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MODELING AND ESTIMATION OF HARVEST PARAMETERS AND ANNUAL SURVIVAL RATES OF WOOD DUCKS IN MINNESOTA
PHASE I: A COMPARISON OF THE EFFECTIVENESS OF THREE TYPES OF TRAPS

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INTRODUCTION

There has been concern about the recent liberalization of harvest regulations on the wood duck (*Aix sponsa*) population in Minnesota. Consequently, I conducted preliminary analyses of band and recovery data to: (1) examine the influence of harvest regulations on harvest parameters and annual survival rates of wood ducks in Minnesota, and (2) estimate the sample size of banded individuals needed to generate precise estimates of these parameters (Berdeen 2012). Results suggested that a greater sample size of banded individuals from most cohorts was needed, and that the greatest spatial need was in the Laurentian Mixed Forest ecological province (Hanson and Hargrave 1996).

Because the target sample size of banded individuals from most cohorts is unlikely to be obtained regularly with current banding efforts, it is necessary to identify and develop more efficacious capture methods to generate more precise estimates and gain a better understanding of the effects of regulatory change. Therefore, I conducted a pilot field study to identify which of 3 trap types was most effective with regard to capturing wood ducks and dabbling ducks in general. The specific objectives to: (1) compare the number of wood ducks captured per trap-day among trap types, (2) compare the number of trapping mortalities of waterfowl per trap-day among trap types, (3) provide project costs, and (4) provide ancillary information that may be used to improve the capture methodology for wood ducks. If this study is successful, funding may be sought to capture and band wood ducks at additional sites in Minnesota. The knowledge gained in this pilot study should benefit future banding efforts and increase the likelihood that sample size targets for each cohort of interest will be attained.

STUDY AREA

This pilot project was conducted in the Laurentian Mixed Forest ecological province near Bemidji. This location was chosen because there is a need to increase the sample size of banded wood ducks in this ecological province (Berdeen 2012). Further, it was logistically and economically more feasible to administer this project out of the Wetlands Wildlife Research and Populations Group office in Bemidji (i.e., purchasing trap materials, supervising seasonal employees on 2 projects, minimizing travel and housing costs for employees).

METHODS

The 3 trap types examined were Benning II traps (Dieter et al. 2009), oval trap with a lead panel (Dieter et al. 2009), and floating y-traps (Rowell 1984). Sites that appeared to be consistent with habitats attractive to wood ducks were identified, and bait (corn) was placed at 12 locations to concentrate ducks near potential trap sites. Wood duck use was sufficient to justify the deployment of traps at 9 prebaited sites. Traps were deployed 4–5 days per week and checked daily when operational.

It would have been advantageous from the perspective of study design and analysis to deploy each type of trap at every trapping site, but this was not logistically possible. Specifically, wetlands had receded in spatial extent because of the long-term drought, thus making it difficult to feasibly situate multiple traps in some wetlands. It also would have been difficult to deploy and check those types of traps (i.e., Benning II, oval with leads) that rested on the substrate of wetlands with especially soft bottoms. It was most practical to deploy floating y-traps at such locations. Further, it was difficult to transport materials used in heavier or bulkier trap types (i.e., Benning II, floating y) to sites at which relatively thick vegetation and soft bottom substrate negatively influenced access.

RESULTS

Capture efforts occurred for 68 trap-days at 9 sites (Table 1), with no capture mortalities of waterfowl occurring. Substantial raccoon activity was detected near traps at 3 sites, and black bear activity destroyed a trap at another site. Field personnel captured and banded a small number of ducks during this pilot study, with the species composition (0.75) skewed toward the target species (Table 1).

I did not conduct formal data analyses because the small sample size of banded ducks precluded findings of significantly different numbers of duck captures per trap-day among trap types or sites. However, summary statistics and field observations provide insights regarding the methods that were successful under field situations and study approaches that can be improved upon.

The number of wood ducks captured per trap-day was greatest for floating y-traps, followed by Benning II traps, and finally oval traps with leads (Table 2). There also appeared to be site-specific variation of the total number of ducks captured per trap-day. Specifically, there no ducks were captured at 5 of 9 sites, but between 0.056 and 1.000 duck per trap-day at 4 other sites (Table 3).

Project costs were as follows: $5119.36 for the wages of one seasonal employee (310 hours [271 regular, 8 holiday, 31 overtime]), $1573.89 for one loaner vehicle (3263 miles and 2 months rental), $135.87 for bait (corn), and $2363.61 for trap materials. The total cost of the pilot project was $9192.73. However, this cost was conservative because fencing material from earlier projects was used in trap construction and seasonal employees from the summer waterfowl banding project assisted with trap construction. Seasonal employees spent approximately 120 hours building 18 traps, with a disproportionate amount of this time allotted to constructing floating y-traps.

DISCUSSION

It is difficult to reliably determine the effectiveness of each trap type because of the relatively small sample size of captured and banded wood ducks. However, the limited information available suggests that the floating y-trap was the most effective of those examined with regard to numbers of wood ducks captured per trap-day (Table 2). This trap type seems especially well-suited for use at wetland sites with bottom substrate that is too soft to traverse, and therefore must be checked via boat. It should be emphasized that y-traps required more time to construct than the others we examined and was difficult to transport to wetlands with poor access because of its bulkiness. Further, floating y-traps required some assembly after being transported to the trapping site.

The difference in the number of captured ducks per trap-day among trapping sites (Table 3) suggests that there were site-specific differences in the number of ducks using these sites. Further, field personnel observed that duck use of each wetland varied temporally, and that many more ducks were observed swimming and feeding near some traps than were caught. Thus, it is necessary to identify the potential sources of such spatiotemporal variation and the variables that may have limited capture success. Disturbance from potential predators and field personnel, temporal changes of food availability (e.g., the ripening of rice during late summer) on the landscape, and the long-term drought may have negatively influenced the number of ducks using capture sites over time. Also, it may be necessary to slightly alter traps to capture a greater number of ducks. Field personnel closely followed schematic plans during trap construction and assembly in the field, but the number of ducks that were near the trap but did not enter suggest that capture efforts may have been more effective if trap doors were widened and bait was distributed differently.

RECOMMENDATIONS

Information regarding the spatiotemporal distribution of wood ducks in Minnesota during August and September currently is not available, but could benefit future capture efforts. This would be especially important for capture efforts in the Laurentian Mixed Forest ecological province, where wood ducks did not appear to have been concentrated during the study period yet a greater sample size of banded individuals is needed.

Once wetlands that are attractive to wood ducks have been identified, it may be important to bait such sites during the early phase of fieldwork to encourage site-fidelity in ducks so that they will be available for capture and banding throughout the field season. It also may be important to identify those sites that are frequented by predators relatively early in the field season, and deal with this situation in an appropriate manner. I will consider modifying trap plans to: (1) use a smaller size of wire mesh to reduce the likelihood that captured ducks escape, and (2) decrease the overall dimensions of the relatively bulky types of traps to facilitate transportation in the field.

During the 2012 field season, primarily one person was assigned to the wood duck banding project and 4 other seasonal personnel assisted as time permitted. To remain within budgetary constraints, the person assigned primarily to wood duck banding could work only 4–5 days per week. Thus, the time that traps were operational ultimately was limited by this constraint. The availability of 2 full-time field personnel would have permitted traps to be operational for 7 days per week, facilitated the moving of traps when necessary, and increased likelihood of increasing the number of wood ducks captured.

ACKNOWLEDGEMENTS

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Table 1. Numbers of 3 species of ducks captured in 3 types of traps placed at 9 study sites in northcentral Minnesota, 2012.

			Number captured		
Site	Trap type	Number of trap-days	Bluewinged teal	Mallard	Wood duck
106	Benning II	7	2		1
	Oval	5			1
Buena Vista wetland	Benning II	3			
	Oval	3			
Burns Lake	Benning II	5			
	Oval	8		1	
	Y	5			
Cub impoundment	Benning II	3			1
	Oval	2			
	Y	2			
Ekstrom wetlands	Benning II	2			
	Y	4			
Hart Lake	N/A[a]				
Hubbard Lake	N/A[a]				
Lake Andrusia	N/A[a]				
MRS	Oval	4			
Ose Lake	Benning II	4			
Tax Forfeit Lake	Y	6			6
Tower Lake	Y	5			

[a] Sites were baited but traps were not deployed at these sites.

Table 2. Species-specific captures per trap-day using 3 types of traps in northcentral Minnesota, 2012.

	Species-specific captures per trap-day		
Trap type	Blue-winged teal	Mallard	Wood duck
Benning II	0.083	0	0.083
Oval with leads	0	0.045	0.045
Floating y	0	0	0.273

Table 3. Total number of ducks captured per trap-day at 9 study sites in northcentral Minnesota, 2012.

Site	Number of ducks captured	Number of trap-days	Number of ducks captured per trap-day
106	4	12	0.333
Buena Vista wetland	0	6	0
Burns Lake	1	18	0.056
Cub Impoundment	1	7	0.143
Ekstrom wetlands	0	6	0
MRS	0	4	0
Ose Lake	0	4	0
Tax Forfeit Lake	6	6	1.000
Tower Lake	0	5	0

EFFICACY OF CO_2 AS A FISH PISCICIDE: FINAL REPORT FOR THE WINTER 2013 PILOT CO_2 STUDY

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SUMMARY OF FINDINGS

Lake managers are in need of additional tools to control undesirable fish populations in lakes. Injection of CO_2 under the ice during winter has great potential due to its lack of environmental persistence and effectiveness as a fish toxicant. However, application at a whole-lake level has never been attempted, and there are several unknowns, including whether it's even logistically possible to elevate CO_2 levels high enough to be lethal to fish, how long CO_2 levels would remain elevated, and how well CO_2 will mix in lake water under the ice. In winter 2013 collaborators from the University of Minnesota, University of St. Thomas, and the Minnesota Department of Natural Resources conducted a pilot study to assess the efficacy and logistics of using CO_2 as a piscicide. We added 545 kg of dry ice to a 0.40 ha pond with a mean depth of 0.5 m on 13 February 2013. We used 2 sondes and 1 CO_2 meter deployed under the ice to make multiple daily estimates of water temperature, pH, and CO_2 levels for 5 days before the treatment and for 26 days afterwards. Our goal was to double CO_2 levels in the lake to approximately 150 mg/L, a level thought to be toxic to fish. Results showed that the addition of the dry ice raised CO_2 levels in the lake from the initial concentration of 19 mg/L to 204 mg/L within three hours, and reached a maximum level of 274 mg/L within one week of application. Similar results were obtained with all sensors, indicating the CO_2 mixed well under the ice. Additionally, CO_2 levels remained above 200 mg/L for 26 days, indicating the technique works well in terms of exposing fish to high CO_2 levels for long periods of time. Assessing toxicity on resident fish and invertebrate populations was not part of this project, but a visual survey of the pond by DNR staff after ice out found dead fish and many dead snails, indicating the CO_2 did create a lethal environment. Overall this study found that adding CO_2 to lakes was logistically possible, the CO_2 mixed well under the ice and went into solution very well, and stayed at elevated levels for a long time. Thus, we recommend the DNR or others take the next steps in developing CO_2 as a fish piscicide. Key questions that need to be addressed in subsequent research include identifying specific CO_2 levels toxic to target fish, assessing how lake chemistry (alkalinity etc.) will impact the amount of CO_2 needed to reach toxic levels, and whole-lake assessment on effects of elevated CO_2 on target fish populations and non-target animals such as aquatic invertebrates.

