Section of Fisheries
INVESTIGATIONAL REPORT

No. 477

INFLUENCE OF WALLEYE FINGERLING PRODUCTION ON WETLAND COMMUNITIES

September 1999

Minnesota

Division of Fish and Wildlife

Minnesota Department of Natural Resources
Investigational Report 477, 1999

Influence of Walleye Fingerling Production on Wetland Communities[1]

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Abstract. The effects of walleye fingerling production on macroinvertebrates, zooplankton, fish, plant communities, and limnological parameters of six prairie wetlands were examined. Following two years of pre-treatment data collection, walleye fry were introduced into three of the six wetlands. Numerous significant changes in the macroinvertebrate communities were observed, but none could be attributed to the presence or absence of walleye. Within pond differences were also common, but were as likely to occur in reference ponds as in treatment ponds. Walleye consumed invertebrates throughout the summer indicating the possibility of competition with waterfowl using these areas for brood rearing or migratory staging. With the exception of one pond which was fishless prior to walleye introduction, there was no evidence that the introduction of walleye reduced macroinvertebrate numbers. Walleye did suppress fathead minnow populations in two of the ponds following a winterkill in 1996-1997.

INTRODUCTION

Nearly five million walleye *Stizostedion vitreum* fingerlings are reared annually in shallow natural wetlands and lakes by the Minnesota Department of Natural Resources, Section of Fisheries (Minnesota DNR 1999) . These wetlands usually do not support permanent fish populations. Walleye fry are stocked in spring, allowed to grow through the summer, and harvested in fall. These fingerlings range in size from 100 to 200 mm total length (TL). The fingerlings are then stocked into permanent fish lakes to supplement walleye populations.

Most of Minnesota's walleye fingerlings are reared in the Northwest Region of the state. Fishery managers harvested 8,250 kg of walleye fingerlings from 17,648 ha of rearing water in this region in 1998 (Minnesota DNR 1999). Private aquaculturists also rear substantial numbers of walleye throughout the region with nearly 18,500 ha of water licensed to private aquaculturists for walleye production in 1998. The Northwest Region of Minnesota is also part of the prairie pothole region, an

[1]This project was funded in part by the Federal Aid in Sport Fish Restoration (Dingell-Johnson) Program. Completion Report, Study 604, D-J Project F-26-R Minnesota

area that produces the vast majority of waterfowl in the Mississippi and Central flyways (Bellrose 1976). Because of this, wildlife managers have expressed concerns over the use of wetlands in the prairie pothole region for fish propagation (Swanson and Nelson 1970; Armstrong and Leafloor 1990; and Bouffard and Hanson 1997). Their concerns center largely on diet overlaps among nesting female waterfowl, the young they produce, and juvenile walleye.

Much of the concern about walleye production is related to demonstrated negative effects of fathead minnows *Pimephales promelas*. Fathead minnows have been shown to negatively influence invertebrate populations and reduce the growth and survival of juvenile mallards *Anas platyrhynchos* (Hanson and Riggs 1995; Cox et al. 1998). There appears to be diet overlap between fathead minnows and waterfowl (Swanson and Duebbert 1989) as well as changes in the invertebrate community due to water quality changes. It is unknown if similar relationships exist between waterfowl and walleye. Walleye in ponds feed primarily on zooplankton until they reach about 60 mm TL when their diet shifts to fish (Walker and Applegate 1976). If zooplankton and fish are not available, however, walleye may feed on macroinvertebrates and potentially compete with waterfowl for food.

The purpose of this study was to examine the effects of walleye introductions on wetland ecosystems. Changes in abundances of fish, macroinvertebrates, zooplankton, and aquatic macrophytes were measured and limnological characteristics were defined in three wetlands stocked with walleye fry and in three wetlands which were not stocked.

STUDY AREA

Six Type V wetlands (Stewart and Kantrud 1971) located in Ottertail, Grant, and Pope counties of west central Minnesota were included in this study (Table 1). Wetlands used in fish propogation are commonly referred to as ponds, therefore wetlands used in this study will be referred to as ponds throughout the rest of this report. Ponds were selected based upon accessibility and on similarities in water chemistry and fish populations. Ponds with black bullhead *Ameirus melas* were excluded. Ponds ranged in size from 7.3 to 12.1 ha and all were located within the boundaries of United States Fish and Wildlife Service Waterfowl Production Areas. Initially, five of the six ponds contained fathead minnows and four had populations of brook sticklebacks *Culaea inconstans*. One pond, Ranch, had no fish prior to the study.

