	Sparganium eurycarpum	Common bur-reed	2.5	1.5
	Potamogeton obtusifolius	Blunt-leaved pondweed	3.1	0.8
	Potamogeton robbinsii	Fern-leaf pondweed	1.9	1.5
	Potamogeton amplifolius	Large-leaf pondweed	1.2	2.3
	Pontederia cordata	Pickerelweed	1.2	1.5
	Zizania palustris	Northern wild rice	1.2	0.0
	Sparganium emersum	Short-stemmed bur-reed	0.6	0.8
	Spirodela polyrhiza	Greater duckweed	0.0	0.8
	Schoenoplectus tabernaemontani	Softstem bulrush	0.6	0.0
	Potamogeton richardsonii	Clasping-leaf pondweed	0.6	0.0
	Potamogeton alpinus	Alpine pondweed	0.6	0.0
	Eleocharis palustris	Creeping spikerush	0.0	0.8
	Calla palustris	Water arum	0.0	0.8
	Fissidens spp. & Fontinalis spp.	Aquatic Moss	0.0	0.8
	Callitriche spp.	Water starwort spp.	0.0	5.3

Elk Lake

	Scientific Name	Common Name	LFOO (%) 2009	LFOO (%) 2019
Dicots	*Myriophyllum spicatum*	Eurasian water milfoil	0.0	0.0
	Nymphaea odorata	White water lily	3.9	7.4
	Ceratophyllum demersum	Coontail	0.8	3.7
	Myriophyllum heterophyllum	Various-leaved water milfoil	0.4	3.7
	Myriophyllum sibiricum	Northern water milfoil	0.0	3.7
	Nuphar variegata	Spatterdock	0.4	0.0
Non-dicots	*Potamogeton crispus*	Curly-leaf pondweed	0.0	0.0
	Potamogeton epihydrus	Ribbon-leaf pondweed	1.2	1.9
	Potamogeton robbinsii	Fern-leaf pondweed	0.0	3.7
	Elodea canadensis	Common waterweed	0.4	1.9
	Potamogeton spirillus	Spiral-fruited pondweed	0.0	1.9
	Potamogeton pusillus	Small pondweed	0.4	0.0
	Potamogeton amplifolius	Large-leaf pondweed	0.4	0.0
	Nitella spp.	Stoneworts	0.0	1.9
	Lemna turionifera	Turion duckweed	0.0	1.9
	Lemna minor	Lesser duckweed	0.4	0.0
	Filamentous algae	Filamentous algae	0.0	1.9

Long Lake

	Scientific Name	Common Name	LFOO (%)	
			2009	2019
Dicots	*Myriophyllum spicatum*	Eurasian water milfoil	1.3	0.0
	Nymphaea odorata	White water lily	2.5	1.5
	Ceratophyllum demersum	Coontail	0.9	2.3
	Myriophyllum heterophyllum	Various-leaved water milfoil	1.3	0.0
	Bidens beckii	Water marigold	0.3	0.8
	Nuphar variegata	Spatterdock	0.0	0.8
	Myriophyllum verticillatum	Whorled water milfoil	0.0	0.8
Non-dicots	*Potamogeton crispus*	Curly-leaf pondweed	0.0	0.0
	Elodea canadensis	Common waterweed	2.8	3.1
	Najas flexilis	Slender naiad	0.3	3.1
	Potamogeton spirillus	Spiral-fruited pondweed	0.6	1.5
	Spirodela polyrhiza	Greater duckweed	0.9	0.0
	Potamogeton zosteriformis	Flat-stem pondweed	0.9	0.0
	Potamogeton pusillus	Small pondweed	0.9	0.0
	Potamogeton epihydrus	Ribbon-leaf pondweed	0.3	1.5
	Potamogeton amplifolius	Large-leaf pondweed	0.9	0.0
	Lemna minor	Lesser duckweed	0.9	0.0
	Potamogeton robbinsii	Fern-leaf pondweed	0.3	0.8
	Potamogeton berchtoldii	Slender pondweed	0.0	1.5
	Potamogeton vaseyi	Vasey's pondweed	0.3	0.0
	Nitella spp.	Stoneworts	0.0	0.8
	Lemna trisulca	Forked duckweed	0.3	0.0

Wilson Lake

	Scientific Name	Common Name	LFOO (%) 2007	2011	2012	2014	2015	2019
Dicots	*Myriophyllum spicatum*	Eurasian watermilfoil	51.7	11.7	0.0	7.2	10.7	41.9
	Ceratophyllum demersum	Coontail	34.1	44.4	9.7	21.6	11.5	29.9
	Nymphaea odorata	White water lily	7.6	9.2	9.7	14.4	13.7	3.4
	Nuphar variegata	Spatterdock	0.0	2.6	2.0	4.8	0.0	1.7
	Ceratophyllum echinatum	Spiny hornwort	0.0	0.0	0.0	0.0	0.0	5.1
	Brasenia schreberi	Watershield	0.0	1.0	0.5	0.0	0.0	1.7
	Utricularia vulgaris	Common bladderwort	0.5	0.0	0.5	0.0	0.0	0.9
	Myriophyllum sibiricum	Northern watermilfoil	0.9	0.5	0.0	0.8	0.0	0.0
	Myriophyllum heterophyllum	Various-leaved watermilfoil	0.0	0.0	0.0	0.0	0.0	0.9
	Myriophyllum verticillatum	Whorled watermilfoil	0.5	0.0	0.0	0.0	0.0	0.0
Non-dicots	*Potamogeton crispus*	Curly-leaf pondweed	0.0	0.0	0.0	0.0	0.0	0.0
	Potamogeton robbinsii	Fern-leaf pondweed	8.5	33.7	14.3	20.0	15.3	12.0
	Elodea canadensis & Elodea nuttallii	Common & Slender waterw	34.1	9.2	4.6	9.6	7.6	21.4
	Elodea canadensis	Common waterweed	34.1	9.2	4.6	4.8	7.6	18.8
	Filamentous algae	Filamentous algae	21.3	5.1	1.0	14.4	15.3	12.0
	Potamogeton pusillus	Small pondweed	5.7	5.6	0.5	0.8	0.0	35.9
	Potamogeton zosteriformis	Flat-stem pondweed	3.3	12.2	1.0	12.8	0.0	8.5
	Potamogeton amplifolius	Large-leaf pondweed	5.7	4.6	7.7	10.4	4.6	5.1
	Chara spp.	Muskgrasses	0.0	3.1	0.0	8.0	5.3	9.4
	Najas flexilis	Slender naiad	1.4	3.1	0.0	4.0	0.0	12.0
	Nitella spp.	Stoneworts	0.9	4.1	6.1	5.6	4.6	1.7
	Potamogeton epihydrus	Ribbon-leaf pondweed	1.4	2.6	0.5	6.4	5.3	4.3
	Lemna trisulca	Forked duckweed	10.0	3.1	0.5	2.4	2.3	0.0
	Potamogeton natans	Floating-leaf pondweed	5.2	1.5	2.0	3.2	0.8	0.9
	Spirodela polyrhiza	Greater duckweed	0.9	2.0	0.5	1.6	0.8	2.6
	Elodea nuttallii	Slender waterweed	0.0	0.0	0.0	4.8	0.0	2.6
	Vallisneria americana	Wild celery	0.5	2.0	0.0	0.0	0.0	1.7
	Potamogeton berchtoldii	Slender pondweed	0.0	0.0	0.0	0.8	0.0	3.4
	Potamogeton obtusifolius	Blunt-leaved pondweed	1.9	0.0	0.0	0.8	0.8	0.9
	Potamogeton foliosus	Leafy pondweed	0.0	0.0	0.0	1.6	3.1	0.9
	Sagittaria sp. (rosette)	Arrowhead sp. (rosette)	0.0	0.5	0.0	0.0	0.0	2.6
	Freshwater sponge	Freshwater sponge	0.0	0.0	2.6	0.0	1.5	0.0
	Sparganium sp.	Bur-reed sp.	2.8	0.0	0.0	0.0	0.0	0.0
	Schoenoplectus tabernaemontani	Softstem bulrush	0.0	0.0	0.0	3.2	0.0	0.9
	Lemna turionifera	Turion duckweed	0.0	0.0	1.0	0.0	0.0	1.7
	Lemna minor	Lesser duckweed	1.4	0.0	0.0	0.0	1.5	0.0
	Potamogeton spirillus	Spiral-fruited pondweed	1.4	0.5	0.0	0.0	0.0	0.0
	Typha spp.	Cattail spp.	0.0	0.5	0.0	0.8	0.0	0.0
	Stuckenia pectinata	Sago pondweed	0.0	0.0	0.0	0.0	0.0	0.9
	Riccia fluitans	Slender riccia	0.0	0.0	0.0	0.0	0.0	0.9
	Potamogeton vaseyi	Vasey's pondweed	0.0	0.0	0.0	0.0	0.0	0.9
	Eleocharis palustris	Creeping spikerush	0.0	1.0	0.0	0.0	0.0	0.0
	Carex sp. 1	Sedge sp. 1	0.0	0.0	0.0	0.8	0.8	0.0
	Fissidens spp. & Fontinalis spp.	Aquatic Moss	0.0	0.0	0.0	0.0	1.5	0.0
	Sparganium emersum var. acaule	Short-stemmed bur-reed	0.0	0.0	0.5	0.0	0.0	0.0
	Schoenoplectus acutus	Hardstem bulrush	0.0	0.0	0.5	0.0	0.0	0.0
	Sagittaria rigida	Stiff arrowhead	0.0	0.5	0.0	0.0	0.0	0.0
	Eleocharis acicularis	Needle spikerush	0.0	0.5	0.0	0.0	0.0	0.0
	Carex comosa	Bristly sedge	0.5	0.0	0.0	0.0	0.0	0.0

E

APPENDIX E

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

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

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

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

Extracted Table of Contents

S.3.3. Herbicide Treatment

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

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

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

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

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

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

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

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

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

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

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

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

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

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

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

Diquat

Registration and Formulations

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

Mode of Action and Degradation

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

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

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

The half-life of diquat in water generally ranges from a few hours to two days depending on water quality and other environmental conditions. Diquat has been detected in the water column from less than a day up towards 38 DAT, and remains in the water column longer when treating waterbodies with sandy sediments with lower organic matter and clay content (Coats et al. 1964; Grzenda et al. 1966; Yeo 1967; Sewell et al. 1970; Langeland and Warner 1986; Langeland et al. 1994; Poovey and Getsinger 2002; Parsons et al. 2007; Gorzerino et al. 2009; Robb et al. 2014). One study reported that diquat is chemically stable within a pH range of 3 to 8 (Florêncio et al. 2004). Due to the tendency of diquat to be rapidly adsorbed to suspended clays and particulates, long exposure periods are oftentimes not possible to achieve in the field. Studies conducted by Wersal et al. (2010a) did not observe differences in target species efficacy between daytime versus night-time applications of diquat. While large-scale diquat treatments are typically not implemented, a study by Parsons et al. (2007), observed declines in both dissolved oxygen and water clarity following the herbicide treatment.
Diquat binds indefinitely to organic matter, allowing it to accumulate and persist in the sediments over time (Frank and Comes 1967; Simsiman and Chesters 1976). It has been reported to have a very long-lived half-life (1000 days) in sediment because of extremely tight soil sorption, as well as an extremely low rate of degradation after association with sediment (Wauchope et al. 1992; Peterson et al. 1994). Both photolysis and microbial degradation are thought to play minor roles in degradation (Smith and Grove 1969; Emmett 2002). Diquat is not known to leach into groundwater due to its very high affinity to bind to soils.