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BACKGROUND

Extensive drainage of wetlands and shallow lakes in Minnesota (hereafter shallow lakes) has placed a priority on managing the remaining basins in ways that optimize ecosystem services. Shallow lakes provide numerous ecosystem services, but providing habitat to waterfowl and other species is of particular importance to natural resource managers. Thus, shallow lake management in Minnesota and elsewhere has often focused on maximizing the suitability of shallow lakes as habitat for shallow lake dependent species.

One primary method for improving habitat quality of shallow lakes is to manage them to be in a clear-water state. Shallow lakes exhibit two alternative stable states; a clear-water state dominated by submerged macrophytes versus a turbid-water state dominated by phytoplankton (Scheffer et al. 1993). The proportion of lakes in a turbid state appears to vary spatially in Minnesota. Zimmer et al. (2009) studied 72 shallow lakes dispersed across Grant and Polk counties in Minnesota, and found that 51% of the sites in Grant were turbid, compared to just 11% in Polk. Current research at a nearly state-wide scale indicates relatively high proportion of lakes in turbid states in the Windom, Twin Cities, and Alexandria areas, with lower proportions observed in Itasca State Park and in the Chippewa National Forest (Zimmer et al. unpublished data). Taken together, these data suggest a north-south statewide gradient where the proportion of turbid lakes is highest in south and lowest in the north, and that a large number of lakes are turbid across large portions of Minnesota.

Shallow lakes do shift between states, with a shift from clear to turbid states driven by a loss of submerged macrophytes due to increased water depth, nutrient loading, or abundance of benthivorous fish. Similarly, reduced abundance of planktivorous and benthivorous fish, or introduction of piscivorous fish, can cause a lake to shift from turbid to clear (reviewed by Scheffer 1998). Higher abundance of macrophytes and aquatic invertebrates increases the suitability of clear-water lakes as habitat for waterfowl and amphibians (Scheffer et al. 2006), and waterfowl use of shallow lakes has increased in response to lakes shifting from turbid to clear (Hanson and Butler 1994). Work in Minnesota and elsewhere has shown that turbid states are often associated with planktivorous and benthivorous fish (Søndergaard et al. 1997, Zimmer et al. 2002), and that reducing abundance of these fish via biomanipulation can improve water clarity and induce a shift to the clear state (Hanson and Butler 1994, Potthoff et al. 2008). Thus, sharp reduction of these fish is potentially a powerful management tool for lake managers in Minnesota and elsewhere.

Managing these shallow lakes for waterfowl is of increased importance recently due to the fact that many of the shallow lakes have been drained in the past 100 years and eutrophication has pushed many of the remaining lakes into a turbid (less desirable) state. In fact, DNR recently proposed a collaborative plan to improve habitat for waterfowl and this approach calls for restoration of 1,800 shallow lakes statewide (Duck Plan Highlights). The current methods for biomanipulation include reverse winter aeration, stocking piscivorous fish, water-level drawdown, and application of fish toxicants such as rotenone. Reverse aeration consists of pumping air under the ice to circulate anoxic water from the bottom and thereby cause a sudden drop in dissolved oxygen to kill fish. This method was assessed by Shroyer (2007), and he reported the technique achieved little success killing species such as black bullhead and fathead minnows and was not recommended for general use. Stocking piscivorous fish has been tested in Minnesota and elsewhere, and has produced mixed success. Søndergaard et al. (1997) found that stocking northern pike improved water clarity, but effects were short lived. Similarly, Potthoff et al. (2008) reported stocking walleye in lakes with fathead minnows improved water clarity, but results suggested stocking would need to done for at least three consecutive years to ensure a shift to the clear water state. Friederichs et al. (2010) examined piscivore effects in shallow Minnesota lakes, and concluded phytoplankton would only be reduced by piscivores in lakes limited to soft-rayed planktivore species (e.g. no bullheads). However, recent surveys have indicated lakes limited to soft-rayed planktivore species comprise just 5% of shallow lakes in western Minnesota (K.D. Zimmer unpublished data), suggesting piscivore stocking will have limited application. Water-level drawdown is commonly done in Minnesota by the DNR, but hydrological considerations limit the number of lakes where it can be used. Rotenone is the most common fish toxicant used in Minnesota and elsewhere, and its application is most often used to achieve biomanipulation in the state. Use of rotenone by Minnesota DNR Fisheries and Wildlife varies year to year, but probably averages 3-5 lakes treated per year (Todd Call MN DNR, personal communication). However, success in reducing fish populations with rotenone has been mixed, especially for species such as fathead minnow. Zimmer at al. (2001) observed average population declines of just 66% following rotenone application to 11 lakes with fathead minnows, and only one lake shifted to a stable clear-water state. Moreover, environmental concerns have been raised regarding ingredients in liquid rotenone formulations, making the long-term availability of the chemical unknown. Given the limitations of reversed aeration, piscivore stocking, drawdown, and rotenone application, there is a pressing need for additional tools for lake managers to control planktivorous and benthivorous fish in Minnesota's shallow lakes.

One proposed new technique for biomanipulation is to inject CO_2 under the ice in winter. The lethal effects of CO_2 on fish have been known for quite some time, but research on CO_2 levels influencing fish has historically focused on its application as an anesthetic (e.g. Bernier and Randall 1998). High concentrations of CO_2 in waters causes fish narcosis and mortality through several mechanisms including hypercapnia and acidosis. Hypercapnia occurs when the CO_2 concentrations are high enough that exchange of O_2 and CO_2 on hemoglobin cannot occur effectively at the gills. Acidosis occurs when the CO_2 (or any other environmental acid) levels are high enough to lower the pH of the blood. One of the best understood toxicity pathways is respiratory acidosis, which leads to Root effect reductions in blood-oxygen capacity, even in high oxygen environment (Bernier and Randall 1998). In addition, metabolic acidosis also develops, and can become a principal contributing factor to the acid-base imbalance. The metabolic acidosis leads to increased H^+ levels in the blood and is more difficult to recover from than respiratory acidosis (Bernier and Randall 1998). Recent studies suggest acid-base imbalance may not be the main mechanism for CO_2 lethality, and that hypercapnia may cause a heart failure via an unknown mechanism that is linked to the decrease in stroke volume, and drop in a blood pressure (Lee et al. 2003).

Although research on fish physiology indicates CO_2 should be an effective fish toxicant, its use at the ecosystem scale of entire lakes or ponds has not been attempted and its use may face several logistical constraints. Here we report the results of a pilot study designed to test whether its logically possible to elevate CO_2 levels under the ice in lakes, whether the CO_2 goes into solution or is mainly lost to the atmosphere, how well the CO_2 is mixed throughout the lake water column, and the length of time CO_2 remains elevated.

METHODS

Our preliminary research on CO_2 application indicated using liquid CO_2 would require customized equipment (hoses, valves) that would not be available in the timeframe of the current pilot project. Thus, we decided to use solid CO_2 (dry ice). Moreover, dry ice may be especially effective due to its “boiling” effect when placed in water, as this turbulence likely helps mix the CO_2 into the entire water column.

We selected a 0.4 ha pond on the property of the Carlos Avery DNR facility near Forest Lake, MN (Figure 1). This site was chosen because of its size (because application to a water body < than 1 ac need not be permitted by EPA) and restricted public access. The pond had a maximum depth of 1.3 m (including ice), an average depth of 0.5 m, and on the day of CO_2

application an ice thickness of 0.3 m. Budget and time constraints precluded us from assessing the fish and non-target communities prior to the treatment, but we observed black bullheads, central mudminnows, and fathead minnows on the day of application.

We deployed two sondes (which measured water temperature, pH, and dissolved oxygen every 2 hours) and one CO_2 meter (which measured CO_2 every 30 minutes) at three locations in the center of the pond spaced equal distances apart from each other. The sondes and CO_2 meter were deployed at the midpoint between the bottom of the ice and sediment surface. The CO_2 meter measured CO_2 directly, while the sondes estimate CO_2 indirectly based on lake pH and alkalinity (measured directly when the sondes were deployed). Thus, the two estimates are independent of each other and serve as cross validation on the accuracy of our estimates. The three monitors were deployed on 8 February, five days before the CO_2 application, to get background data. We also collected water samples on 8 February to measure pH, alkalinity, and dissolved inorganic carbon (DIC) back in the lab as a validation of estimates from the three meters at the beginning of the experiment. Similarly, water samples were also collected when the meters were removed after 26 days of deployment to serve as a validation of the meters at the end of the experiment.

Based on the literature regarding toxicity of CO_2 to fish, we set a target concentration of 150 mg/L. We collected water samples in January to estimate lake alkalinity, DIC, and pH, which were used to estimate of CO_2 concentrations and the amount of CO_2 required to raise lake levels to 150 mg/L. However, these estimates weren't available before we ordered the dry ice, so we assumed CO_2 and alkalinity would be similar to other lakes we've worked in. We also assumed some CO_2 would be lost in transport, and some would be lost to the atmosphere before the application holes refroze. Based on these considerations, we purchased 545 kg of dry ice CO_2 (hereafter CO_2) through the UStores at the University of Minnesota.