METHODS

Three ponds, Morrison, Ranch and Stammer were identified as treatment ponds and were used for walleye rearing during 1996 and 1997. Walleye fry were stocked into each treatment pond at rates approximating 12,300 to 19,900 per ha (Table 1). Three remaining

Table 1. Physical characteristics of the study ponds and the fry stocking regimes used in treatment ponds.

Pond	Size (ha)	Maximum depth (m)	Status	Number of fry stocked 1996	Number of fry stocked 1997
Hagstrom	9.7	2.1	Reference	none	none
Steinlick	12.1	4.3	Reference	none	none
Mavis	12.0	2.3	Reference	none	none
Morrison	11.3	2.1	Treatment	150,000	225,000
Ranch	7.3	3.3	Treatment	90,000	90,000
Stammer	8.7	2.3	Treatment	150,000	150,000

ponds, Hagstrom, Steinlick, and Mavis, were used as reference ponds and were not stocked. Walleye were left in the treatment ponds until late September when they were harvested with trap nets. A subsample of approximately 100 walleye were measured and weighed from each pond at the time of harvest. The total numbers and kilograms of walleye harvested from each pond were recorded.

Fish, macroinvertebrates, zooplankton, aquatic plants, and water chemistry characteristics were monitored from 1994 through 1997 in all six ponds. Fish were sampled monthly from May through September, 1995 - 1997, by randomly placing six minnow traps (154 x 154 x 459 mm, 13 mm mesh) in each pond. In 1994, fish were sampled only during June and August.

Zooplankton were sampled during May, June, July, and September, 1994-1997, using a 120 mm diameter Wisconsin plankton net with153 μm mesh. Five vertical tows were collected at random locations from each pond, samples pooled, and preserved in 4% formalin. The depth of each tow was recorded. During processing, sample volumes were adjusted to 100 ml to concentrate zooplankton and 5 subsamples were collected using a 5 ml Hansen-Stempel pipet. Zooplankton from each 5 ml aliquot were categorized as either cladocerans or copepods, and counted using a counting wheel and a variable power dissecting microscope. Means of these subsamples were used to calculate organisms per liter of water sampled.

Activity traps (Swanson 1979) were used to sample near-shore invertebrates. During 1994, six activity traps were placed randomly in each pond and samples were collected monthly from May through September. From 1995 through 1997, nine activity traps were placed randomly in each pond during May, June, July, and September. Each trap was in place for approximately 24 hours. Upon retrieval, the trap contents were poured through a U.S. Standard Number 30 sieve. The remaining macroinvertebrates were preserved in reagent alcohol and later identified and counted.

An Eckman dredge was used to collect benthic invertebrates. Six random samples were collected from each pond monthly from May through September 1994. Beginning in 1995 and continuing through 1997, six random samples were collected from each pond in May, July, and September. Dredge samples were consolidated in the field, preserved in reagent alcohol, and stored until processing. During processing, samples were poured through a U.S. Standard Number 30 sieve, rinsed thoroughly, placed in an evaporation dish and dyed with rose bengal stain. Samples were stained for a minimum of 4 hours. Samples were then rinsed thoroughly and macroinvertebrates were picked from the remaining detritus. Macroinvertebrates were identified to family, counted, and preserved in reagent alcohol. Macroinvertebrate densities were converted to numbers per m^2 by multiplying by appropriate factors.

Analyses of benthic and free-swimming invertebrates collected with activity traps were limited to the numbers of total dipterans, chironomids, *Chaoborus* spp., corixids, and amphipods. This was due to their reported importance as waterfowl food (Swanson 1984; Afton et al. 1991; Krapu and Reinecke 1992), and because these taxa composed the majority of the organisms we collected in benthic and activity trap samples. Analyses were conducted separately using annual means (combined catches from all months 1994-1997), and July means for macroinvertebrate groups. Macroinvertebrate data collected in July were analyzed separately since any walleye predation on these groups would likely be most evident several months after walleye were stocked, but prior to a shift in walleye diet to mainly fish (Walker and Applegate 1976).