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

There are no restrictions on swimming or eating fish from waterbodies treated with diquat. Depending on the concentration applied, there is a 1-3 day waiting period after treatment for drinking water. However, in one study, diquat persisted in the water at levels above the EPA drinking water standard for at least 3 DAT, suggesting that the current 3-day drinking water restriction may not be sufficient under all application scenarios (Parsons et al. 2007). Water treated with diquat should not be used for pet or livestock drinking water for one day following treatment. The irrigation restriction for food crops is five days, and for ornamental plants or lawn/turf, it varies from one to three days depending on the concentration used. A study by Mudge et al. (2007) on the effects of diquat on five popular ornamental plant species (begonia, dianthus, impatiens, petunia, and snapdragon) found minimal risks associated with irrigating these species with water treated with diquat up to the maximum use rate of 0.37 ppm.

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

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

Diquat has been shown to control a variety of invasive submerged and floating aquatic plants, including Eurasian watermilfoil (*Myriophyllum spicatum*), curly-leaf pondweed (*Potamogeton crispus*), parrot feather (*Myriophyllum aquaticum*), Brazilian waterweed (*Egeria densa*), water hyacinth, water lettuce (*Pistia stratiotes*), flowering rush (*Butomus umbellatus*), and giant salvinia (*Salvinia molesta*; Netherland et al. 2000; Nelson et al. 2001; Poovey et al. 2002; Langeland et al. 2002; Skogerboe et al. 2006; Martins et al. 2007, 2008; Wersal et al. 2010a; Wersal and Madsen 2010a; Wersal and Madsen 2012; Poovey et al. 2012; Madsen et al. 2016). Studies conducted on the use of diquat for hydrilla (*Hydrilla verticillata*) and fanwort (*Cabomba caroliniana*) control have resulted in mixed reports of efficacy (Van et al. 1987; Langeland et al. 2002; Glomski et al.

2005; Skogerboe et al. 2006; Bultemeier et al. 2009; Turnage et al. 2015). Non-native phragmites (*Phragmites australis* subsp. *australis*) has been shown to not be significantly reduced by diquat (Cheshier et al. 2012).

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

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

Flumioxazin

Registration and Formulations

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

Mode of Action and Degradation

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

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

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

Toxicology

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

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

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

Species Susceptibility

The maximum target concentration of flumioxazin is 0.4 ppm (400 ppb). At least one study has shown that flumioxazin (at or below the maximum label rate) will control the invasive species fanwort (*Cabomba caroliniana*), hydrilla (*Hydrilla verticillata*), Japanese stiltgrass (*Microstegium vimineum*), Eurasian watermilfoil (*Myriophyllum spicatum*), water lettuce (*Pistia stratiotes*), curly-leaf pondweed (*Potamogeton crispus*), and giant salvinia (*Salvinia molesta*), while water hyacinth (*Eichhornia crassipes*) and water pennyworts (*Hydrocotyle* spp.) do not show significant impacts (Bultemeier et al. 2009; Glomski and Netherland 2013a; Glomski and Netherland 2013b; Mudge 2013; Mudge and Netherland 2014; Mudge and Haller 2012; Mudge and Haller 2010). Flowering rush (*Butomus umbellatus*; submersed form) showed mixed success in herbicide trials (Poovey et al. 2012; Poovey et al. 2013). Native species that were significantly impacted (in at least one study) include coontail (*Ceratophyllum demersum*), water stargrass (*Heteranthera* *dubia*), variable-leaf watermilfoil (*Myriophyllum heterophyllum*), America lotus (*Nelumbo lutea*), pond-lilies (*Nuphar* spp.), white waterlily (*Nymphaea odorata*), white water crowfoot (*Ranunculus aquatilis*), and broadleaf cattail (*Typha latifolia*), while common waterweed (*Elodea canadensis*), squarestem spikerush (*Eleocharis quadrangulate*), horsetail (*Equisetum hyemale*), southern naiad (*Najas guadalupensis*), pickerelweed (*Pontederia cordata*), Illinois pondweed (*Potamogeton illinoensis*), long-leaf pondweed (*P. nodosus*), broadleaf arrowhead (*Sagittaria latifolia*), hardstem bulrush (*Schoenoplectus acutus*), common three-square bulrush (*S. pungens*), softstem bulrush (*S. tabernaemontani*), sago pondweed (*Stuckenia pectinata*), and water celery (*Vallisneria americana*) were not impacted relative to controls. Other species are likely to be susceptible, for which the effects of flumioxazin have not yet been evaluated.

S.3.3.2. Submersed, Relatively Slow-Acting Herbicides

2,4-D

Registration and Formulations

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

Mode of Action and Degradation

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

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

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

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

Toxicology

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

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

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

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

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

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

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

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

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

<u>Species Susceptibility</u>

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

Efficacy and selectivity of 2,4-D is a function of concentration and exposure time (CET) relationships, and rates of 0.5-2.0 ppm coupled with exposure times ranging from 12 to 72 hours have been effective at achieving Eurasian watermilfoil control under laboratory settings (Green and Westerdahl 1990). In addition, long exposure times (>14 days) to low-concentrations of 2,4-D (0.1-0.25 ppm) have also been documented to achieve milfoil control (Hall et al. 1982; Glomski and Netherland 2010).

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

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

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

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

Fluridone

Registration and Formulations

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

Mode of Action and Degradation

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

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

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

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

Toxicology

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

Species Susceptibility

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

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

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

Endothall

Registration and Formulations

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

Mode of Action and Degradation

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

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

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

Toxicology

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

Dipotassium salt formulations

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

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

Dimethylalkylamine salt formulations

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

The dimethylalkylamine formulations are more active on aquatic plants than the dipotassium formulations, but also are 2-3 orders of magnitude more toxic to non-target aquatic organisms (EPA RED Endothall 2005; Keckemet 1969). The 2005 reregistration decision document limits aquatic use of the dimethylalkylamine formulations to algae, Indian swampweed (*Hygrophila polysperma*), water celery (*Vallisneria americana*), hydrilla (*Hydrilla verticillata*), fanwort (*Cabomba caroliniana*), bur reed (*Sparganium* sp.), common waterweed (*Elodea canadensis*), and Brazilian waterweed (*Egeria densa*). Coontail (*Ceratophyllum demersum*), watermilfoils (*Myriophyllum* spp.), naiads (*Najas* spp.), pondweeds (*Potamogeton* spp.), water stargrass

(*Heteranthera dubia*), and horned pondweed (*Zannichellia palustris*) were to be removed from product labels (EPA RED Endothall 2005).

Species Susceptibility

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

Dipotassium salt formulations

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

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

Species which were not significantly impacted or which recovered quickly include watershield (*Brasenia schreberi*; Skogerboe and Getsinger 2001), muskgrasses (*Chara* spp.; Yeo 1970; Wells and Clayton 1993; Hofstra and Clayton 2001), common waterweed (Yeo 1970; Wells and Clayton

1993; Skogerboe and Getsinger 2002), water stargrass (Skogerboe and Getsinger 2001), water net (*Hydrodictyon reticulatum*; Wells and Clayton 1993), the freshwater macroalgae *Nitella clavata* (Yeo 1970), yellow pond-lily (*Nuphar advena*; Skogerboe and Getsinger 2002), swamp smartweed (*Polygonum hydropiperoides*; Skogerboe and Getsinger 2002), pickerelweed (*Pontederia cordata*; Skogerboe and Getsinger 2001), softstem bulrush (*Schoenoplectus tabernaemontani*; Skogerboe and Getsinger 2001), and broadleaf cattail (*Typha latifolia*; Skogerboe and Getsinger 2002).

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

Dimethylalkylamine salt formulations

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

Imazamox

Registration and Formulations

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

Mode of Action and Degradation

Imazamox is a systemic herbicide that moves throughout the plant tissue and prevents plants from producing a necessary enzyme, acetolactate synthase (ALS), which is not found in animals. Susceptible plants will stop growing soon after treatment, but plant death and decomposition will occur over several weeks (Mudge and Netherland 2014). If used as a post-emergence herbicide, imazamox should be applied to plants that are actively growing. Resistance to ALS-inhibiting herbicides has appeared in weeds at a higher rate than other herbicide types in terrestrial environments (Tranel and Wright 2002).
Dissipation studies in lakes indicate a half-life ranging from 4 to 49 days with an average of 17 days. Herbicide breakdown does not occur readily in deep, poorly-oxygenated water where there

is no light. In this part of a lake, imazamox will tend to bind to sediments rather than breaking down, with a half-life of approximately 2 years. Once in soil, leaching to groundwater is believed to be very limited. The breakdown products of imazamox are nicotinic acid and di- and tricarboxylic acids. It has been suggested that photolytic break down of imazamox is faster than other herbicides, reducing exposure times. However, short-term imazamox exposures have also been associated with extended regrowth times relative to other herbicides (Netherland 2011).