The CO_2 was applied to the lake on 13 February 2013. The CO_2 arrived in two large coolers on wheels, minimizing loss when being transported. The logistics were also relatively easy, as the coolers were transferred from the delivery truck to DNR trucks without use of any heavy equipment. At the lake we drilled a grid of holes spaced 10 m apart, for a total of 48 holes across the 0.4 ha lake. The holes were placed the maximum distance (5m) from the three meters to test how well the CO_2 mixed under the ice. Holes were drilled at a 45 degree angle to minimize loss of CO_2 to the air and to reduce turbulence from the dry ice keeping the hole from refreezing. Additionally, we used an HVAC pipe inserted into the hole down to the sediment to deploy the ice as far from the hole as possible (Figure 2). We split the 24x24 cm blocks with a spud to get them small enough to fit into the drilled holes. We put ca. 11.3 kg of CO_2 into each of the 48 holes to get a uniform application. We had 2 people drill the holes with power augers, and seven people inserted the CO_2 into the holes. We initially used sleds to move the CO_2 from the truck out on to the pond, but a six-wheel ATV made the job much easier.

RESULTS

The application took just over two hours from start to finish, with a considerable amount of this time used for initial planning since this had never been done. As expected, the CO_2 caused enormous turbulence in the water, even though the holes were drilled at an angle. On several occasions the CO_2 caused bubbling in holes 10 m away. CO_2 could be seen sublimating from the holes, but we found that packing the holes immediately with ice and snow effectively sealed each hole. It should be noted that the air temperature was at the freezing point on the day of the application, so holes would refreeze more quickly with colder, more seasonal air temperatures.

The treated pond was checked weekly for any signs the CO_2 was effecting the surface ice or was being released through cracks. Nothing unusual was observed. A period of warm weather forced us to remove the three meters 26 days after CO_2 application. When drilling holes in the ice to retrieve the 3 meters we noticed the water appeared to be carbonated ("bubbly"), and we saw several dead fish. Results from both the CO_2 meter and the sonde indicated the application was highly successful (Figure 3a). Within three hours after application, CO_2 levels went from the background level of 19 mg/L to over 204 mg/L, and reached a maximum of 274 mg/L one week after application. Additionally, levels stayed above 200 mg/L for the entire 26 days. Assuming the CO_2 rate of decline stayed constant and the lake remained frozen, CO_2 levels would have remained above 150 mg/L for a total of 45 days. The CO_2 addition also dropped the lake pH from 7.03 to 5.97 within three hours of application and it remained below 6.0 for all 26 days (Figure 3b). Finally, the water temperature in the lake dropped $2^{o}C$ after CO_2 addition, and didn't warm to pre-treatment levels until two weeks later (Fig. 3b). We note the peak CO_2 level of 274 mg/L was 83% higher than our target level of 150 mg/L, due to both lower lake alkalinity and less water volume than we expected when we ordered the dry ice.

A formal assessment of the fish and community response to the CO_2 addition were beyond the scope of this pilot study. However, we asked a DNR staff member from Carlos Avery to walk the perimeter of the treated pond after ice out and look for evidence of mortality from the CO_2 application. He reported: "I walked around the pond today and saw 4 dead minnows that have been dead a while, 1 painted turtle that seemed very fresh, a crayfish claw that has been there a while, and what I estimate to be a couple thousand dead snails. I also walked around the adjacent pond and saw only a dozen or so dead snails. I did not note any live creatures in the water today."

Although anecdotal, his observations indicate CO2 levels were lethal.

DISCUSSION AND CONCLUSIONS

Results of this study indicate under-ice CO_2 application to shallow lakes is logistically feasible, that CO_2 mixes well under the ice, that it can reach levels high to be potentially lethal to fish, and that high CO_2 concentrations persist for several weeks following treatment. Thus, our recommendation is that the DNR or other interested parties take the next steps in testing the efficacy of CO_2 as a fish toxicant. We think the key remaining questions are:

1) What CO_2 concentration and duration are required to kill target fish such as bullhead and fathead minnow?

2) Is it more effective to apply CO_2 early or late in the winter due to adjustments of fish physiology to increasing CO_2 as the winter progresses?

3) How will the five-fold variability in lake alkalinity known for lakes across the state influence efficacy of CO_2 addition?

4) What are the non-target impacts of CO_2 addition on invertebrates, amphibians, etc.? Related, how fast will lake chemistry recover, and will changes in pH influence mercury dynamics?

It should also be noted that any future research that scales up to lakes larger than 0.42 ha (1 ac) will require special permits from the US EPA. Requirements for obtaining permits will necessitate planning at least a year in advance of application.

Finally, we have a few other observations based on our experiences with this project. Future CO_2 applications will be quicker and will require staff on site when ATVs or snowmobiles are used to transport CO_2 around the lake. We expect that CO_2 dispersal would be just as extensive with far fewer holes drilled in a given lake, and more CO_2 added to each hole. Based on our observations, CO_2 generated extreme turbulence under the ice, and we think the resulting mixing will extend over distances much larger than we tested, probably in the range of 25 m or more. This will save a lot of time and effort in a CO_2 application, as drilling numerous holes in the ice at an angle was the hardest part. Overall results of this pilot study indicate that CO_2 has good promise as a tool to control populations of undesirable fish in Minnesota lakes.

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Figure 1. Location of the study pond at the MN DNR Carlos Avery station.

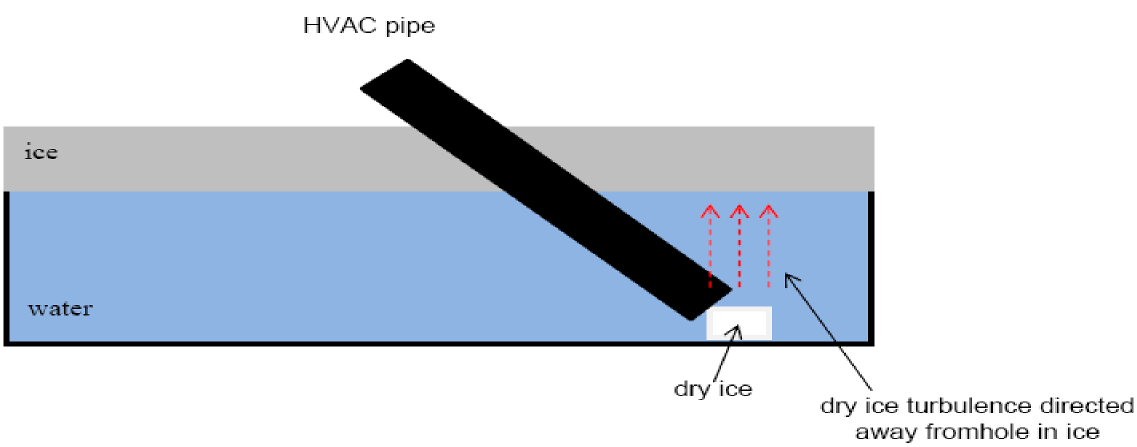

Figure 2. Application strategy showing how angled hole in ice and HVAC pipe minimized dry ice turbulence from reaching the hole in the ice.

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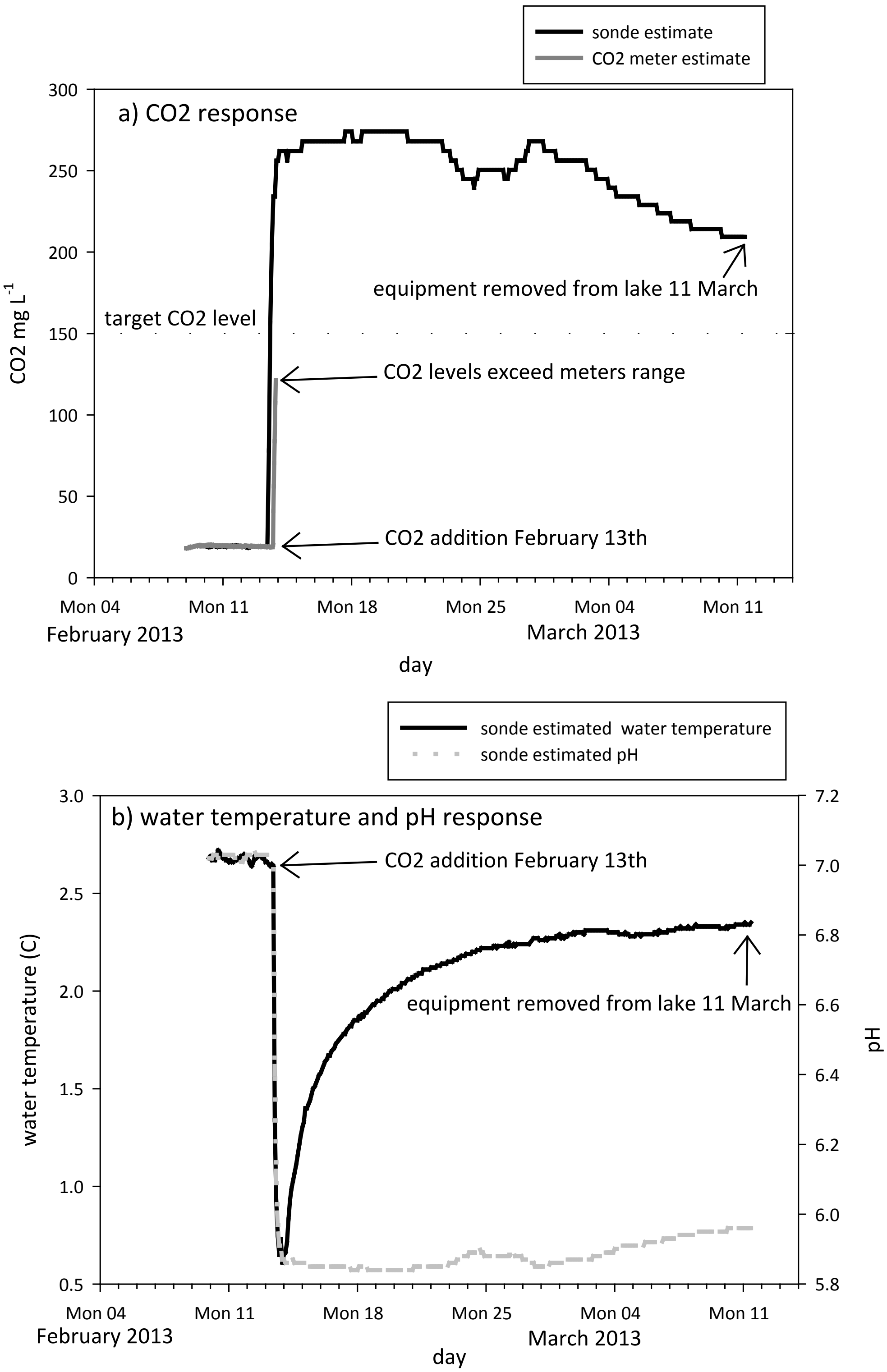

Figure 3. (a) Reponse of CO2 to dry ice addition as estiamted by both sondes and the CO2 meter. (b) Reponse of water temperature and pH to dry ice addition as measured by sondes.