Relative abundances and species composition of the plant communities in each pond were sampled using a weighted rake at six randomly selected stations (Jessen and Lound 1969). The six stations were then standardized for the duration of the study. Macrophytes were sampled twice annually, in June and again in late August or early September.

Water samples were collected annually in July from each pond. Two-L samples were collected from a randomly selected location near the center of each pond, placed in a cooler, and sent to the Minnesota Department of Agriculture Chemistry Lab for analysis. Analyses included pH, total alkalinity (ppm); total dissolved solids (ppm), conductivity (μMHO/cm), total phosphorus (ppm), and chlorophyll-*a* (ppb). Secchi disc measurements were taken monthly to measure water clarity.

Walleye fingerlings were sampled to monitor growth and obtain diet information. Several techniques were used to collect juvenile walleye including beach seines and backpack electrofishing gear, however, a purse seine (31m x 2.4m, 6.3 mm mesh) was the most successful means of collecting fingerlings. All walleye were preserved in formalin and returned to the laboratory where they were measured and weighed. Stomachs were removed and contents identified to lowest possible taxon (usually family), counted, and weighed. Small sample sizes made it necessary to pool diet samples for the entire growing season for each pond in both years.

Catches per unit effort of fathead minnows and brook sticklebacks were log transformed [ln (n+1)]. Analyses were conducted separately using annual means and July means to match analyses of macroinvertebrates. A generalized linear analysis of variance model and a Kruskal-Wallis one-way analysis of variance were used to test for differences in means of July catches of macroinvertebrates and fathead minnows among years within each pond. Tukey's HSD test was used to test for significant differences among means. Statistical significance is reported at the $P<0.05$ level for all tests.

RESULTS

Walleye Production

Walleye fry introductions in 1996 failed in Morrison and Stammer ponds as no fish were harvested from either pond, but 84 kg of fingerlings were harvested from Ranch Pond. In 1997, fry introductions were successful in all treatment ponds. Fingerlings weighing a total of 109 kg were harvested from Ranch Pond and 204 kg were harvested from Stammer Pond. All indications were that the walleye fry stocking in Morrison Pond was likely as successful as the Stammer Pond introduction because fingerlings were collected prior to September for diet analysis. In early September, a large concentration (>500) of double-crested cormorants *Phalacrocroax auritus* was observed feeding on the pond. One double-crested cormorant was observed regurgitating 9 walleye fingerlings. Trap netting two weeks later failed to catch any fingerlings. Based on the size of fingerlings and consumption rates of double-crested cormorants (Madenjian and Gabrey 1995), we estimated the loss of fingerling production to be more than 200 kg.

Walleye diets and growth

Walleye in the study ponds appeared to feed on a variety of invertebrates. During 1996, 56 walleye were collected from Ranch Pond in July but none thereafter. Most of the fish collected in early July had consumed *Chaoborus* spp., while copepods comprised the majority of diet items of those fish collected later in the month (Table 2). Walleye growth was excellent in Ranch Pond during 1996. The mean length of walleye collected on 1 July was 62 mm (n=29), increased to 91 mm (n=19) by 30 July, and to 143 mm (n=40) when the fish were harvested on 16 October 1996.

During 1997, 10 walleye averaging 35 mm TL were collected from Ranch Pond in early June. Their diet consisted mainly of cladocerans. Only 7 walleye were collected the rest of the season, 3 in August and 4 in September. The diets of these fish consisted mainly of amphipods. Growth of walleye was much slower in 1997. Fish collected on 20 August averaged 94 mm TL. One month later they had grown to 107 mm and by mid-October were only 114 mm TL, considerably smaller than those harvested in 1996.

Table 2. Percent composition by number of food items in the stomachs of walleye collected during 1996 and 1997 from Morrison, Ranch, and Stammer ponds.

	Morrison 1997 (n=37)	Ranch 1996 (n=56)	Ranch 1997 (n=17)	Stammer 1996 (n=6)	Stammer 1997 (n=48)
Number with food	30	54	17	1	16
Diet Item					
Crustacea					
Cladocera	67	5	53		55
Copepoda		30			
Amphipoda	3	2	29		40
Insecta					
Ephemeroptera					
Ceanidae		trace			
Odonata					
Zygoptera	14	trace	6		
Hemiptera					
Corixidae	10				
Diptera					
Chironomidae	6	2	12		5
Culucidae					
Chaoborinae		61			
Fish					
Ameirus melas				100	
Total	100	100	100	100	100

During 1997, 23 walleye were collected from Morrison Pond in June, 6 in July, and 8 in August. June diets consisted entirely of cladocerans while August diets consisted mainly of zygoptera, chironomids, and amphipods (Table 2). None of the fish collected in July contained food. The mean lengths of these fish were 43 mm in June, 71 mm in July, and 103 mm in August.