Toxicology

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

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

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

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

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

Species Susceptibility

In Wisconsin, imazamox is used for treating non-native emergent vegetation such as non-native phragmites (*Phragmites australis* subsp. *australis*) and flowering rush (*Butomus umbellatus*). Imazamox may also be used to treat the invasive curly-leaf pondweed (*Potamogeton crispus*). Desirable native species that may be affected could include other pondweed species (long-leaf pondweed (*P. nodosus*), flat-stem pondweed (*P. zosteriformis*), leafy pondweed (*P. foliosus*), Illinois pondweed (*P. illinoensis*), small pondweed (*P. pusillus*), variable-leaf pondweed (*P. gramineus*), water-thread pondweed (*P. diversifolius*), perfoliate pondweed (*P. perfoliatus*), large-leaf pondweed (*P. amplifolius*), watershield (*Brasenia schreberi*), and some bladderworts (*Utricularia* spp.). Higher rates of imazamox will control Eurasian watermilfoil (*Myriophyllum* *spicatum*) but would also have greater non-target impacts on native plants. Imazamox can also be used during a drawdown to prevent plant regrowth and on emergent vegetation.

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

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

Florpyrauxifen-benzyl

Registration and Formulations

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

Mode of Action and Degradation

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

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

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

Toxicology

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

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

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

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

<u>Species Susceptibility</u>

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

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

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

S.3.3.3. Emergent and Wetland Herbicides

Glyphosate

Registration and Formulations

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

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

Mode of Action and Degradation

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

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

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

Toxicology

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

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

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

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

Species Susceptibility

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

Imazapyr

Registration and Formulations

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

Mode of Action and Degradation

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

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

Toxicology

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

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

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

Species Susceptibility

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

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

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

S.3.3.4. Herbicides Used for Submersed and Emergent Plants

Triclopyr

Registration and Formulations

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

Mode of Action and Degradation

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

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

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

Toxicology

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

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

Species susceptibility

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

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

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

Penoxsulam

Registration and Formulations

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

Mode of Action and Degradation

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

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

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

Toxicology

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

Species Susceptibility

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

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

S.3.4. Physical Removal Techniques

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

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

Manual and Mechanical Cutting

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

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

The timing of cutting operations, as for other management approaches, is important. For species dependent on vegetative propagules, control methods should be taken before the propagules are formed. However, for species with rhizomes, cutting too early in the season merely postpones growth while later-season cutting can better reduce plant abundance (Dall Armellina et al. 1996a, 1996b). Eurasian watermilfoil regrowth may be slower if cutting is conducted later in the summer (June or later). Cutting in the fall, rather than spring or summer, may result in the lowest amount of Eurasian watermilfoil regrowth the year after management (Kimbel and Carpenter 1981). However, managing early in the growing season may reduce non-target impacts to native plant populations when early-growing non-native plants are the dominant targets (Nichols and Shaw 1986). Depending on regrowth rate and management goals, multiple harvests per growing season may be necessary (Rawls 1975).

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

Hand Pulling and Diver-Assisted Suction Harvesting

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

Ecological Impacts of Physical Removal Techniques

Plants accrue nutrients into their tissues, and thus plant removal may also remove nutrients from waterbodies (Boyd 1970), though this nutrient removal may not be significant among lake types. Cutting and harvesting of aquatic plants can lead to declines in fish as well as beneficial zooplankton, macroinvertebrate, and native plant and mussel populations (Garner et al. 1996;

Aldridge 2000; Torn et al. 2010; Barton et al. 2013). Many studies suggest leaving some vegetated areas undisturbed to reduce negative effects of cutting on fish and other aquatic organisms (Swales 1982; Garner et al. 1996; Unmuth et al. 1998; Aldridge 2000; Greer et al. 2012). Recovery of these populations to cutting in the long-term is understudied and poorly understood (Barton et al. 2013). Effects on water quality can be minimal but nutrient cycling may be affected in wetland systems (Dall Armellina et al. 1996a; Martin et al. 2003). Cutting can also increase algal production, and turbidity temporarily if sediments are disturbed (Wile 1978; Bailey and Calhoun 2008).

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

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

F

APPENDIX F

WDNR Fisheries Materials

Summary of Fishery Surveys
Phillips Chain of Lakes, Price County, 2007 – 2008 and 2013 – 2014

WDNR’s Fisheries Management Team from Park Falls completed fyke netting and electrofishing surveys in 2007 – 2008 and 2013 – 2014 to assess the status of important fish populations in Elk, Duroy, Long, and Wilson lakes, collectively known as the Phillips Chain of Lakes. Fyke netting in October yielded useful information on black crappies. Fyke nets set shortly after the spring thaw targeted walleye, muskellunge, northern pike, and yellow perch and provided complementary information on black crappies and bluegills. Electrofishing surveys in late spring documented the abundance and size structure of smallmouth bass, largemouth bass, and bluegill populations.

Survey Effort

2007 – 2008

	Fall Fyke Nets	Early Spring Fyke Nets	Late Spring Electrofishing			
	Oct 8-11, 2007	Apr 27-May 1, 2008	June 2-10, 2008			
Water Temp	57-62° F	40-47° F	62-69° F			
			Gamefish		Panfish	
	Net-nights	Net-nights	Miles	Hours	Miles	Hours
Elk	3	9	2.66	1.03	1.01	0.43
Duroy	5	15	4.12	1.50	2.60	0.90
Long	5	10	4.04	1.60	2.54	1.00
Wilson	5	5	4.15	1.80	1.05	0.40
Combined	18	39	14.97	5.93	7.20	2.73

2013 – 2014

	Fall Fyke Nets	Early Spring Fyke Nets	Late Spring Electrofishing			
	Sept 30-Oct 4, 2014	Apr 27-May 3, 2014	May 29-June 3, 2014			
Water Temp	59-63° F	39-44° F	69-78° F			
			Gamefish		Panfish	
	Net-nights	Net-nights	Miles	Hours	Miles	Hours
Elk	6	12	3.00	1.22	1.00	0.43
Duroy	10	10	2.90	1.20	1.40	0.50
Long	10	20	3.00	1.18	1.50	0.58
Wilson	10	10	3.10	1.38	1.05	0.52
Combined	36	52	12.00	4.98	4.95	2.03

Each survey occurred over similar ranges of water temperature in both periods, and in most cases net locations and electrofishing routes were duplicated. With the noted exceptions, we are confident that our samples were well-timed to accurately represent their respective targets and to compare population status between years. Quality, preferred, and memorable sizes referenced in this summary are based on standard proportions of world record lengths developed for each species by the American Fisheries Society. "Keeper size" is based on known angler behavior.

Habitat Characteristics

The Phillips Chain of Lakes is a 1,236-acre impoundment on the Elk River, ranking second to the Pike Lake Chain of Lakes in total surface area among Price County waters. About 40% of the Chain lies within the City of Phillips, Wisconsin. Before dam construction, the waters presently known as Duroy, Elk, and Long lakes were natural lakes on the mainstream. In their unimpounded condition, Duroy and Elk lakes had expansive surface areas and moderate depths (8 to 15 feet), whereas Long Lake had a narrow, elongated shape and a maximum depth of about 44 feet. Wilson Lake did not exist before these waters were dammed. That shallow arm of the Chain, sometimes called Wilson Creek Flowage, was formed over flooded wetlands adjacent to the Wilson Creek tributary.

Water clarity is relatively low in the Phillips Chain, indicating a fertile lake system with moderately high nutrient levels that occasionally produce mid-summer algae blooms. Average summer Secchi disk visibility ranged from 3.1 feet in Wilson Lake to 4.2 feet in Long Lake.

Public boat access to the Chain is sufficient to accommodate the demand without crowding. Improved boat landings with concrete ramps, boarding piers, and parking for vehicles and trailers provide no-fee access to Elk Lake from County Highway H and to Wilson Lake from County Highway W. Additional boat access with fewer improvements is available from several town roads and private sites on all four lakes. Most recreational watercraft, including most pontoon boats, can navigate through the large culvert under Highway W that connects Wilson Creek Flowage and Long Lake.

Despite subtle differences among lakes, we manage the Chain's fishery as a unit with exceptions as necessary under the stakeholder-supported goals and objectives outlined in the ***Fishery Management Plan—Phillips Lake Chain, February 2008.***

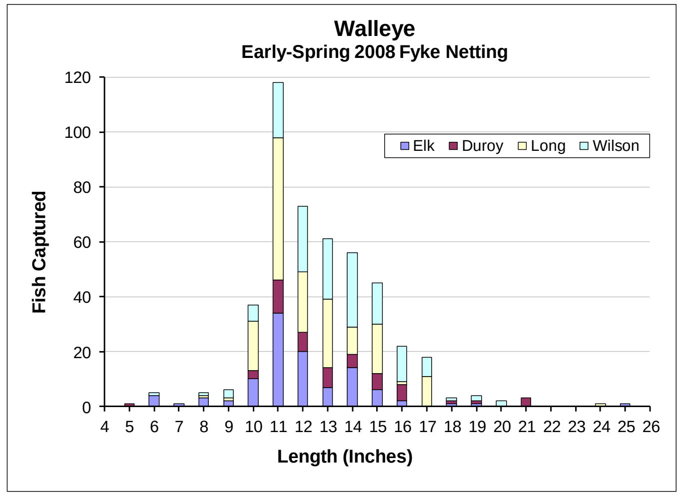
Summary of Results

The fish community's diversity can be attributed to the variety of habitat in the Phillips Chain and its tributaries. We captured 20 fish species in our netting and electrofishing surveys, including several riverine species. The principle predators in the Chain were walleyes, northern pike, and muskellunge. Their important prey included white suckers, northern hog suckers, shorthead redhorse, silver redhorse, golden redhorse, and yellow perch (whose cylindrical shape predators prefer) as well as young bluegills and black crappies. Catch rates of bluegill and black crappies were generally higher in 2014 than in 2008, whereas gamefish and yellow perch catch rates were higher in the 2008 survey than in 2014.