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***Shallow lake rehabilitation: still lots to learn?**

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Shallow lake rehabilitation is complex and often produces unexpected-even disappointing-results. Relatively few management tools are available to lake managers and commonly used general approaches have changed little during the past 50 years. The most widely used strategies are drawdown, removal of undesirable fishes, or combinations of both approaches. These efforts are complicated by multiple factors. For example, inducing drawdown can be confounded by high precipitation, extensive surface water connectivity, and expectations of recreational lake users. Fish removal is also difficult and usually requires application of chemical toxicants, and fish typically re-colonize lakes within a few years after treatment. Restoration of grasslands or other native vegetation within lake watersheds is thought to favor improved water quality in shallow lakes, but these projects are extremely costly and benefits are slow to develop (if they occur at all). We review results of shallow lake restorations and 12 case study lakes and suggest the following generalizations. Shallow lake rehabilitation often triggers improved water quality and habitat suitability for wildlife, but deteriorated conditions typically recur within 5-10 years. Lake sediments show evidence of greatly increased nutrient loading since European settlement. While restoration of upland cover within lake watersheds has many beneficial effects, in-lake characteristics are difficult to link with watershed-scale land cover variables. Accelerated monitoring is critical if managers hope to refine current rehabilitation methods and planning strategies for shallow lakes in Minnesota.

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NESTING ECOLOGY OF RING-NECKED DUCKS IN THE BOREAL FOREST OF NORTHERN MINNESOTA

Charlotte Roy

SUMMARY OF FINDINGS

We have completed 5 years of fieldwork on this research project. We searched 147 wetlands, located 115 ring-necked duck (*Aythya collaris*) nests, marked 66 hens, and followed 32 broods. Nest success during this study ranged from 0.12-0.46, which is comparable to previous studies in Minnesota. Hen survival during the breeding season has not been previously estimated in Minnesota, and ranged 0.54-0.88. Brood survival also has not been estimated previously in Minnesota and was 0.22 for 2008-2012.

INTRODUCTION

The ring-necked duck is a characteristic and important species for the Laurentian Mixed Forest province of Minnesota (Minnesota Department of Natural Resources [MNDNR] 2006), also known as the Boreal or Coniferous Forest biome. Recent surveys near Bemidji have indicated declines in ring-necked duck numbers, despite increases elsewhere in their breeding range (Zicus et al. 2005). Unfortunately, basic information on nest success, hen survival, and brood survival in north-central Minnesota are unavailable, limiting informed interpretation of these local survey data and our understanding of how vital rates affect population growth of ring-necked ducks in the forest. These data are particularly pertinent given the increasing development and recreational use in the forest (MNDNR 2006) and predictions that the spruce-fir forest will shift north of Minnesota as a result of global climate change (Iverson and Prasad 2001).

Nest success, hen survival, and brood survival in the boreal forest are largely unknown. Some data are available for nest success and brood survival in the boreal forest in north-central Minnesota (Hohman and Eberhardt 1998), Michigan (Sarvis 1972) and Maine (McAuley and Longcore 1988, 1989), but this data was collected decades ago. Limited data are available for nest success outside the forest; Maxson and Riggs (1996) studied nest success of ring-necked ducks in the forest-prairie transition during 1985–1987, and Koons and Rotella (2003) compared nest success of ring-necked ducks to that of lesser scaup (*Aythya affinis*) in the parkland of Manitoba. However, neither study examined hen or brood survival during the breeding season. In general, nesting and brood-rearing information for diving ducks are limited in comparison to the data available for dabbling ducks (Yerkes 2000).

Gathering information on vital rates during the breeding season is an important first step to understanding recent population patterns of ring-necked ducks in Minnesota. Although sensitivity analyses of vital rates on population growth rates are not available for ring-necked ducks, sensitivity analyses for mid-continent mallards indicated that nest success explained the most variation (43%) in population growth rates (Hoekman et al. 2002). A similar analysis for the Great Lakes Region indicated that duckling survival (32%) and nest success (16%) accounted for the greatest variation in mallard population growth rates during the breeding season (Coluccy et al. 2008). =

OBJECTIVES

1. To obtain baseline information on ring-necked duck nest success, hen survival, and brood survival before fledging in the forest.

2. To examine how these vital rates vary along a gradient of development and recreational use (e.g., number of dwellings, boat access, proximity to roads).

METHODS

We used multiple methods and data sources to identify lakes to search, including locations of pairs and lone males from a ring-necked duck helicopter survey conducted during 2004–2010 and ground surveys conducted on 10–14 lakes in the Bemidji area beginning in 1969. The survey data were used to identify land cover attributes of wetlands that ring-necked ducks used (US Geological Survey Gap Analysis Program [GAP] types 12 and 13 surrounded by GAP types 10, 14, and 15). We identified 103 lakes within a 40-km (25-mile) radius of Bemidji with land cover attributes similar to those used in the 2 surveys. In 2009, we scouted wetlands in early spring and focused nest-searching efforts on the wetlands where ring-necked ducks had been seen. We excluded lakes considered unsafe to search or where we had been denied access. Each year we added wetlands where we observed ring-necked ducks. During the 5 years of the study, 147 wetlands were searched (Fig. 1).

To locate nests, we searched emergent vegetation on floating bog mats and along wetland margins using bamboo poles and nest drags. When a nest was located, we determined the stage of incubation by candling eggs (Weller 1956) and from the appearance of new eggs in the nest. Nests were monitored every 4–7 days to determine fate (abandoned, depredated, or successful) and Mayfield nest success (Mendall 1958, Mayfield 1975). We determined water depth and distance to open water at each nest after it hatched or failed.

We trapped hens on nests with Weller traps (Weller 1957) to attach radio-transmitters late in incubation. Because we were initially concerned that a surgical transmitter attachment method might be too disruptive to incubating hens, we tried a bib-type transmitter attachment method, which had been used with previous success in wood ducks (Montgomery 1985). This attachment method was faster and less invasive than surgical methods. Hens received a transmitter fastened to a Herculite® fabric bib with dental floss and superglue (total weight of approximately 11 g). We modified the method used unsuccessfully with redheads (*Aythya americana*) by Sorenson (1989) by securing the bib more tightly and by preening the bib into the breast feathers as in Montgomery (1985). After the transmitter was in place, we trimmed any excess fabric so that feathers concealed the transmitter. Due to concerns about low hen survival in 2009 and low brood survival during 2008 and 2009, we changed the transmitter attachment method in 2010. We tried the surgical transmitter attachment method that we had been using for the MNDNR-funded study on post-fledging ring-necked ducks (Korschgen et al. 1996). However, we used a local anesthetic (i.e., lidocaine) instead of isoflurane so that we could do surgeries in the field (Corcoran et al. 2007). We also used propofol, injected intravenously, to reduce nest abandonment (Rotella and Ratti 1990, Machin and Caulkett 2000) on 6 hens in 2010 and on all hens in 2011 and 2012. When propofol was used, hens were placed on nests rather than being released from the edge of the wetland.

After nests hatched, we attempted to monitor broods every 3–7 days. During each observation, we counted the ducklings present, and when possible, aged them from a distance based on plumage characteristics (Gollop and Marshall 1954). Broods were monitored until ducklings reached age Class III (39-49 days old) or until total brood loss occurred. We considered hens to have lost their entire brood when hens were observed without any ducklings for 3 consecutive observations or if the hen was found >16 km (10 miles) from the nesting lake. We continued to monitor hens after the brood-rearing period for as long as they could be tracked before migration to examine their survival using the Kaplan-Meier method (Kaplan and Meier 1958).

In 2011, the state government shutdown occurred 1-20 July, during peak weeks of ring-necked duck hatching. We were still finding nests and 5 nests were still active at the time of the shutdown. We attempted to check nests that had been active and locate broods when state government activities resumed. However, the shutdown precluded data collection according to the methods described above.

RESULTS

We located 115 active nests, marked 66 hens, and followed 32 broods. We searched for nests on 37 wetlands for a total of 67 searches (17 wetlands searched once and 20 wetlands searched >1 time) between 22 May and 22 July 2008, 37 wetlands searched 55 times (21 wetlands once and 16 wetlands searched >1 time) between 29 May and 22 July 2009, 72 wetlands searched 128 times (33 wetlands once and 39 wetlands searched >1 time) between 19 May and 12 July 2010, and 76 wetlands were searched 107 times (54 wetlands once and 22 wetlands searched >1 time) between 23 May and 30 June 2011. We searched 79 wetlands 140 times (35 wetlands once, and 44 wetlands searched >1 time) between 15 May and 10 July 2012.