Only six walleye were collected from Stammer Pond in July of 1996. One fish contained food, a young of the year black bullhead. These fish averaged 127 mm TL. During 1997, walleye were collected in June and on two occasions in September. Only one of the fish caught in June contained food. It had eaten several small chironomids (Table 2). The majority of the stomachs from walleye collected in early September were empty but most of those collected later in the month had consumed either amphipods or cladocerans. The 16 fish collected in June averaged 53 mm TL; the 32 fish collected in September averaged 130 mm.

Fish Communities

Fathead minnows were common in five of the six ponds throughout the study. Brook sticklebacks were also present in five of the six ponds but in limited numbers. New species invaded three of the ponds. White suckers *Catostomus commersoni* were caught from Steinlick in 1995, black bullheads from Morrison and Stammer in 1995, and Iowa darters *Etheostoma exile* from Stammer in 1994. No white suckers or Iowa darters were found in 1996 or 1997. Black bullheads were present in Morrison Pond in 1996, but anoxic conditions during the winter of 1996-97 eliminated the population. Although anoxic conditions also occurred in Stammer Pond the same year, black bullheads re-invaded Stammer Pond in 1997.

Fathead minnow populations fluctuated widely (Table 3). In Hagstrom and Steinlick ponds, few fathead minnows were caught in 1994 but populations in both ponds increased in 1995 and 1996. Fathead minnow catches differed among years in Steinlick Pond with the peak catch in 1996. July catches were significantly lower in 1997. By 1997, fathead minnow numbers had declined to levels approximating those in 1994. In Hagstrom Pond, the annual and July mean catches of fathead minnows were significantly higher in 1996 than in other years. Fathead minnow catches fluctuated seasonally in Mavis Pond. July catches were significantly higher in 1997 than in other years, but annual means did not differ significantly among years.

Catches of fathead minnows were very similar in Morrison and Stammer ponds, the two treatment ponds with existing fathead minnow populations. Significantly lower annual and July catches were observed in 1997 in both ponds (Table 4). With the exception of introduced walleye, no fish were found in Ranch Pond throughout the study.

Free-swimming macroinvertebrates

There were few notable patterns in the macroinvertebrate community samples collected with activity traps. One exception was total activity trap catches (all organisms and all taxa), which were highest in all six ponds in 1994 (Table 5). Catches in Steinlick, Mavis, Morrison and Ranch ponds were significantly higher in 1994 than in the succeeding three years of the study.

Although numerous significant differences among the means were found for each pond for the July samples, they were not consistent among ponds. With the exception of Stammer Pond, July catches of all macroinvertebrates were highest in all ponds in 1994. Similarly, amphipod numbers were highest in 1994, then decreased in succeeding years in five of the six ponds (Table 6). Although significant changes in the annual and July catches of corixids were observed, no clear patterns were noted (Table 7). No differences between ponds with walleye and those without were observed, nor were any differences observed for the treatment ponds that could be attributed to the introduction of walleye.

Benthic macroinvertebrates

There were no differences in benthic macroinvertebrate populations between treatment and reference ponds or within treatment ponds that could be attributed to the introduction of walleye. No significant differences in the annual mean catches of dipterans from Hagstrom or Steinlick ponds were found (Table 8). In the remaining four ponds, a general pattern of high abundance of dipterans in 1994 was followed by significant declines in 1995 or 1996 and by increases in 1997 (Table 8). In general, chironomid numbers followed this same pattern (Table 9). On the other hand, significant increases in *Chaoborus* spp. abundances were noted over the course of the study in all six ponds (Table 10).

Zooplankton

Declining abundances of cladocerans were observed in five of the six ponds (Table 11). The most dramatic declines were noted in Steinlick, Morrison, and Stammer ponds.

Table 3. Catch rates (number per trap) of fathead minnows from 1994 through 1997 in those ponds not stocked with walleye fry. Significant differences in mean annual catches as determined by Tukey's pairwise comparison are noted with letters a, b, and c. Significant differences in July means as determined by Kruskal-Wallace tests are noted with letters x, y, and z.