Walleye

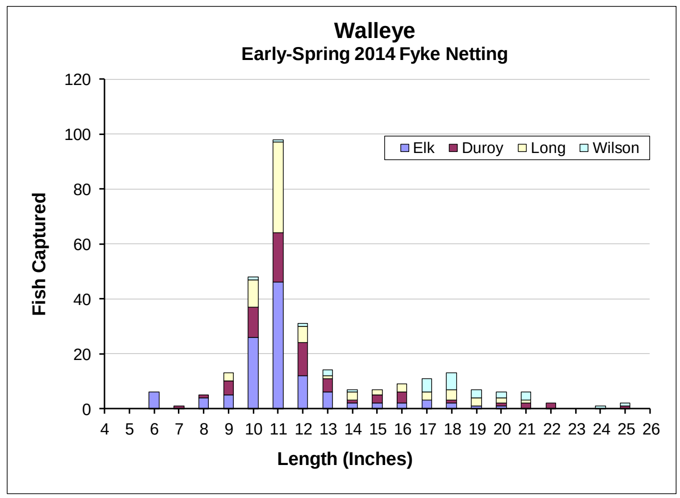
Early-Spring Fyke Netting
2008

	Number per net-night ≥ 10"	Quality Size ≥ 15"	Preferred Size ≥ 20"	Memorable Size ≥ 25"
Elk	11	11%	1%	1%
Duroy	3.5	33%	6%	0%
Long	16	20%	0.6%	0%
Wilson	28	29%	1%	0%
Combined	12	22%	2%	0.2%



Early-Spring Fyke Netting
2014

	Number per net-night ≥ 10"	Quality Size ≥ 15"	Preferred Size ≥ 20"	Memorable Size ≥ 25"
Elk	8.6	11%	1.0%	0%
Duroy	6.1	23%	10%	2%
Long	3.6	25%	4%	0%
Wilson	2.8	78%	26%	4%
Combined	5.1	24%	6%	0.8%



Across the entire Chain early spring fyke nets captured walleye at a rate nearly 2½ times higher in 2008 than in 2014. Long and Wilson lakes experienced the greatest declines, while catch rate increased in Duroy Lake and remained relatively unchanged in Elk Lake. We did not estimate walleye density in 2008 or 2014, so we do not know whether the population meets our objective for 3 – 5 adults per acre in the Chain; (fyke net capture rates in early spring are not statistically associated with adult walleye density).

Our indices of walleye size structure were generally within the objective range (20 – 40% at least 15 inches long), though walleye in Elk Lake fell short of the goal in both years. A proposed fishing regulation would protect and improve the size distribution of the walleye population in the Phillips Chain. Focusing angler harvest toward abundant, slow-growing walleyes of intermediate size 10 – 13 inches long while allowing conservative harvest of one walleye > 14 inches should maintain or increase the proportion of adult walleyes longer than 15 inches. If approved, the new rule would take effect in April 2016: "Walleye of any length may be kept, but only one can be over 14 inches."

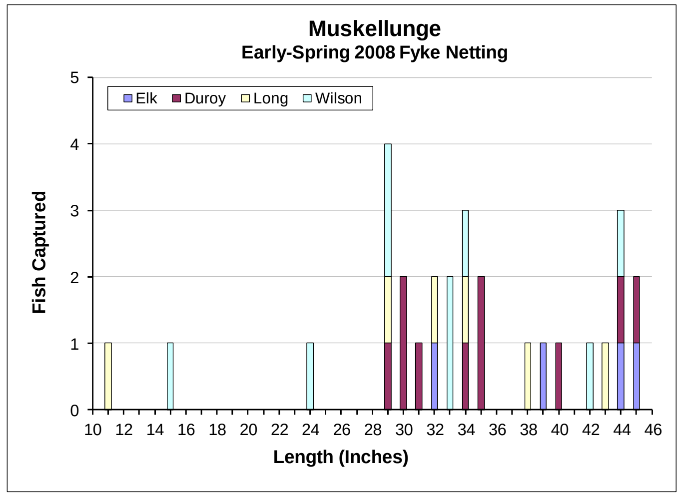
Comparing walleyes in early spring 2008 and 2014 fyke nets, the steep decline in capture rate, coupled with a sharp increase in the proportion of walleye 15 inches and longer, raise concerns about low recruitment to the sub-population in Wilson Lake since the last 2-inch fingerlings were stocked into the

Phillips Chain in 2002 and 2004. Habitat and fish community characteristics in shallow, weedy Wilson Lake appear to be less favorable for walleye compared to the rest of the Chain, and the narrow, shallow culvert connecting these waters may limit fish movements and interactions. If fall 2019 electrofishing and early spring 2020 fyke netting surveys suggest that walleye reproductive success is still low in Wilson Lake, then we should consider stocking walleye fingerlings there again to advance our goals for the Chain's walleye and panfish populations.

Muskellunge

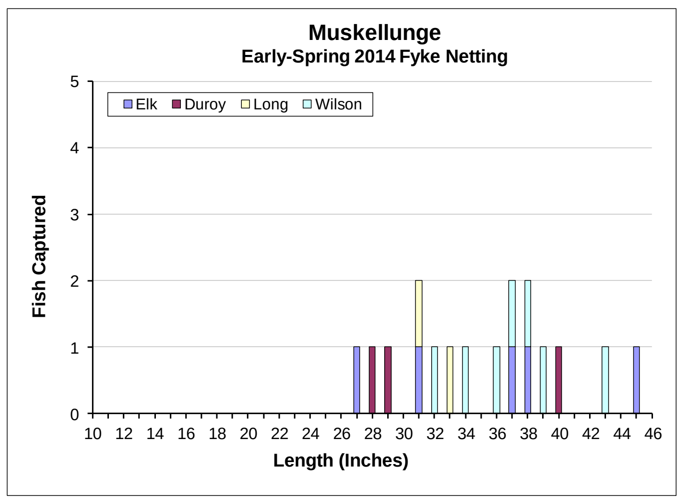
Early-Spring Fyke Netting
2008

	Number per net-night ≥ 20"	Quality Size ≥ 30"	Preferred Size ≥ 38"	Memorable Size ≥ 42"
Elk	0.4	100%	75%	50%
Duroy	0.7	90%	30%	20%
Long	0.6	80%	40%	20%
Wilson	1.6	63%	25%	25%
Combined	0.7	81%	37%	26%



Early-Spring Fyke Netting
2014

	Number per net-night ≥ 20"	Quality Size ≥ 30"	Preferred Size ≥ 38"	Memorable Size ≥ 42"
Elk	0.4	80%	40%	20%
Duroy	0.3	33%	33%	0%
Long	0.1	100%	0%	0%
Wilson	0.7	100%	43%	14%
Combined	0.3	82%	35%	12%



Except in Elk Lake, catch rates of muskellunge in early spring fyke nets were lower in 2014 than in 2008, though we do not believe the apparent decline points to a decrease in musky abundance. Because walleyes were our primary target in both years, fyke nets set in 2014 at average water temperature 40 – 43°F may not have represented the spawning muskellunge population's status as well as those set in 2008 when the average water temperature we recorded throughout the Chain (40 – 47°F) was closer to the optimal temperature (55°F) at which muskies spawn. In the next surveys scheduled in 2020, we recommend shifting the primary purpose to a deliberate assessment of muskellunge size distribution and abundance by fyke netting at warmer water temperatures (49 – 60°F) later in spring (SN2 protocol) to intercept mature muskies during their spawning activities.

In spring 2008 the combined fyke net capture rate across the entire Chain ranked in the 43rd percentile statewide among Class A2 muskellunge populations that offer the best "action" fishing opportunities.

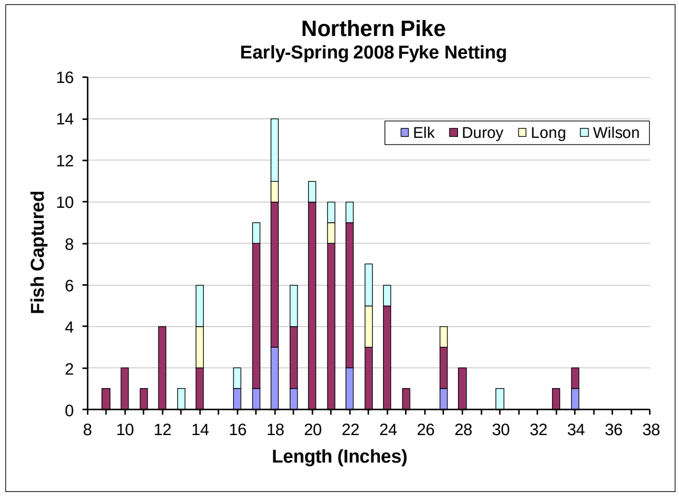
Though samples were small, the 2008 survey revealed that muskellunge had attained our objective that 15 – 30% should be 42 inches or longer. With even smaller samples in 2014 we cannot determine whether size structure has changed since 2008.

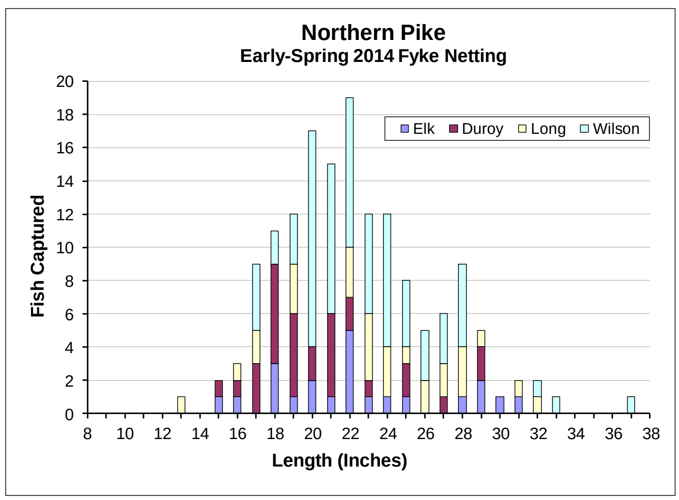
Electronic records dating to 1972 show that muskellunge were planted into the Phillips Chain almost annually for two decades at rates of 1 – 4 fingerlings per acre. Musky stocking resumed in 2003 with fewer and larger fish planted less frequently. Most recently, new recruits are added to the muskellunge population by a combination of natural reproduction and stocking large fingerlings 10 – 12 inches long at a rate of 0.5 per acre in alternate years.