Nest Survival

We located 14 active ring-necked duck nests on 10 wetlands in 2008, 20 active nests on 11 wetlands in 2009, 32 active nests on 17 wetlands in 2010, 22 active nests on 16 wetlands in 2011, and 27 active nests on 14 wetlands in 2012. In 2008, 8 nests hatched, 3 were depredated, and 3 nests were flooded by rising water levels following rain events. Average clutch size for nests that were incubated was 9.1 $\pm$ 0.6 (mean $\pm$ SE, range = 7–15, n = 12 nests with 109 eggs) and 86.6 $\pm$ 0.1% of eggs hatched in successful nests. In 2009, 7 nests hatched, 9 were depredated, and 4 were abandoned, with at least 2 cases of abandonment likely due to trapping and 1 due to flooding. The average clutch size for incubated nests was 8.3 ± 0.3 (range = 7–11, n = 19 nests with 158 eggs) and 89.5 ± 0.6% of the eggs hatched in nests that were successful. In 2010, 13 nests hatched, 9 were depredated, 6 were abandoned after trapping and transmitter attachment, and 2 were abandoned for other reasons. We could not determine the outcome of 1 nest based on evidence at the nest site, and 1 failed because the hen died during transmitter-implantation surgery. We began using propofol on all hens captured later in the field season because 5 of 13 hens marked without propofol had abandoned their nests. Average clutch size for incubated nests was 8.3 $\pm$ 0.3 (range = 5–10, n = 30 nests with 250 eggs) and 84.5 $\pm$ 0.1% of eggs hatched. In 2011, 6 nests hatched, 3 were abandoned (2 to investigator disturbance and 1 for unknown reasons), and 13 were depredated. Average clutch size was 8.8 $\pm$ 0.4 (range = 4-11, n = 19 nests with 166 eggs) and hatching success was 85.0 $\pm$ 0.2%. In 2012, 11 nests hatched, 5 were abandoned (1 to investigator disturbance, 2 to flooding), 7 were depredated, 1 was unviable, 2 hens were killed before the nest could hatch, and 1 fate was unknown. Average clutch size was 7.8 $\pm$ 0.3 (range = 4-10, n = 27 nests with 210 eggs) and hatching success was 85.2 + 8.4%. Mayfield nest success for a 35-day period of laying and incubation was 30% in 2008, 27% in 2009, 46% in 2010, 12% in 2011, and 23% for 2012.

Hen Survival

We radio-marked 8 hens in 2008, 14 in 2009, 19 in 2010, 9 in 2011, and 16 in 2012. In 2008, 2 hens died due to predation during the tracking season; 1 lost her nest late in incubation and the other had a brood. Both of these birds had been observed preening more than other birds with transmitters, although this behavior occurred during the first 2 weeks after marking and then subsided. Both deaths occurred after this period, one 3 weeks post-marking and the other 4 weeks post-marking. All birds in 2008 continued to nest and rear broods after transmitter attachment, with the exception of birds that lost their nests to flooding. In 2009, 6 hens died during the monitoring period (17, 20, 32, 33, 55, and 84 days post-marking). Evidence obtained at the recovery sites indicated that radioed birds were either depredated or scavenged by avian predators (3) or by mammalian predators (1). Additionally, there were 2 cases in which a probable cause of death could not be determined, because the transmitter was underwater and no carcass was found. All of the hens that died did not have broods at the time of death; 3 lost their nest late in incubation, 1 abandoned her nest due to trapping, and 2 lost broods early after hatching. In 2010, only 1 hen died during the monitoring period. She died 17 days after marking and appeared to have been killed by a mammalian predator. She did not have a brood. Twelve of 19 transmitters dehisced 55.1 ± 6.0 days (range = 30–121 days) after attachment. In 2011, 2 hens were depredated, one by a mink and the other by an unknown predator. One hen did not recover from anesthesia and her nest was censored. Another hen dehisced her transmitter in mid-August, 53 days after marking. In 2012, 2 hens were depredated during incubation and one was depredated during brood-rearing by unknown predators. One hen did not recover from propofol and her nest was censored. Three transmitters dehisced 42.7 + 8.4 days (range = 28-53 days) after marking. Hen survival through mid-September was 0.80 ± 0.18, 0.54 ± 0.08, 0.88 ± 0.11 for 2008-2010, respectively. In 2011 and 2012, tracking was terminated in mid-August because few birds were located in the study area. In previous years, tracking success was higher because a concurrent telemetry study on post-fledging ring-necked ducks allowed for a larger search area at no additional cost. Hen survival through mid-August in 2011 and 2012 was 0.69 ± 0.19 and 0.70 ± 0.10 respectively.

Brood and Duckling Data

In 2008, 7 radio-marked hens had broods (*n* = 57 ducklings). One brood survived to fledge 5 ducklings. Other broods dwindled slowly, with total brood loss at the IA (1), IB (1), IC (1), and IIA (2) age classes (Gollop and Marshall 1954). The fate of 1 brood could not be determined, because the hen died when the brood was at the IIA stage, and we could no longer relocate the ducklings without the marked hen. We also monitored the brood of 1 unmarked hen that was not trapped in time to give her a transmitter. Her brood made it to the IC stage, but they were not observed again and their fate was uncertain.

Seven broods were monitored in 2009 (*n* = 56 ducklings). Total brood losses occurred at IA (3), IB (1), and IC (1) age classes. One brood fledged 2 young. Another brood matured to IIA before the hen left the wetland, after which time 1 duckling was seen on the wetland and no hens were present.

We observed 6 broods in 2010 (*n* = 40 ducklings); 3 broods survived to age Class III and likely fledged 14 ducklings, 1 brood was located as Class IA ducklings, but the hen was not located again, 1 brood survived until age Class 1A, and another brood survived to age Class IB. Seven marked hens were believed to have hatched ducklings, but were not located with broods before total brood loss.

In 2011, following the government shutdown, we were able to locate 5 hens and follow 2 broods that were still alive. Both broods fledged; one brood of 3 ducklings made it to flight (50 days) and the other had 6 ducklings survive until at least class III (42 days, and most likely flight).

In 2012, we observed 7 broods (*n* = 39 ducklings); 2 broods were last observed as IA ducklings, 1 brood made it to Class IB, 3 broods made it to Class IIA, and 1 brood survived to Class III and likely fledged 4 ducklings. Two additional broods were never observed and were likely lost as Class IA ducklings. Thus, brood survival to fledging for all years was 0.22 ± 0.07.

Brood movements also were observed during various stages of brood-rearing (Classes I and II). Distances moved ranged 148-2,273 m but were generally less than 860 m (*n* = 8). Movements were to both larger and smaller wetlands in the vicinity of nests.

DISCUSSION

Our success finding nests was comparable to that in other studies that found ring-necked duck nests (45 nests in 3 years, Maxson and Riggs 1996; 35 nests in 2 years, Koons and Rotella 2003, 188 nests in 6 years by R. T. Eberhardt). Our nest survival rates were lower than Eberhardt's estimates of 44% during 1978-1984 in northern Minnesota (*n* = 188, Hohman and Eberhardt 1998), but more similar to the 34.1% reported by Maxson and Riggs (1996) for west central Minnesota during 1985-1987, *n* = 26). Interestingly, nest success was lower for all diving ducks in the 80's than in the 60's (43.3% reported by Jessen et al. 1964 vs. 32.7% reported by Maxson and Riggs 1996). However, early estimates of nest success did not adjust for the stage at which nests were found, and thus early estimates are likely biased high (Mayfield 1975). If we did not correct for nesting stage, our nest success estimate was 44%, which was the same as Eberhardt's earlier estimate. Thus, nest success appeared to be comparable to historical levels in Minnesota.

The causes of nest failure in our study (11% flooding, 75% depredation, 11% abandonment, 3.5% unknown fate, and 1.8% unviable) were similar to other studies (16–24% flooding, 67–80% depredation, and 5% abandonment; Mendall 1958, McAuley and Longcore 1989), when we exclude nests where abandonment was attributed to investigator disturbance (following nest trapping or surgery). Abandonment may have been slightly higher than other studies, but losses to flooding and abandonment might have been slightly confounded if flooding was interpreted as abandonment because signs of flooding were temporary. Thus, we believe that the causes of nest failure in our study were comparable to other studies.

Estimates of egg hatching success appeared to be lower than those of Eberhardt's previous study in north-central Minnesota (94%, Hohman and Eberhardt 1998), as well as studies in Maine (91%, McAuley and Longcore 1989) and Michigan (90%, Sarvis 1972). Springs during this study were wet and rainy, which may have chilled and flooded some eggs in nests. However, we found evidence that some eggs may have been unviable when laid. For example, we found a clutch where candling indicated that 2 of 5 eggs were not viable the day it was found, and this was later confirmed when only 3 eggs hatched. Similarly, we were able to tell that 1 egg of 7 was bad during the first visit of another nest. This nest was later depredated. Other instances of eggs not developing were also noted, but were not immediately detected when the nest was first found. For example, in a nest of 15 eggs we observed small dark spots in several eggs when candling midway through incubation and only 4 eggs hatched. Another case occurred with a clutch of 9 eggs where candling indicated that some eggs were not developing and it was later abandoned. In another nest of 4 eggs that was abandoned, we opened the eggs and it was obvious that no development had occurred while the hen was incubating. We did not open most eggs that did not hatch, so we do not know how common this was. Parasitism may have also contributed to poor hatching success; 15 eggs is a very large clutch for a ring-necked duck. Further investigation would be necessary to understand the factors influencing egg viability and any role of parasitism.

Hen survival rates for the period June–mid-September were comparable to reports for hen mallards during April–September (0.80, Cowardin et al. 1985; 0.60, Blohm et al. 1987; 0.67, Brasher et al. 2006). However, given the shorter duration of measurement in our study, we expected hen survival to be higher. Furthermore, mortality was expected to be greatest during incubation, and hens were not marked for the entirety of incubation, so survival was lower than expected. Brood survival rates also seemed low. Brood survival in ring-necked ducks has only been examined previously in Maine (77% to 45 days, n = 64, McAuley and Longcore 1988). Duckling survival in the same study was 37% (n = 381).

Fieldwork for this study was completed in 2012. Data are being proofed and analyzed for a final report and manuscript preparation. Results should be viewed as preliminary and are subject to change.