	Hagstrom					Steinlick					Mavis				
	94	95	96	97		94	95	96	97		94	95	96	97	
Mean	1^{b}	18^{b}	98^{a}	11^{b}	P=0.0043	26^{b}	73^{ab}	179^{a}	49^{b}	P=0.0034	98	60	87	180	P=0.1626
May		13	231	50			215	788	235			145	366	166	
June	1	5	135	1		2	63	60	8		26	99	5	2	
July		0^{x}	29^{y}	<1^{z}	P=0.0001		47^{x}	39^{x}	<1^{y}	P<0.0001		32^{x}	13^{x}	148^{y}	P=0.0051
August	1	12	38	0		48	3	1	<1		170	13	20	126	
September		59	58	1			36	9	<1			13	33	456	

* Fish populations were sampled only during June and August in 1994.

Table 4. Catch rates (number per trap) of fathead minnows from 1994 through 1997 in ponds stocked with walleye fry in 1996 and 1997*. Significant differences in mean annual catches as determined by Tukey's pairwise comparison are noted with letters a, b, and c. Significant differences in July means as determined by Kruskal-Wallace tests are noted with letters x, y, and z.

	Morrison					Ranch				Stammer				
	94	95	96	97		94	95	96	97	94	95	96	97	
Mean	120^{a}	217^{a}	96^{a}	3^{b}	P<0.0001					235^{a}	239^{a}	179^{a}	9^{b}	P<0.0001
May		284	12	<1			0	0	0		516	227	4	
June	3	405	123	14		0	0	0	0	25	244	135	39	
July		178^{x}	155^{x}	<1^{y}	P=0.0001		0	0	0		123^{x}	239^{x}	3^{y}	P=0.0001
August	233	164	65	0		0	0	0	0	440	117	62	0	
September		53	126	0			0	0	0		193	234	0	

* Fish populations were sampled only during June and August in 1994.

Table 5. Annual and monthly mean relative abundances of all organisms collected with activity traps, 1994-1997. Significant differences in mean annual catches as determined by Tukey's pairwise comparison are noted with letters a, b, and c. Significant differences in July means as determined by Kruskal-Wallace tests are noted with letters x, y, and z.

Pond	Mean	May	June	July	September
Hagstrom					
1994	40^{a}	60	63	33^{x}	4
1995	9^{b}	19	9	8^{y}	1
1996	10^{b}	6	8	9^{xy}	17
1997	14^{ab}	6	14	22^{x}	12
	P=0.0001			P=0.0047	
Steinlick					
1994	25^{a}	15	34	38^{x}	14
1995	5^{b}	9	8	3^{y}	1
1996	15^{b}	13	5	36^{x}	8
1997	8^{b}	6	8	13^{xy}	7
	P<0.0001			P=0.0071	
Mavis					
1994	84^{a}	187	95	39	16
1995	20^{b}	31	34	12	3
1996	22^{b}	13	34	33	8
1997	21^{b}	8	34	34	6
	P=0.0001			P=0.0921	
Morrison					
1994	139^{a}	396	81	68^{x}	12
1995	13^{b}	14	11	24^{xy}	4
1996	19^{b}	13	24	31^{xy}	6
1997	70^{b}	5	8	63^{x}	202
	P=0.0006			P=0.0525	
Ranch					
1994	550^{a}	1404	252	153^{x}	391
1995	37^{b}	40	47	41^{y}	19
1996	93^{b}	264	46	22^{yz}	38
1997	14^{c}	19	24	4^{z}	8
	P<0.0001			P<0.0001	
Stammer					
1994	80^{a}	195	105	17^{xy}	2
1995	42^{ab}	55	78	33^{xy}	2
1996	13^{b}	13	28	7^{y}	5
1997	25^{ab}	8	42	38^{x}	10
	P=0.0327			P=0.0113	

Table 6. Monthly mean relative abundance of amphipods (*Gammarus spp.* and *Hyallela azteca* combined) collected with activity traps, 1994-1997. Significant differences in mean annual catches as determined by Tukey's pairwise comparison are noted with letters a, b, and c. Significant differences in July means as determined by Kruskal-Wallace tests are noted with letters x, y, and z.