Northern Pike

Early-Spring Fyke Netting
2008

	Number per net-night ≥ 14"	Quality Size ≥ 21”	Preferred Size ≥ 28"	Memorable Size ≥ 34"
Elk	1.3	40%	10%	10%
Duroy	4.0	51%	7%	2%
Long	0.8	57%	0%	0%
Wilson	3.2	38%	6%	0%
Combined	2.5	48%	7%	2%



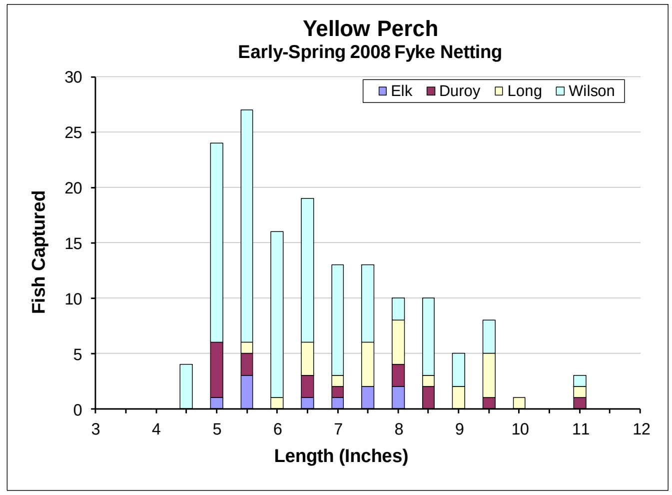


Early-Spring Fyke Netting
2014

	Number per net-night ≥ 14"	Quality Size ≥ 21”	Preferred Size ≥ 28"	Memorable Size ≥ 34"
Elk	2.0	64%	23%	0%
Duroy	3.4	42%	6%	0%
Long	1.6	78%	22%	0%
Wilson	7.5	69%	11%	1%
Combined	3.2	64%	14%	0.7%

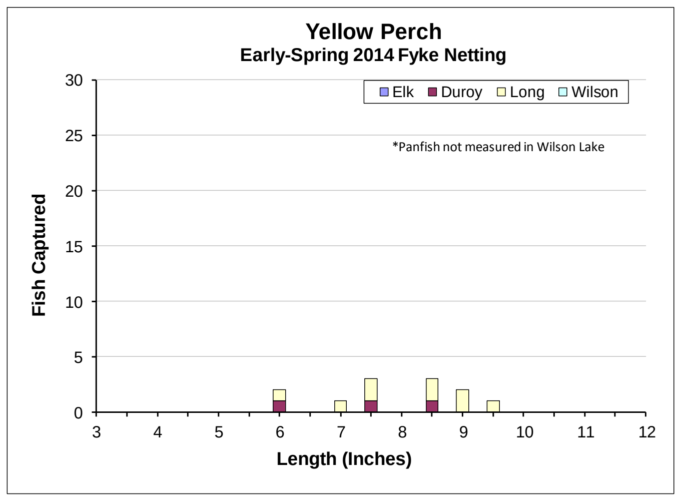
Overall, early spring fyke nets captured northern pike at a slightly higher rate in 2014 than in 2008, possibly indicating a small increase in pike abundance, or perhaps because netting in 2014 occurred at water temperatures (40 – 43°F) closer to the optimal range of pike spawning activity (34 – 40°F) than in 2008 (40 – 47°F). Catch rates in Long and Wilson lakes doubled from 5 years earlier. Any increase in pike abundance did not diminish their size structure. Rather, the proportion of pike ≥ 28 inches in the Chain also doubled. No goals were written for northern pike in the 2008 Fishery Management Plan as pike were of low or even negative interest to most local stakeholders. Nonetheless, at moderate density with better-than-average size structure, northern pike are easily catchable much of the time, providing additional angling opportunity in the Phillips Chain under statewide fishing regulations.

Yellow Perch



Early-Spring Fyke Netting
2008

	Number per net-night ≥ 5"	Quality Size ≥ 8"	Preferred Size ≥ 10"
Elk	1.1	20%	0%
Duroy	1.1	38%	6%
Long	2.3	57%	9%
Wilson	20	16%	1%
Combined	3.8	25%	3%



Early-Spring Fyke Netting
2014

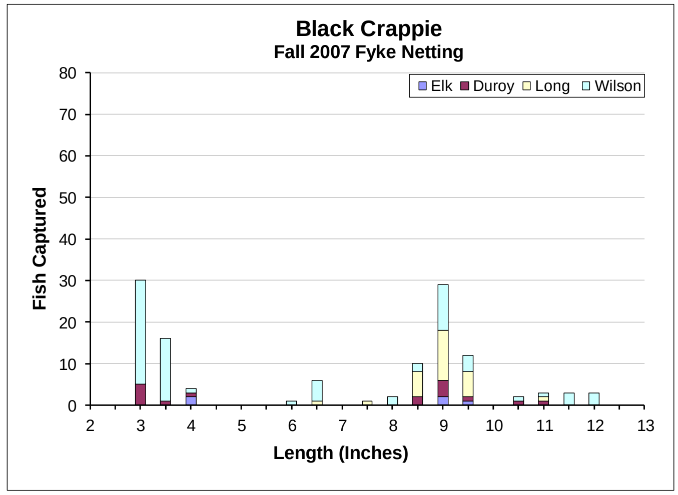
	Number per net-night ≥ 5"	Quality Size ≥ 8"	Preferred Size ≥ 10"
Elk	0.8	56%	0%
Duroy	0.3	33%	0%
Long	1.0	56%	0%
Wilson	-	-	-
Combined	0.6	38%	0%

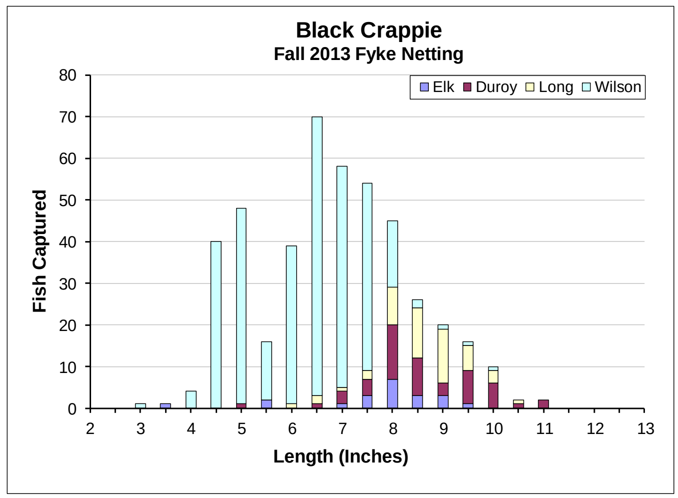
When yellow perch were present, their capture rate in early spring fyke nets was highly variable (SD = 45; mean = 18) in 49 surveys that our Team completed in 2008 – 2014. Consequently, we are not sure how to interpret these results. Nonetheless, similar fyke net capture rates indicated low yellow perch abundance in Duroy, Elk, and Long lakes in early spring 2008 and 2014. This apparent scarcity of perch may be attributed to predation by walleye, muskellunge, and northern pike, all of which prefer to eat tube-shaped perch over dish-shaped sunfish. In Wilson Lake panfish were too numerous to measure or count in spring 2014 fyke nets, but we subjectively rated perch as “moderately abundant” and similar to the level we found there in 2008 when fyke nets in Wilson Lake captured perch at the highest rate in the Chain. Compared to local lakes, both surveys showed the Phillips Chain had unusually high proportions of perch longer than 8 inches. No specific management goals were set for yellow perch in the 2008 management plan. However if approved, an experimental fishing regulation intended to increase the average size of black crappies and bluegills by allowing anglers to harvest 5 sunfish, 5 crappies, and 5 perch in a daily bag limit of 15 panfish, may also help to improve the size structure of the Phillips Chain’s yellow perch population.

Black Crappie

Fall Fyke Netting
2007

	Number per net-night ≥ 5"	Quality Size ≥ 8"	Preferred Size ≥ 10"	Memorable Size ≥ 12"
Elk	1.0	100%	0%	0%
Duroy	1.8	100%	22%	0%
Long	5.4	93%	4%	0%
Wilson	6.6	82%	24%	9%
Combined	4.0	89%	15%	4%





Fall Fyke Netting
2013

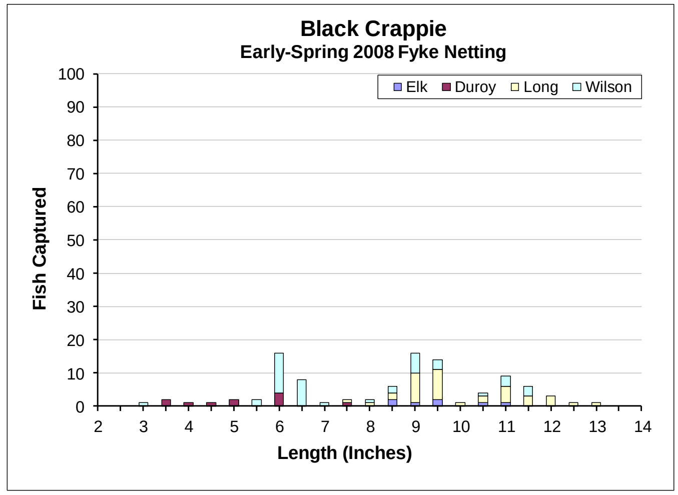
	Number per net-night ≥ 5"	Quality Size ≥ 8"	Preferred Size ≥ 10"	Memorable Size ≥ 12"
Elk	3.3	70%	0%	0%
Duroy	5.1	82%	18%	0%
Long	5.0	88%	8%	0%
Wilson	29	7%	0.4%	0%
Combined	11	30%	3%	0%

Capture rates of black crappies in fall fyke nets were about 3 – 4 times higher in 2013 than in 2007, except in Long Lake where fall catch rates were similar in both years. However, spring 2014 fyke nets revealed a higher abundance of 2- and 3-year-old crappies (presumed ages) in Long Lake that were not well represented in the fall 2013 sample. Crappies did not attain the desired range of moderate abundance (10 – 20 crappies per net-night ≥ 5 inches in fall fyke nets), except in Wilson Lake where crappies surpassed that goal in fall 2013. In both fall surveys crappies also fell short of our size objective (30 – 40% of crappies ≥ 5 inches should be 10 inches or longer). Based on ages estimated from scales taken in 2007, we can cautiously forecast better fishing opportunity for crappies throughout the Chain, especially in Long and Wilson lakes, as the strong year classes produced in 2012 and 2013 grow to the sizes that anglers prefer. From casual observations we believe crappies in the Phillips Chain receive moderate, but consistent fishing pressure. In our opinion, it is unlikely that we will achieve our crappie population objectives under current regulations that permit anglers to harvest 25 panfish daily with no minimum length limit. A pending proposal to limit angler harvest to 5 crappies, 5 sunfish, and 5 yellow perch in a daily bag limit of 15 panfish combined, should serve to improve crappie size structure, moderate the extremes of fluctuating crappie abundance, and distribute the harvest more equitably among anglers who frequent the Phillips Chain. If approved, the new fishing regulation would take effect in April 2016.