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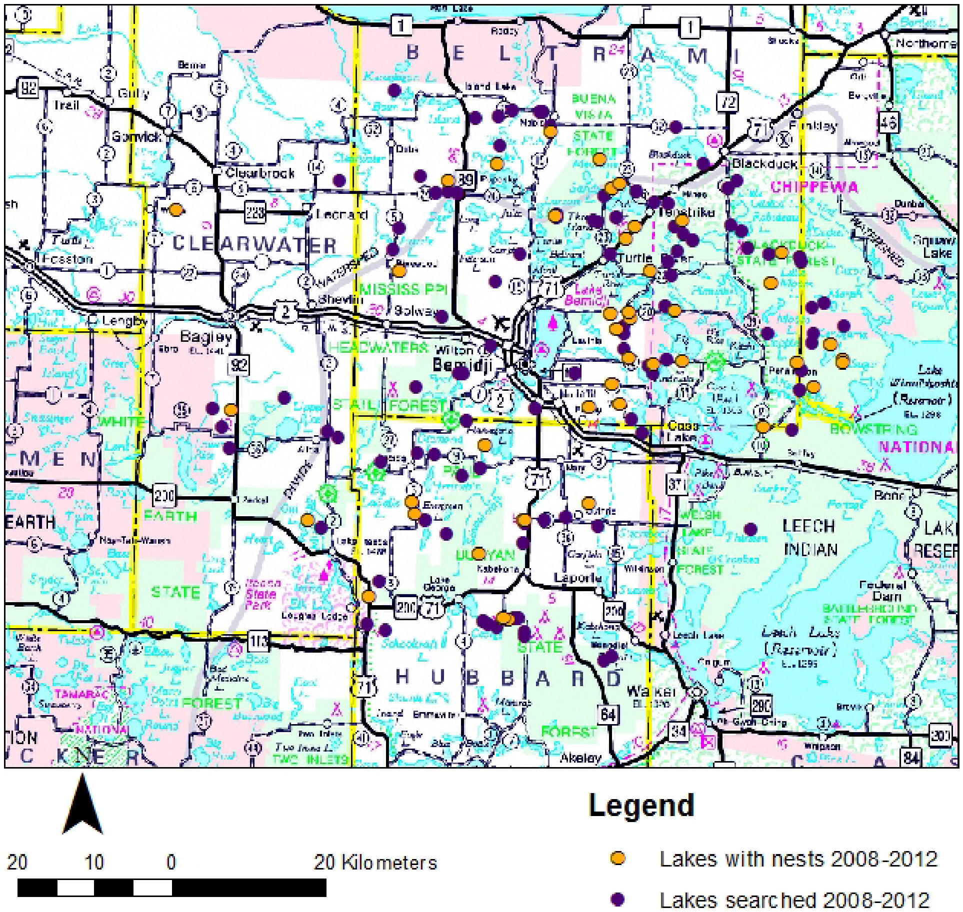

Figure 1. Wetlands searched for ring-necked duck nests in north-central Minnesota during 2008–2012.

INVESTIGATION OF TREMATODES AND FAUCET SNAILS RESPONSIBLE FOR LESSER SCAUP DIE-OFFS

Charlotte Roy

SUMMARY OF FINDINGS

Two waterfowl die-offs occurred on Lake Winnibigoshish in 2012. During spring, several hundred lesser scaup (*Aythya affinis*) were observed moribund, unable to fly and/or keep their heads up, and necropsy later confirmed trematodiasis. During the fall of 2012, few lesser scaup were observed on Winnibigoshish, but those that were observed were sick. Trematodiasis was again confirmed through necropsy of dead individuals. We also observed <100 sick birds at Bowstring and Round lakes, but we were unable to collect birds from these locations.

The invasive faucet snail is the only known first host of the trematodes that cause the die-offs, so we sampled faucet snails at all sites where they were known to occur in interior Minnesota; Lake Winnibigoshish, Upper and Lower Twin lakes, the Shell River, First Crow Wing Lake, and the Crow Wing River. In 2012, several new sites were added due to new detections of the faucet snails. We added Second Crow Wing Lake and several ponds on the White Earth Nation. We also sampled Bowstring and Round lakes for faucet snails, because these lakes are known to be important to migrating lesser scaup. We report the first detection of faucet snails in Bowstring Lake.

INTRODUCTION

During the autumns of 2007 and 2008, thousands of lesser scaup and hundreds of American coots (*Fulica americana*) died on Lake Winnibigoshish in north-central Minnesota. These deaths were attributed to trematodiasis caused by non-native intestinal trematodes (*Cyathocotyle bushiensis*, *Sphaeridiotrema spp.,* and *Leyogonimus polyoon)* and concerned both waterfowl hunters and non-consumptive users.

The trematode species responsible for the die-offs have a complex life cycle that involves two intermediate hosts. The faucet snail (*Bithynia tentaculata*), a non-native species from Europe (Sauer et al. 2007), is the only known first intermediate host of these trematodes in the Midwest and also serves as the second host for *C. bushiensis* and *Sphaeridiotrema spp.* The second host of *L. polyoon* is one of a variety of larval aquatic insects, including damselflies (Zygoptera) and dragonflies (Odonata) (US Geological Survey, National Wildlife Health Center (NWHC), unpubl. data). Adult trematodes develop in waterfowl after they consume infected snails and in American coots (*Fulica americana*) and common moorhens (*Gallinula chloropus*) after consumption of infected insects. Parasite eggs are then defecated by sick birds and later ingested by snails, continuing the cycle. Because of this complex life cycle, the dynamics of faucet snail distribution and transmission of these parasites to lesser scaup and other birds are poorly understood.

The first U.S. detection of the faucet snail was in Lake Michigan in 1871 (Mills et al. 1993). It has since been documented in the mid-Atlantic states, the Great Lakes Region, and Montana, and undoubtedly will continue to spread (Sauer et al. 2007). In 2002, the faucet snail was detected in the Upper Mississippi River. Since then, trematodiasis has killed an estimated 52,000-65,000 waterbirds, primarily lesser scaup and American coots, but also dabbling ducks such as blue-winged teal (*Anas discors*), northern shoveler (*Anas clypeata*), mallard (*Anas platyrhynchos*), American black duck (*Anas rubripes*), and northern pintail (*Anas acuta*); diving ducks such as ring-necked ducks (*Aythya collaris*) and redheads (*Aythya americana*); and other waterfowl such as ruddy ducks (*Oxyura jamaicensis*), buffleheads (*Bucephala albeola*), and tundra swans (*Cygnus columbianus,* R. Cole, NWHC, pers. comm.).

The faucet snail was detected in Lake Winnibigoshish in the spring of 2008, following the loss of 7,000 Lesser Scaup and a few hundred coots to trematodiasis the previous fall (Lawrence et al. 2008). In 2008, 2,000 more birds died (Lawrence et al. 2009). The severity of the outbreaks seems to have lessened in Lake Winnibigoshish over time. This may be because fewer birds are stopping over on the lake during migration or there may be another explanation related to the disease cycle. In any event, these outbreaks are highly visible and attract the media, which can spur public concern and a desire for action.

In recent years, new areas have been designated as infested with faucet snails in north central Minnesota. The faucet snail was first detected in Upper and Lower Twin lakes and the Shell River in 2009. In 2010, the Crow Wing River was designated as infested with faucet snails, and in 2011, First Crow Wing Lake and Second Crow Wing Lake were added to the list of waters infested with faucet snails. In 2012, several new ponds were designated as infested on the White Earth Nation. These newly designated sites may afford us additional opportunities to learn about this disease cycle.

We examined the factors associated with faucet snail abundance and distribution, parasite prevalence within snails, and the influence of snail densities and site attributes (e.g., water depth, distance from shore, substrate composition) on lesser scaup foraging. For example, depth influences the amount of work that scaup have to do against buoyancy. Shallow depths are thus important to foraging scaup (Jones and Drobney 1986, Mitchell 1992). If such depths are also preferred by faucet snails, then the potential for exposure will be much higher than if snails prefer dissimilar water depths. The profitability of food items will vary as a function of depth, density, and prey type among other things (Lovvorn and Jones 1991, Lovvorn et al. 1991, Beauchamp et al. 1992, de Leeuw and van Eerden 1992, Lovvorn 1994).

OBJECTIVES

1- Improve understanding of lesser scaup foraging as it relates to faucet snail and other food source distribution and density, including water depth, distance from shore, and substrate composition
2- Examine factors (e.g., temperature, substrate, vegetation, other snail species) that are associated with the distribution and movement of faucet snails
3- Examine the factors that influence the prevalence of the parasites in faucet snails (e.g., snail density, temperature, microhabitat, time of year)
4- Examine how faucet snail distribution varies during spring, summer, and fall

METHODS

During 2012, we sampled faucet snails at the same locations sampled in 2011, and we added sampling sites at the newly designated First Crow Wing Lake, Crow Wing River, and Second Crow Wing Lake (Figure 1a,b). New faucet snail infestations were discovered in ponds at the White Earth Nation during the summer of 2012, so we added these ponds to our sampling schedule in the fall of 2012 (Figure 1c). We sampled during spring, summer, and fall at the same points within a lake or river (Table 1a,b). In small lakes (<405 ha), we used transects that traversed the entire length of the lake and across a range of depths. In large lakes, we used index areas with points stratified by depth for sampling. In Lake Winnibigoshish, we had 2 index areas, the West Winni Index Area and the East Winni Index Area, which were 5-6 km along the longest dimension and approximately 2 km in width. In rivers, we sampled points at regular intervals (500 m) along the infested corridor for a maximum length of 10 km. In small ponds, we placed sample points ~100 m apart in such a way as to attempt to maximize the number of sampling locations in each pond (diameter 75-320 m).

We used 2 sampling methods; we used a bottomless sampling cylinder (0.2 m^2) at 30 and 60 cm depths for comparisons with an ongoing study on the Upper Mississippi River, and we also sampled with a benthic sled to standardize our protocol for all depths. We dragged the sled a distance of 1.2 m at deeper depths to examine how snail distribution varied within a water body. We collected data on microhabitat variables at each point to examine relationships to snail distribution, the snail community, and parasite prevalence. These included substrate (e.g., silt, rock, sand, vegetated, muck), temperature (C^o), water depth (cm), and a secchi depth (cm) reading was taken 8 times (4 times on the way down and 4 times on the way up) from the shaded side of the boat and averaged. At each snail collection site, we determined pH, dissolved oxygen (mg/L), conductivity (μS/cm), and salinity ($^o/_{oo}$) with a Hach Company (Loveland, Colorado) HQd portable meter that was calibrated daily for pH and weekly for conductivity. Flow (mps) was measured at 60% of the total depth (from the surface) with a Global Water Instrumentation (Gold River, California) flow probe when flow was detectable and averaged over a 40 s interval (the USGS "6 tens method").