Pond		Mean	May	June	July	September
Hagstrom						
	1994	20^{a}	36	37	4	2
	1995	3^{bc}	6	3	1	1
	1996	2^{c}	3	2	<1	<1
	1997	3^{ab}	3	6	<1	6
		P<0.0001			P=0.0925	
Steinlick						
	1994	16^{a}	5	28	21^{x}	10
	1995	3^{b}	4	4	2^{y}	1
	1996	2^{b}	2	<1	1^{y}	3
	1997	2^{b}	2	2	2^{y}	<1
		P<0.0001			P<0.0001	
Mavis						
	1994	56^{a}	110	65	35^{x}	14
	1995	10^{b}	26	10	2^{y}	1
	1996	8^{b}	8	15	6^{xy}	4
	1997	11^{b}	5	11	26^{x}	2
		P<0.0001			P=0.0014	
Morrison						
	1994	126^{a}	360	71	65^{x}	9
	1995	6^{b}	11	4	7^{y}	3
	1996	2^{b}	4	1	5^{y}	<1
	1997	71^{c}	4	4	53^{x}	221
		P<0.0001			P<0.0001	
Ranch						
	1994	563^{a}	1465	207	133^{x}	448
	1995	21^{b}	28	15	25^{y}	13
	1996	81^{b}	246	30	14^{y}	35
	1997	4^{c}	3	<1	9^{y}	3
		P<0.0001			P<0.0001	
Stammer						
	1994	41^{a}	79	71	13	2
	1995	5^{b}	6	4	7	1
	1996	7^{ab}	13	11	2	1
	1997	6^{b}	2	15	7	<1
		P=0.0021			P=0.3438	

Table 7. Annual and monthly mean relative abundance of corixids collected with activity traps, 1994-1997. Significant differences in mean annual catches as determined by Tukey's pairwise comparison are noted with letters a, b, and c. Significant differences in July means as determined by Kruskal-Wallace tests are noted with letters x, y, and z.

Pond		Mean	May	June	July	September
Hagstrom						
	1994	3^{ab}	<1	4	7	<1
	1995	2^{b}	1	2	5	1
	1996	5^{a}	1	3	3	13
	1997	13^{a}	1	37	8	4
		P=0.0013			P=0.0464	
Steinlick						
	1994	3^{ab}	2	1	3^{x}	0
	1995	1^{b}	2	3	1^{x}	<1
	1996	11^{a}	10	2	26^{y}	4
	1997	3^{ab}	1	3	6^{xy}	3
		P=0.0052			P=0.0086	
Mavis						
	1994	$<1^{b}$	1	1	0^{x}	0
	1995	7^{b}	1	17	8^{xy}	0
	1996	13^{a}	5	18	25^{y}	3
	1997	6^{b}	1	8	12^{y}	3
		P=0.0001			P=0.0053	
Morrison						
	1994	6^{b}	22	<1	1^{xy}	<1
	1995	4^{ab}	<1	5	10^{x}	1
	1996	13^{a}	4	21	23^{x}	5
	1997	2^{b}	<1	1	4^{y}	2
		P=0.0002			P=0.0096	
Ranch						
	1994	2^{ab}	<1	4	2^{xy}	0
	1995	4^{a}	7	5	3^{x}	0
	1996	2^{ab}	<1	3	3^{x}	<1
	1997	$<1^{b}$	<1	<1	$<1^{y}$	<1
		P=0.0004			P=0.0069	
Stammer						
	1994	6^{b}	14	9	$<1^{y}$	0
	1995	12^{a}	20	10	17^{x}	<1
	1996	5^{a}	3	12	3^{xy}	3
	1997	6^{ab}	3	13	4^{xy}	3
		P=0.0101			P=0.0062	

Table 8. Density of dipterans (number/m^2) collected in benthic samples from 1994 through 1997. Significant differences in mean annual catches as determined by Tukey's pairwise comparison are noted with letters a, b, and c. Significant differences in July means as determined by Kruskal-Wallace tests are noted with letters x, y, and z.