Black Crappie

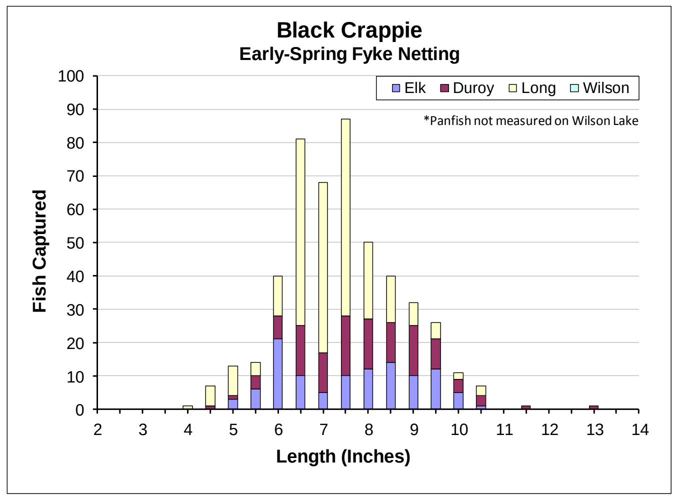
Early-Spring Fyke Netting
2008

	Number per net-night ≥ 5"	Quality Size ≥ 8"	Preferred Size ≥ 10"	Memorable Size ≥ 12"
Elk	0.8	100%	29%	0%
Duroy	0.5	0%	0%	0%
Long	3.8	97%	42%	13%
Wilson	8.4	45%	17%	0%
Combined	2.4	67%	27%	5%



Early-Spring Fyke Netting
2014

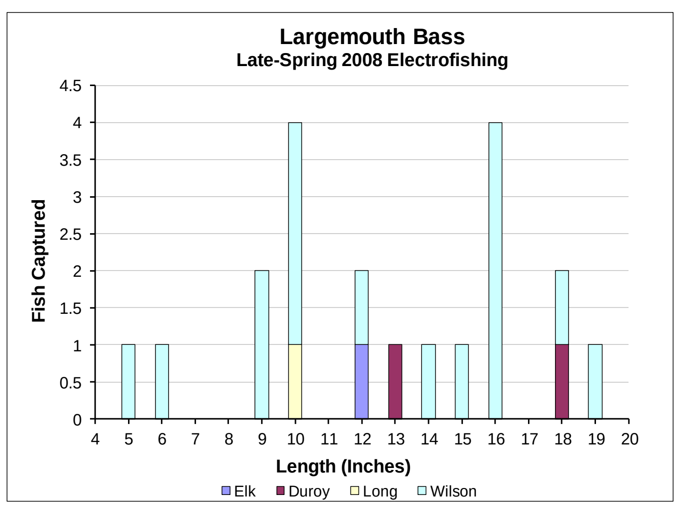
	Number per net-night ≥ 5"	Quality Size ≥ 8"	Preferred Size ≥ 10"	Memorable Size ≥ 12"
Elk	9.1	50%	6%	0%
Duroy	12	51%	8%	0.9%
Long	28	22%	2%	0%
Wilson	-	-	-	-
Combined	15	36%	4%	0.2%



Largemouth Bass

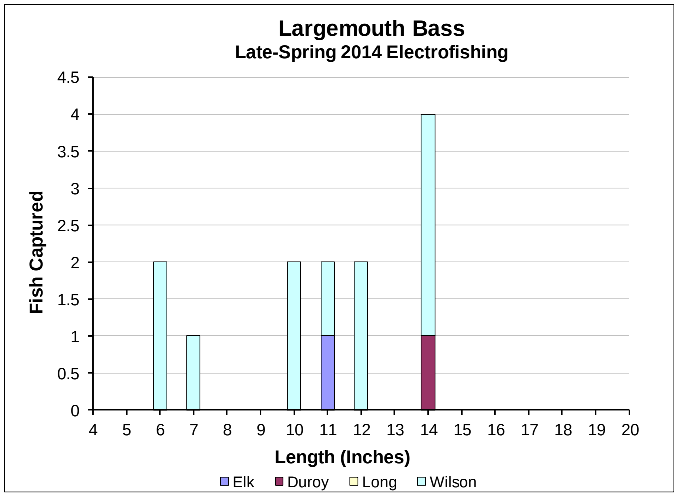
Late-Spring Electrofishing
2008

	Number per mile ≥ 8"	Number per hour ≥ 8"	Quality Size ≥ 12"	Preferred Size ≥ 15"
Elk	0.4	1.0	100%	0%
Duroy	0.5	1.3	100%	50%
Long	0.2	0.6	0%	0%
Wilson	3.4	7.8	64%	50%
Combined	1.2	3.0	67%	44%



Late-Spring Electrofishing
2014

	Number per mile ≥ 8"	Number per hour ≥ 8"	Quality Size ≥ 12"	Preferred Size ≥ 15"
Elk	0.3	0.8	0%	0%
Duroy	0.3	0.8	100%	0%
Long	0	0	0%	0%
Wilson	2.6	5.8	63%	0%
Combined	0.8	2.0	60%	0%

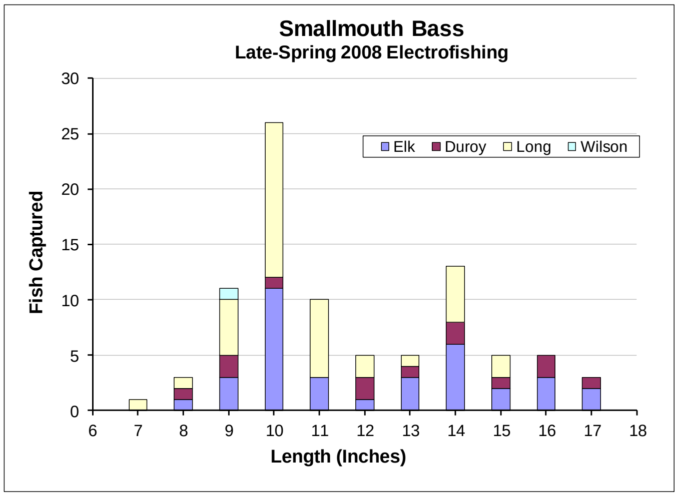


Not surprisingly, electrofishing capture rates in late spring 2008 and 2014indicated largemouth bass at very low abundance throughout the Chain. Habitat and fish community characteristics in this system do not favor largemouth bass reproduction and survival. Predictably, in both years our electrofishing catch rate of largemouth bass was highest in shallow, weedy Wilson Lake, where habitat seems better suited for largemouth bass. At such low abundance largemouth bass should attain preferred size and add diversity to the fishery without compromising our ability to attain objectives for more important species.

Smallmouth Bass

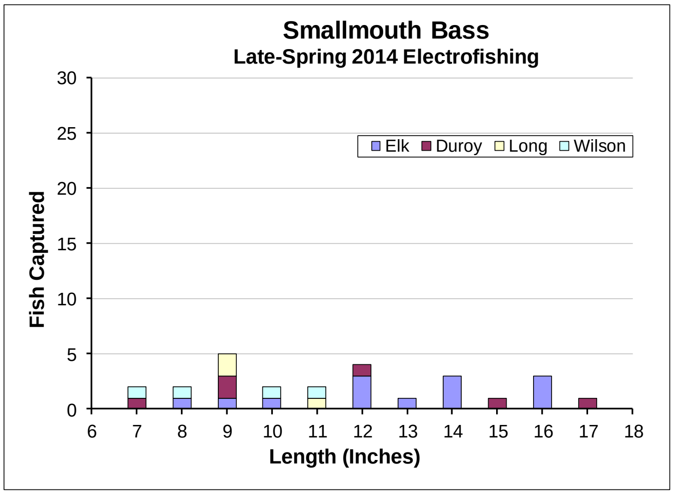
Late-Spring Electrofishing
2008

	Number per mile ≥ 7"	Number per hour ≥ 7"	Quality Size ≥ 11"	Preferred Size ≥ 14"	Memorable Size ≥ 17"
Elk	13	34	57%	37%	6%
Duroy	3.2	8.7	69%	46%	8%
Long	9.4	24	45%	18%	0%
Wilson	0.2	0.6	0%	0%	0%
Combined	5.8	15	53%	30%	3%



Late-Spring Electrofishing
2014

	Number per mile ≥ 7"	Number per hour ≥ 7"	Quality Size ≥ 11"	Preferred Size ≥ 14"	Memorable Size ≥ 17"
Elk	4.3	11	77%	46%	0%
Duroy	2.1	5.0	50%	33%	17%
Long	1.0	2.5	33%	0%	0%
Wilson	1.3	2.9	25%	0%	0%
Combined	2.2	5.2	58%	31%	4%

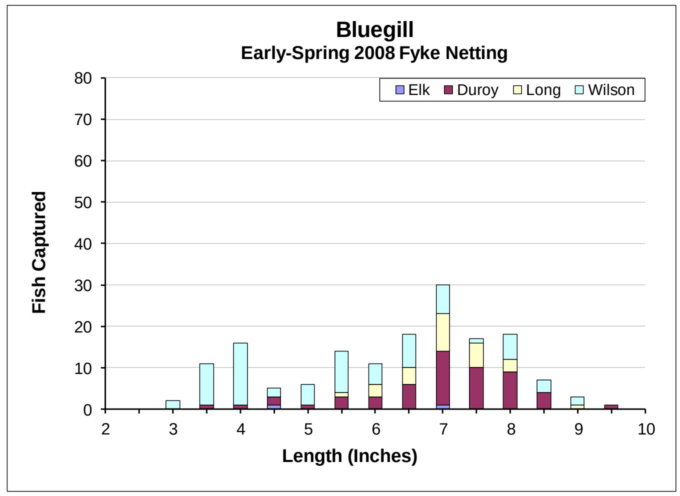


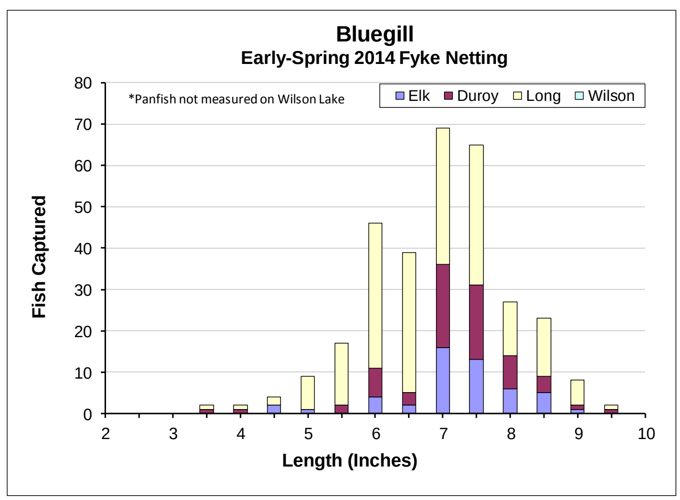
Late spring electrofishing surveys in 2008 and 2014 both revealed that the Phillips Chain's smallmouth bass population fell far short of the goals set in the *2008 Fishery Management Plan,* suggesting perhaps that our objectives for abundance (25 – 50 smallmouth bass ≥ 7 inches per electrofishing hour) and size structure (50 – 70% at least 14 inches long) may be too ambitious in comparison to the area's highest quality smallmouth bass fisheries. Most recently Elk Lake had the highest smallmouth bass abundance and best size structure of the four lakes. We do not know why overall our electrofishing capture rate decreased nearly two-thirds from 2008 to 2014, though it's possible that crayfish, the favorite food of smallmouth bass, had experienced a similar decline. Because of the strong catch-and-release ethic among bass anglers, we suspect that few bass are taken under statewide harvest regulations. A late spring 2014 survey in nearby Solberg Lake in the same watershed revealed that smallmouth bass there also did not attain objectives for size and number. If our spring 2020 surveys show similar results in these waters, we should revise the Fishery Management Plans to reflect more realistic objectives for their smallmouth bass populations.