Invertebrate samples were stored in the refrigerator until processed. We used a magnifying lens and microscope as needed to identify all invertebrates to Order and noted their presence in each sample. We identified all snails to genus and counted their numbers in each sample. We determined the size of *B. tentaculata* and similarly sized *Amnicola* spp. with calipers, as measured along the central axis from the apex. Parasite prevalence was determined for all samples possessing at least 50 *B. tentaculata* (R. Cole, NWHC, unpubl. data). For samples possessing 10-49 *B. tentaculata,* we collected additional snails while in the field from the same location at the same time to increase the number of samples for which we could do prevalence. These additional snails were not used in the determination of snail abundance at the site. Trematode stages (cercariae or metacercariae), species (*C. bushiensis, S. globulus, L. polyoon*), and numbers were also recorded in the lab.

Each season, we collected a water sample at each sample pond, lake, or river and sent it to the Minnesota Department of Agriculture for analysis. Total phosphorus (ppm), nitrite plus nitrate nitrogen (ppm), chlorophyll a (ppb), total alkalinity (ppm), ammonia nitrogen (ppm), and calcium (ppm) were quantified but have not yet been interpreted.

We also identified sites where lesser scaup foraged and collected benthic samples at these locations. These sites were identified through observations of birds from shore or from a boat. We determined the location of rafts of scaup using a compass from 2-3 observation points, which was plotted in ArcMap version 10 (Environmental Systems Research Institute, Inc., Redlands, California) to determine the area occupied by the birds. We then placed a transect through this area and sampled at 100 m intervals. Food densities, water depths, distance from shore, lake size, and substrate composition at these foraging locations were recorded using the same techniques as snail sampling.

We also collected scaup carcasses during die-offs at study lakes for confirmation of trematodiasis by the NWHC in Madison, Wisconsin. Additionally, Bowstring and Round lakes are known for having large number of scaup, particularly in the fall, and have been the sites of trematodiasis die-offs in the past. We monitored Bowstring and Round lakes for scaup die-offs during the spring and fall. Staff from the Minnesota Department of Natural Resources-Grand Rapids office also made regular visits to Winnibigoshish, Round, and Bowstring lakes throughout the fall season to check for sick birds.

RESULTS

Faucet snails

We detected faucet snails at both index areas on Lake Winnibigoshish, Upper and Lower Twin lakes, the Shell River, First Crow Wing Lake, Crow Wing River, and the newly sampled White Earth Ponds (Tables 1a,b). We also report the first detection of faucet snails in Bowstring Lake.

Preliminary analysis indicates that faucet snails are more abundant during the summer than other seasons. Faucet snails also appear to move into shallower depths in the summer, perhaps to reproduce on vegetation, with a return to deeper depths in fall and spring. Additional data collection will help determine whether this is a robust seasonal pattern. At the Twin lakes, populations of faucet snails may be expanding and increasing. Further data collection will be necessary to determine whether separation of population growth and expansion from seasonal patterns is possible. More formal analyses will be included in subsequent reports.

Trematodes

Both *C. bushiensis* and *Sphaeridiotrema spp.* were detected on Lake Winnibigoshish, Lower Twin Lake, the Shell River, Bowstring Lake, First Crow Wing Lake, the Crow Wing River, and the White Earth ponds. Only *C. bushiensis* was detected in samples from Upper Twin Lake this year, likely because snails are not abundant there yet and thus samples sizes were small. *Sphaeridiotrema spp*. has been detected there in previous years of this study. Prevalence (proportion of snails infected) of *C. bushiensis* was generally higher than that of *Sphaeridiotrema spp.* within a water body. Prevalence was generally highest at the West Winni Index Area and Shell River. The intensity (number of parasites in infected snails) of parasite infections was also highest at these two locations.

Scaup

We observed large rafts of scaup during our visits to Bowstring, Round, and Winnibigoshish lakes (Table 2). Foraging was always confirmed in the spring, but in the fall we were not always able to confirm foraging before the birds began to flush. Two die-offs occurred on Lake Winnibigoshish in 2012. During spring 2012 a raft of 2,000 birds were observed on the west side and several hundred sick birds were documented (Table 2). Nine birds (2 greater scaup, 7 lesser scaup) were collected and sent to the NWHC. Trematodiasis was confirmed. During fall 2012, few birds were observed on Winnibigoshish but sick lesser scaup were observed at two locations; near Mallard Point on the east side of Lake Winnibigoshish and north of Sugar Lake on the west side of Winnibigoshish. Five lesser scaup were collected from this die-off and sent to the NWHC. Again, trematodiasis was confirmed. Sick birds were also observed at Bowstring (n ~ 25) and Round (n ~ 50-75) lakes but birds could not be collected for confirmation of trematodiasis because they were still able to swim away at the time of site visits.

DISCUSSION

This report summarizes activities for the second year of field work (spring, summer and fall 2012). Fall 2010, a pilot season, and the 2011 field season were included in earlier reports. Data entry and analysis are preliminary and still underway. More formal analyses will be included in subsequent reports. We plan to continue using the same methodology, adding additional lakes with faucet snails as they become known, through fall 2013.

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Table 1a. Sampling sites for faucet snails in infested northern Minnesota water bodies during spring, summer, and fall 2012. Number of points refers to the number of points sampled each season of sampling.

Location	No. seasons sampled	No. sample points	Faucet snails detected
East Winni Index Area	3	80	Yes
West Winni Index Area	3	80	Yes
Upper Twin Lake	3	24	Yes
Lower Twin Lake	3	39	Yes
First Crow Wing Lake	3	37	Yes
Second Crow Wing Lake	3	18	No
Crow Wing River	3	18	Yes
White Earth Ponds	1(Fall)	24	Yes
Shell River	3	22	Yes
Total		978	

Table 1b. Sampling sites for faucet snails associated with lesser acaup in northern Minnesota water bodies during spring and fall 2012. Number of points refers to the number of points sampled each season of sampling.

Location	Season sampled	No. sample points	Faucet snails detected
Lake Winnibigoshish	Spring	10	Yes
Crow Wing River	Spring	1	Yes
Bowstring Lake	Spring/Fall	6,14	Yes
Round Lake	Spring/Fall	12,13	No
Total		56	

Table 2. Reports of scaup observed by Minnesota Department of Natural Resources staff on lakes in northern Minnesota during spring and fall 2012. Scaup that failed to escape approach or had drooping heads were considered to be sick; dead birds were typically found along the shoreline.

Location	**Date**	**Total no. of scaup observed**	**No. of sick or dead scaup observed**
		SPRING	
Winnibigoshish	4/4/12	60	None[a]
Lower Twin	4/5/12	2	0
Winnibigoshish	4/12/12	700-800	None[a]
Lower Twin	4/14/12	64	None
Winnibigoshish	4/17/12	3,500	None[a]
Winnibigoshish	4/22/12	5,000	35 sick[b], 200 suspect
Winnibigoshish	4/23/12	3,500	1 dead[a]
First Crow Wing	4/29/12	60-70	None[a]
Winnibigoshish (Third River)	5/3/12	2,500	50 sick
Bowstring	5/3/12	250	None[a]
Round	5/3/12	450	None[a]
Winnibigoshish (Third River)	5/5/12	200-250	3 sick[a]
Bowstring	5/5/12	560	None
Round	5/5/12	360	None[a]
Bowstring	5/9/12	300	None
Round	5/9/12	350	None[a]
		FALL	
Winnibigoshish	10/19/12	Not reported	Sick scaup
Bowstring	10/24/12	2,000	None[a]
Round	10/24/12	4,000	50[a]
Winnibigoshish	10/24/12	Few scaup	Sick scaup reported by hunter[b]
Winnibigoshish	10/27/12	Few scaup	Dead scaup[b]
Bowstring	10/31/12	1,100	25 sick, ~100 suspect[a]
Round	11/3/12	1,500-2,000	None[a]
Winnibigoshish	11/5/12	0	20 dead scaup, few suspect
Round	11/7/12	500-600	15-20 sick scaup[a]
Winnibigoshish	11/12/12	No raft	<25 sick scaup
Bowstring, Round	11/13/12	Freezing up	None observed

[a] Benthic samples collected below scaup.
[b] Scaup were collected and sent to the NWHC to be tested for trematodiasis.