Bluegill

Early-Spring Fyke Netting
2008

	Number per net-night ≥ 3"	Quality Size ≥ 6"	Keeper Size ≥ 7"	Preferred Size ≥ 8"
Elk	0.2	50%	50%	0%
Duroy	3.6	85%	69%	26%
Long	2.7	96%	70%	15%
Wilson	15	42%	25%	14%
Combined	4.1	66%	48%	18%





Early-Spring Fyke Netting
2014

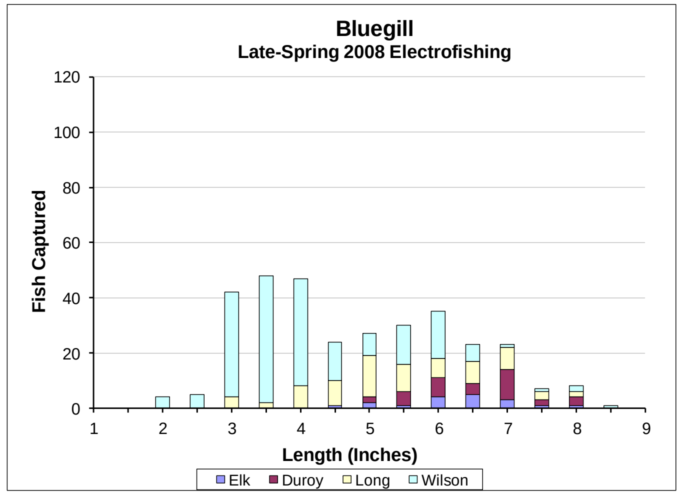
	Number per net-night ≥ 3"	Quality Size ≥ 6"	Keeper Size ≥ 7"	Preferred Size ≥ 8"
Elk	4.2	94%	82%	24%
Duroy	6.6	94%	79%	21%
Long	20	86%	51%	17%
Wilson	-	-	-	-
Combined	9.8	89%	62%	19%

In Elk, Duroy, and Long lakes electrofishing capture rates of bluegills in late spring 2008 and 2014 were near the objective range (50 – 100 bluegill ≥ 3 inches per hour) selected to represent the desired moderate population abundance. The same measures indicated very high bluegill abundance in Wilson Lake. In both periods bluegills attained or nearly attained the desired size structure (5 – 10% at least 8 inches) in the three lakes with moderate abundance. Wilson Lake had the highest bluegill abundance and the lowest proportions keeper- and preferred-size bluegills longer that 7 and 8 inches. The noted differences in bluegill abundance and size between Wilson Lake and the rest of the Chain are probably related to differences in habitat and the effectiveness of predators to control panfish density. Early spring fyke nets captured higher percentages of large bluegills than late spring electrofishing did. A regulation intended to increase the average length of bluegills in the Phillips Chain by allowing anglers to harvest 5 sunfish, 5 black crappies, and 5 yellow perch in a daily bag limit of 15 panfish will take effect in April 2016, if approved.

Bluegill

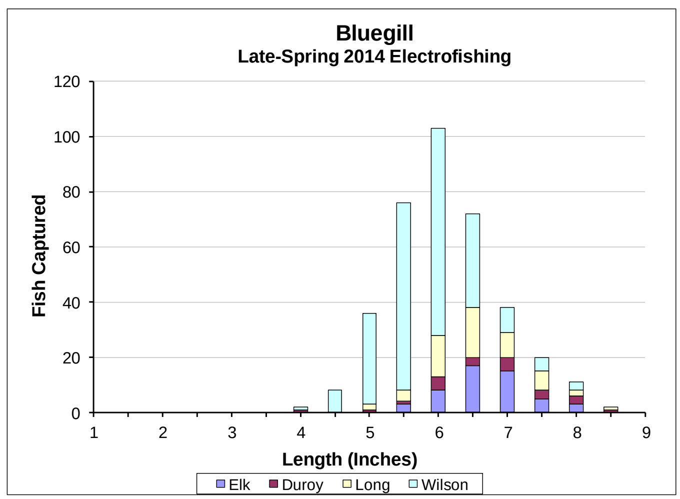
Late-Spring Electrofishing
2008

	Number per mile ≥ 3"	Number per hour ≥ 3"	Quality Size ≥ 6"	Keeper Size ≥ 7"	Preferred Size ≥ 8"
Elk	18	42	78%	28%	6%
Duroy	13	38	79%	47%	9%
Long	30	76	37%	17%	3%
Wilson	178	468	15%	3%	2%
Combined	44	115	31%	12%	3%



Late-Spring Electrofishing
2014

	Number per mile ≥ 3"	Number per hour ≥ 3"	Quality Size ≥ 6"	Keeper Size ≥ 7"	Preferred Size ≥ 8"
Elk	51	119	94%	45%	6%
Duroy	16	46	87%	52%	17%
Long	39	100	90%	33%	5%
Wilson	225	454	53%	7%	1%
Combined	74	181	67%	19%	4%



Survey data collected and analyzed by: Bill Loeffler, Kendal Patrie, Greg Rublee, Jeff Scheirer, Jeanette Wendler, and Jess Zakovec—WDNR Fishery Team, Park Falls.

Written by: Chad Leanna—Fishery Technician and Jeff Scheirer—Fishery Biologist, December 8, 2014.

Reviewed and approved for web posting by: Mike Vogelsang—acting Hayward Field Unit Supervisor, December 16, 2014.

Lake	Year	Species	Strain (Stock)	Age Class	# Fish Stocked	Avg Fish Length (in)
Duroy Lake	1972	Muskellunge	Unspecified	Fingerling	350	13
Duroy Lake	1973	Muskellunge	Unspecified	Fingerling	175	13
Duroy Lake	1974	Muskellunge	Unspecified	Fingerling	350	11
Duroy Lake	1975	Muskellunge	Unspecified	Fingerling	67	9
Duroy Lake	1976	Muskellunge	Unspecified	Fingerling	820	6.33
Duroy Lake	1977	Muskellunge	Unspecified	Fingerling	670	6
Duroy Lake	1978	Muskellunge	Unspecified	Fingerling	344	10
Duroy Lake	1979	Muskellunge	Unspecified	Fingerling	670	7
Duroy Lake	1980	Muskellunge	Unspecified	Fingerling	670	11
Duroy Lake	1981	Muskellunge	Unspecified	Fingerling	335	9
Duroy Lake	1982	Muskellunge	Unspecified	Fingerling	300	11
Duroy Lake	1983	Muskellunge	Unspecified	Fingerling	670	9
Duroy Lake	1984	Muskellunge	Unspecified	Fingerling	500	11
Duroy Lake	1985	Muskellunge	Unspecified	Fingerling	760	11
Duroy Lake	1986	Muskellunge	Unspecified	Fingerling	760	11
Duroy Lake	1987	Muskellunge	Unspecified	Fingerling	1,140	11
Duroy Lake	1988	Muskellunge	Unspecified	Fingerling	760	9
Duroy Lake	1989	Muskellunge	Unspecified	Fingerling	380	12
Duroy Lake	1990	Muskellunge	Unspecified	Fingerling	380	13
Duroy Lake	1991	Muskellunge	Unspecified	Fingerling	760	11
Duroy Lake	1992	Muskellunge	Unspecified	Fingerling	758	10
Duroy Lake	1993	Muskellunge	Unspecified	Fingerling	758	9
Duroy Lake	1999	Muskellunge	Unspecified	Large Fingerling	190	12.5
Duroy Lake	2003	Muskellunge	Unspecified	Large Fingerling	379	10.9
Duroy Lake	2005	Muskellunge	Unspecified	Large Fingerling	379	10.6
Duroy Lake	2007	Muskellunge	Upper Chippewa River	Large Fingerling	252	12.3
Duroy Lake	2009	Muskellunge	Upper Chippewa River	Large Fingerling	379	10
Duroy Lake	2011	Muskellunge	Upper Chippewa River	Large Fingerling	379	9.9
Duroy Lake	2013	Muskellunge	Upper Chippewa River	Large Fingerling	190	11.2
Duroy Lake	2014	Muskellunge	Upper Chippewa River	Large Fingerling	190	11.3
Duroy Lake	2015	Muskellunge	Upper Chippewa River	Large Fingerling	379	12.25
Duroy Lake	2017	Muskellunge	Upper Chippewa River	Large Fingerling	37	11.5
Duroy Lake	2019	Muskellunge	Upper Chippewa River	Large Fingerling	88	12.6
Elk Lake	1993	Muskellunge	Unspecified	Fingerling	176	12
Elk Lake	1972	Muskellunge	Unspecified	Fingerling	150	13
Elk Lake	1973	Muskellunge	Unspecified	Fingerling	75	13
Elk Lake	1975	Muskellunge	Unspecified	Fingerling	25	9
Elk Lake	1976	Muskellunge	Unspecified	Fingerling	142	5
Elk Lake	1977	Muskellunge	Unspecified	Fingerling	142	9
Elk Lake	1978	Muskellunge	Unspecified	Fingerling	70	11
Elk Lake	1979	Muskellunge	Unspecified	Fingerling	140	7
Elk Lake	1980	Muskellunge	Unspecified	Fingerling	140	11
Elk Lake	1981	Muskellunge	Unspecified	Fingerling	70	9
Elk Lake	1982	Muskellunge	Unspecified	Fingerling	70	13
Elk Lake	1983	Muskellunge	Unspecified	Fingerling	140	11
Elk Lake	1984	Muskellunge	Unspecified	Fingerling	180	8
Elk Lake	1985	Muskellunge	Unspecified	Fingerling	180	11