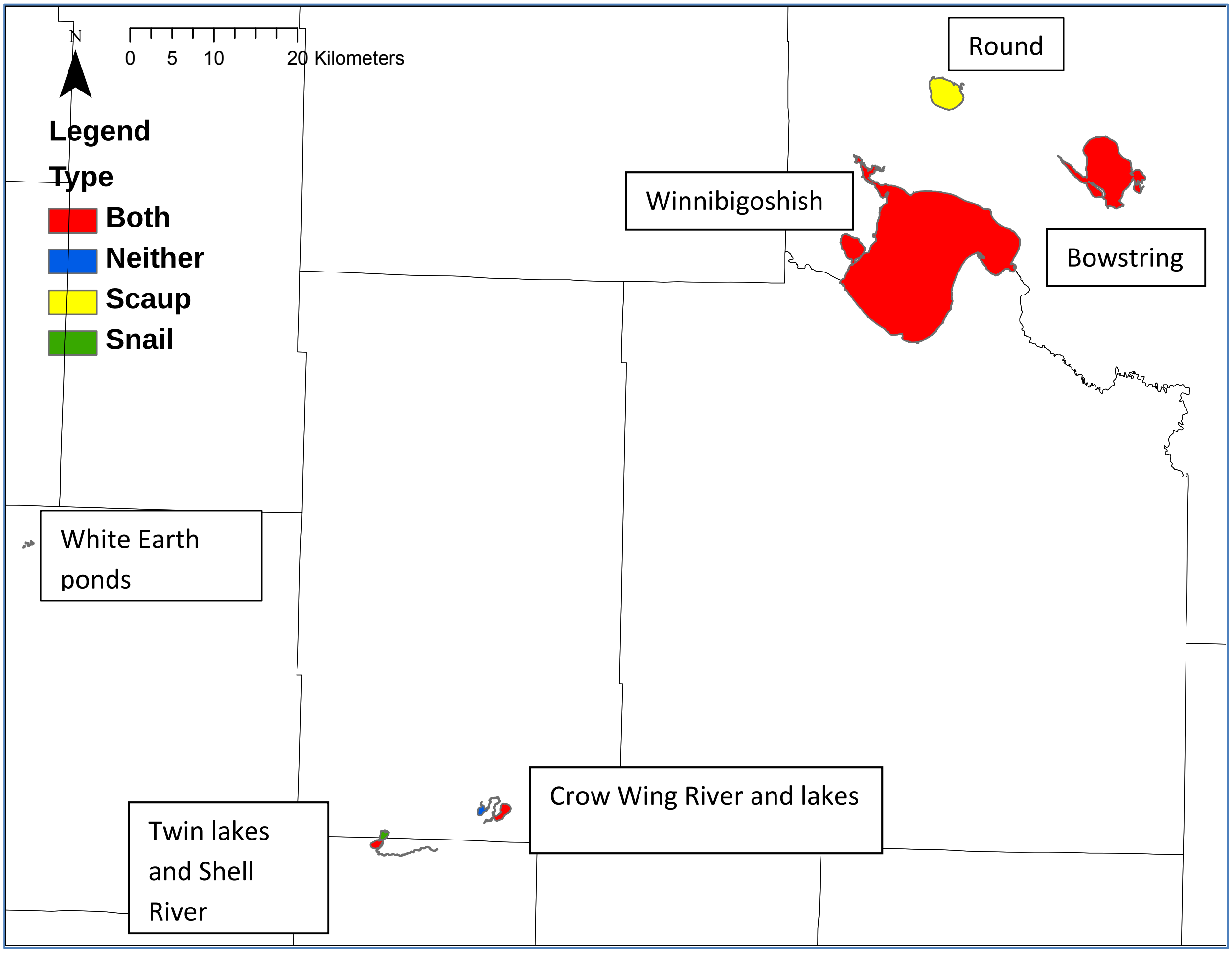

Figure 1a. Presence of faucet snails and scaup at lakes and rivers sampled in northern Minnesota during 2012. County lines are shown. Figures 1b and 1c zoom into the southern sampling area and White Earth ponds, respectively.

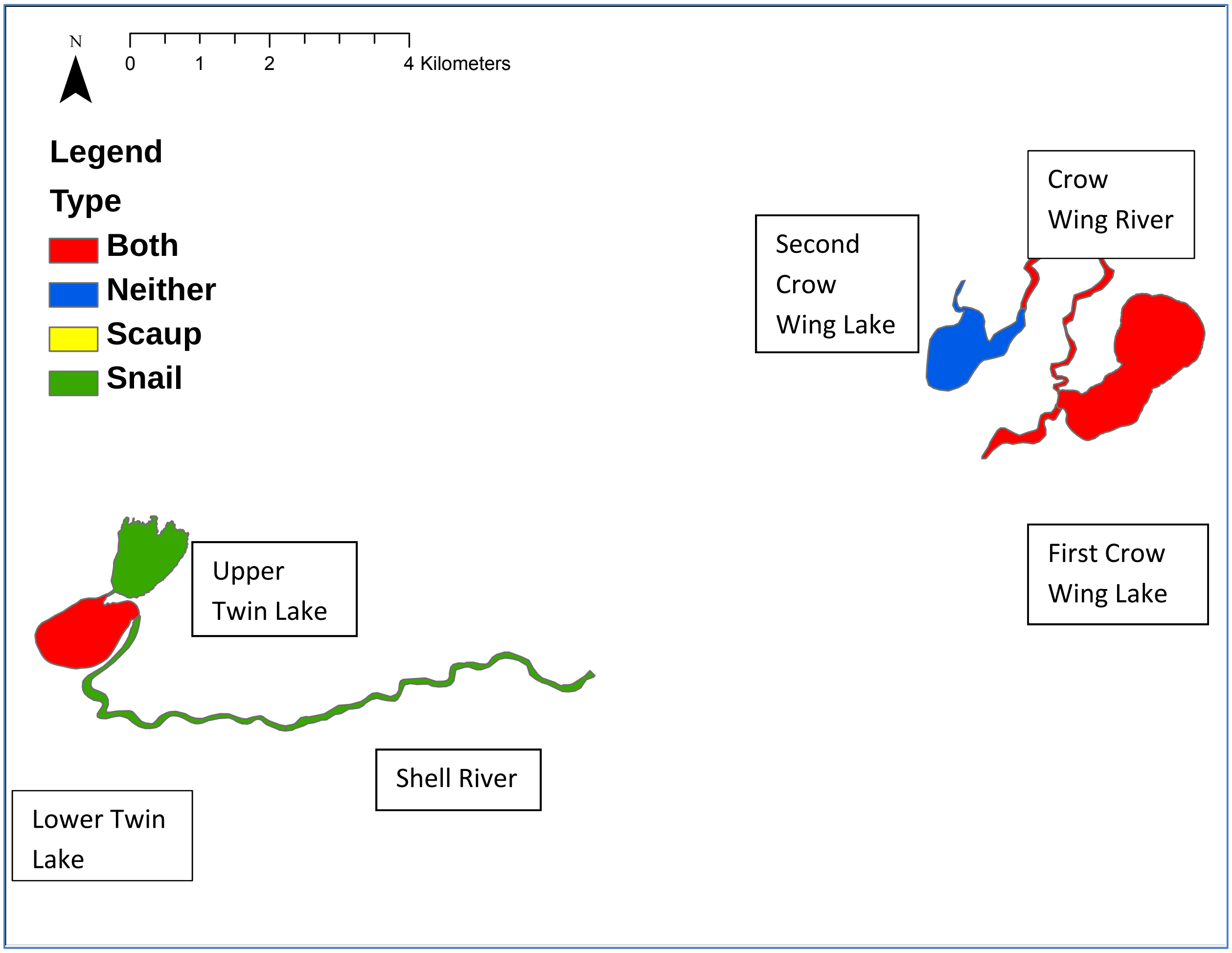

Figure 1b. Presence of faucet snails and scaup in lakes and rivers sampled in the southern portion of the sampling area in north-central Minnesota during 2012.

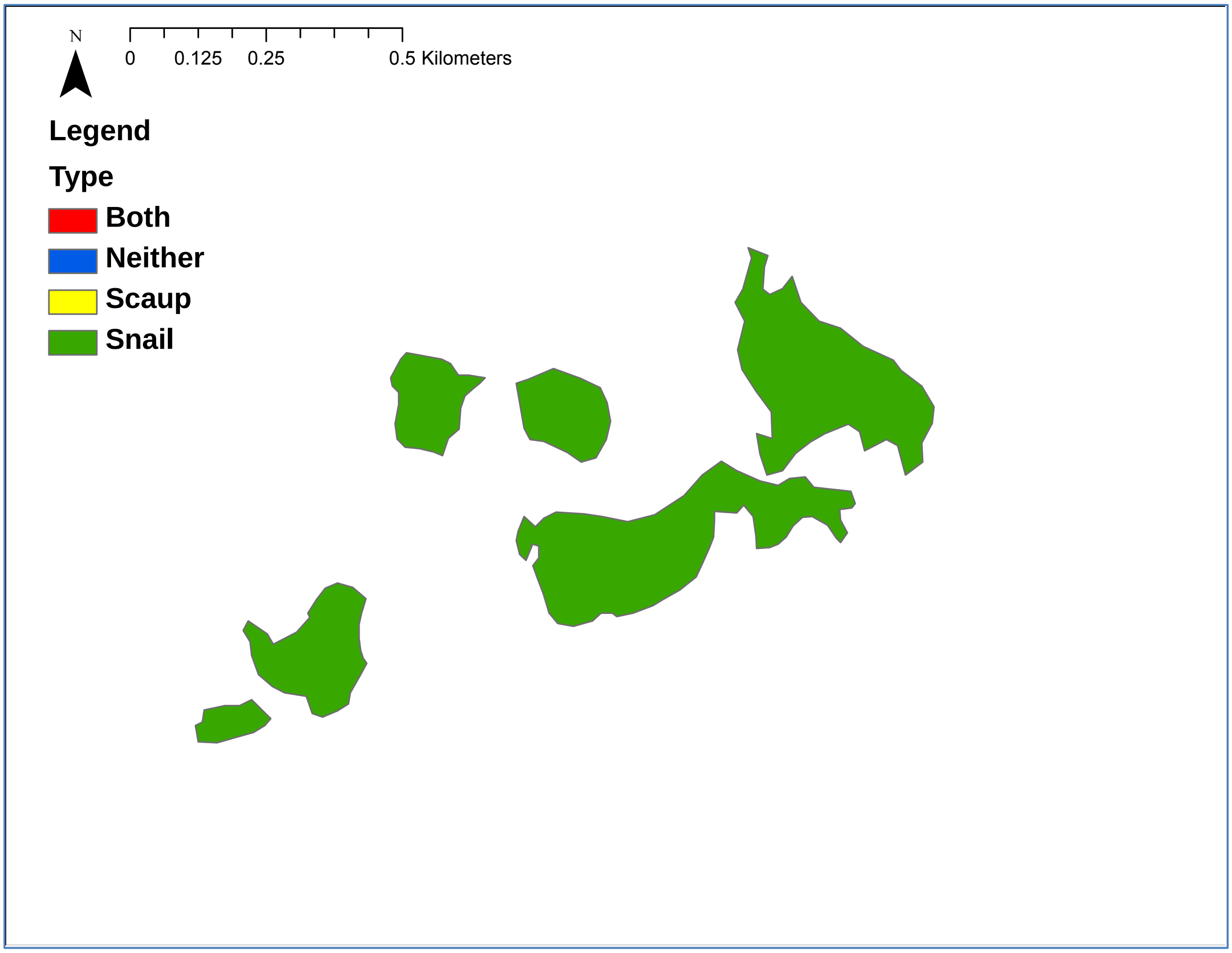

Figure 1c. Infested ponds on the White Earth Nation of northern Minnesota that were sampled for faucet snails and their trematodes during 2012. Scaup were not observed at these ponds during our visits.