Lake	Year	Species	Strain (Stock)	Age Class	# Fish Stocked	Avg Fish Length (in)
Elk Lake	1986	Muskellunge	Unspecified	Fingerling	180	11
Elk Lake	1987	Muskellunge	Unspecified	Fingerling	420	11
Elk Lake	1988	Muskellunge	Unspecified	Fingerling	180	9
Elk Lake	1989	Muskellunge	Unspecified	Fingerling	90	12
Elk Lake	1991	Muskellunge	Unspecified	Fingerling	180	11
Elk Lake	1992	Muskellunge	Unspecified	Fingerling	176	10
Elk Lake	1999	Muskellunge	Unspecified	Large Fingerling	44	11.8
Elk Lake	2003	Muskellunge	Unspecified	Large Fingerling	88	10.9
Elk Lake	2005	Muskellunge	Unspecified	Large Fingerling	88	10.6
Elk Lake	2007	Muskellunge	Upper Chippewa River	Large Fingerling	59	12.3
Elk Lake	2009	Muskellunge	Upper Chippewa River	Large Fingerling	88	10
Elk Lake	2011	Muskellunge	Upper Chippewa River	Large Fingerling	88	9.9
Elk Lake	2013	Muskellunge	Upper Chippewa River	Large Fingerling	44	11.2
Elk Lake	2014	Muskellunge	Upper Chippewa River	Large Fingerling	44	11.3
Elk Lake	2015	Muskellunge	Upper Chippewa River	Large Fingerling	88	12.25
Elk Lake	2017	Muskellunge	Upper Chippewa River	Large Fingerling	11	11.5
Elk Lake	2019	Muskellunge	Upper Chippewa River	Large Fingerling	22	12.6
Long Lake	1993	Muskellunge	Unspecified	Fingerling	1672	12
Long Lake	1972	Muskellunge	Unspecified	Fingerling	430	13
Long Lake	1973	Muskellunge	Unspecified	Fingerling	460	9
Long Lake	1974	Muskellunge	Unspecified	Fingerling	850	11
Long Lake	1975	Muskellunge	Unspecified	Fingerling	84	9
Long Lake	1976	Muskellunge	Unspecified	Fingerling	1036	6.33
Long Lake	1977	Muskellunge	Unspecified	Fingerling	836	7
Long Lake	1979	Muskellunge	Unspecified	Fingerling	836	7
Long Lake	1980	Muskellunge	Unspecified	Fingerling	836	11
Long Lake	1981	Muskellunge	Unspecified	Fingerling	250	9
Long Lake	1982	Muskellunge	Unspecified	Fingerling	400	13
Long Lake	1983	Muskellunge	Unspecified	Fingerling	485	9
Long Lake	1984	Muskellunge	Unspecified	Fingerling	620	11.5
Long Lake	1985	Muskellunge	Unspecified	Fingerling	836	11
Long Lake	1986	Muskellunge	Unspecified	Fingerling	836	11
Long Lake	1987	Muskellunge	Unspecified	Fingerling	1254	11
Long Lake	1988	Muskellunge	Unspecified	Fingerling	836	9
Long Lake	1989	Muskellunge	Unspecified	Fingerling	418	12
Long Lake	1990	Muskellunge	Unspecified	Fingerling	418	11
Long Lake	1991	Muskellunge	Unspecified	Fingerling	836	11
Long Lake	1992	Muskellunge	Unspecified	Fingerling	836	10
Long Lake	1999	Muskellunge	Unspecified	Large Fingerling	212	12.5
Long Lake	2003	Muskellunge	Unspecified	Large Fingerling	418	10.9
Long Lake	2005	Muskellunge	Unspecified	Large Fingerling	418	10.6
Long Lake	2007	Muskellunge	Upper Chippewa River	Large Fingerling	278	12.3
Long Lake	2009	Muskellunge	Upper Chippewa River	Large Fingerling	418	10
Long Lake	2011	Muskellunge	Upper Chippewa River	Large Fingerling	418	9.8
Long Lake	2013	Muskellunge	Upper Chippewa River	Large Fingerling	209	11.2
Long Lake	2014	Muskellunge	Upper Chippewa River	Large Fingerling	209	11.3

Lake	Year	Species	Strain (Stock)	Age Class	# Fish Stocked	Avg Fish Length (in)
Long Lake	2015	Muskellunge	Upper Chippewa River	Large Fingerling	418	12.25
Long Lake	2017	Muskellunge	Upper Chippewa River	Large Fingerling	44	11.5
Long Lake	2019	Muskellunge	Upper Chippewa River	Large Fingerling	105	12.6
Wilson Lake	1972	Muskellunge	Unspecified	Fingerling	350	15
Wilson Lake	1973	Muskellunge	Unspecified	Fingerling	350	11
Wilson Lake	1974	Muskellunge	Unspecified	Fingerling	865	11
Wilson Lake	1975	Muskellunge	Unspecified	Fingerling	70	9
Wilson Lake	1976	Muskellunge	Unspecified	Fingerling	902	6.33
Wilson Lake	1977	Muskellunge	Unspecified	Fingerling	702	6
Wilson Lake	1978	Muskellunge	Unspecified	Fingerling	350	11
Wilson Lake	1979	Muskellunge	Unspecified	Fingerling	900	9
Wilson Lake	1980	Muskellunge	Unspecified	Fingerling	700	11
Wilson Lake	1981	Muskellunge	Unspecified	Fingerling	250	9
Wilson Lake	1982	Muskellunge	Unspecified	Fingerling	400	12
Wilson Lake	1983	Muskellunge	Unspecified	Fingerling	700	9
Wilson Lake	1984	Muskellunge	Unspecified	Fingerling	600	11
Wilson Lake	1985	Muskellunge	Unspecified	Fingerling	700	11
Wilson Lake	1986	Muskellunge	Unspecified	Fingerling	700	11
Wilson Lake	1988	Muskellunge	Unspecified	Fingerling	700	9
Wilson Lake	1989	Muskellunge	Unspecified	Fingerling	350	12
Wilson Lake	1990	Muskellunge	Unspecified	Fingerling	350	11
Wilson Lake	1991	Muskellunge	Unspecified	Fingerling	700	11
Wilson Lake	1992	Muskellunge	Unspecified	Fingerling	700	10
Wilson Lake	1993	Muskellunge	Unspecified	Fingerling	1400	12
Wilson Lake	2003	Muskellunge	Unspecified	Large Fingerling	175	10.9
Wilson Lake	2007	Muskellunge	Upper Chippewa River	Large Fingerling	115	12.3
Wilson Lake	2009	Muskellunge	Upper Chippewa River	Large Fingerling	176	10
Wilson Lake	2011	Muskellunge	Upper Chippewa River	Large Fingerling	175	9.9
Wilson Lake	2013	Muskellunge	Upper Chippewa River	Large Fingerling	88	11.2
Wilson Lake	2015	Muskellunge	Upper Chippewa River	Large Fingerling	175	12.4
Wilson Lake	2017	Muskellunge	Upper Chippewa River	Large Fingerling	37	11.5
Wilson Lake	2019	Muskellunge	Upper Chippewa River	Large Fingerling	141	12.6

Lake	Year	Species	Strain (Stock)	Age Class	# Fish Stocked	Avg Fish Length (in)
Wilson Lake	1987	Northern Pike X Muskellunge	Unspecified	Fingerling	700	11

Lake	Year	Species	Strain (Stock)	Age Class	# Fish Stocked	Avg Fish Length (in)
Duroy Lake	1994	Walleye	Unspecified	Fingerling	9,545	3
Duroy Lake	1995	Walleye	Unspecified	Fingerling	9,800	2.6
Duroy Lake	1996	Walleye	Unspecified	Fingerling	18,950	1.5
Duroy Lake	1997	Walleye	Unspecified	Large Fingerling	10,000	2.7
Duroy Lake	2000	Walleye	Unspecified	Small Fingerling	22,950	2.25
Duroy Lake	2002	Walleye	Mississippi Headwaters	Small Fingerling	18,950	1.7
Duroy Lake	2004	Walleye	Mississippi Headwaters	Small Fingerling	19,125	1.2
Elk Lake	1996	Walleye	Unspecified	Fingerling	4400	1.5

Lake	Year	Species	Strain (Stock)	Age Class	# Fish Stocked	Avg Fish Length (in)
Elk Lake	1994	Walleye	Unspecified	Fingerling	2275	3
Elk Lake	1995	Walleye	Unspecified	Fingerling	2400	2.6
Elk Lake	1997	Walleye	Unspecified	Large Fingerling	2000	2.7
Elk Lake	2002	Walleye	Mississippi Headwaters	Small Fingerling	4390	1.3
Elk Lake	2004	Walleye	Mississippi Headwaters	Small Fingerling	4400	1.2
Long Lake	1994	Walleye	Unspecified	Fingerling	10525	3
Long Lake	1995	Walleye	Unspecified	Fingerling	10000	2.6
Long Lake	1996	Walleye	Unspecified	Fingerling	20900	1.5
Long Lake	1997	Walleye	Unspecified	Large Fingerling	10000	2.7
Long Lake	2000	Walleye	Unspecified	Small Fingerling	20900	1.7
Long Lake	2002	Walleye	Mississippi Headwaters	Small Fingerling	20900	1.7
Long Lake	2004	Walleye	Mississippi Headwaters	Small Fingerling	20898	1.2
Wilson Lake	1994	Walleye	Unspecified	Fingerling	8840	3
Wilson Lake	1995	Walleye	Unspecified	Fingerling	9000	2.6
Wilson Lake	1996	Walleye	Unspecified	Fingerling	17550	1.5
Wilson Lake	1997	Walleye	Unspecified	Large Fingerling	9000	2.7
Wilson Lake	2000	Walleye	Unspecified	Small Fingerling	17299	2.7
Wilson Lake	2002	Walleye	Mississippi Headwaters	Small Fingerling	17500	1.4