Pond		Mean	May	July	September
Hagstrom					
	1994	762	675	422	1190
	1995	913	1604	689	445
	1996	1838	4101	866	547
	1997	824	711	502	1258
		P=0.4602		P=0.6266	
Steinlick					
	1994	2315	1519	2168^{x}	3257
	1995	1981	2622	711^{y}	2609
	1996	1549	1910	1733^{xy}	1004
	1997	2095	3199	1199^{xy}	1888
		P=0.5032		P=0.0362	
Mavis					
	1994	3060^{a}	3066	4532^{x}	1582
	1995	799^{b}	266	799^{y}	1333
	1996	1111^{ab}	1155	1289^{y}	889
	1997	3346^{a}	7506	1288^{y}	1244
		P=0.0045		P=0.0087	
Morrison					
	1994	5229^{a}	2933	9376^{x}	3377
	1995	696^{b}	667	222^{z}	1199
	1996	1718^{ab}	933	2889^{y}	1333
	1997	3481^{b}	8088	2267^{y}	89
		P=0.0007		P<0.0001	
Ranch					
	1994	2444^{a}	1022	5866^{x}	444
	1995	6458^{a}	711	17998^{x}	667
	1996	459^{b}	889	356^{z}	133
	1997	1266^{a}	578	1975^{y}	1244
		P=0.0015		P<0.0001	
Stammer					
	1994	2577^{a}	4398	1733	1599
	1995	1096^{b}	2222	800	267
	1996	1555^{ab}	3289	622	755
	1997	2177^{ab}	1866	2578	2089
		P=0.0191		P=0.2199	

Table 9. Number of chironomids (number/m^2) collected in benthic samples from 1994 through 1997. Significant differences in mean annual catches as determined by Tukey's pairwise comparison are noted with letters a, b, and c. Significant differences in July means as determined by Kruskal-Wallace tests are noted with letters x, y, and z.

Pond		Mean	May	July	September
Hagstrom					
	1994	717	633	380	1139
	1995	745	1434	506	295
	1996	1561	3545	717	422
	1997	264	542	125	125
		P=0.0193		*P*=0.0763	
Steinlick					
	1994	1997^{a}	1519	1899^{x}	2574
	1995	1069^{ab}	1012	422^{y}	1772
	1996	464^{b}	844	0^{z}	548
	1997	597^{b}	890	695^{y}	208
		P<0.0001		*P*<0.0001	
Mavis					
	1994	2475^{a}	2827	3503^{x}	1097
	1995	126^{b}	169	126^{y}	84
	1996	126^{b}	211	84^{y}	84
	1997	1305^{ab}	3514	284^{y}	118
		P=0.0001		*P*<0.0001	
Morrison					
	1994	4585^{a}	2658	8397^{x}	2700
	1995	239^{b}	465	168^{z}	84
	1996	478^{b}	549	548^{y}	338
	1997	2400^{b}	6269	869^{y}	63
		P<0.0001		*P*<0.0001	
Ranch					
	1994	2199^{a}	886	5502^{x}	211
	1995	5879^{a}	548	16711^{x}	380
	1996	281^{b}	633	211^{z}	0
	1997	609^{ab}	292	1466^{y}	70
		P=0.0051		*P*<0.0001	
Stammer					
	1994	1997^{a}	3882	1435	675
	1995	1195^{b}	1477	295	84
	1996	323^{b}	464	295	211
	1997	1635^{a}	1515	1800	1591
		P<0.0001		*P*=0.0551	

Table 10. Density of *Chaoborus* spp. (number/m^2) collected in benthic samples from 1994 through 1997. Significant differences in mean annual catches as determined by Tukey's pairwise comparison are noted with letters a, b, and c. Significant differences in July means as determined by Kruskal-Wallace tests are noted with letters x, y, and z.

Pond		Mean	May	July	September
Hagstrom					
	1994	17^{c}	0	45^{y}	7
	1995	107^{bc}	67	142^{y}	111
	1996	348^{ab}	760	116^{y}	169
	1997	497^{a}	0	368^{x}	1124
		$P<0.0001$		$P=0.0080$	
Steinlick					
	1994	189^{b}	0	133^{y}	435
	1995	739^{ab}	1369	169^{y}	680
	1996	1114^{a}	1155	1577^{x}	609
	1997	1310^{a}	1999	333^{y}	1599
		$P=0.0011$		$P=0.0011$	
Mavis					
	1994	184^{b}	0	297	257
	1995	493^{ab}	58	578	844
	1996	918^{a}	844	1155	755
	1997	948^{a}	889	889	1067
		$P<0.0001$		$P=0.2094$	
Morrison					
	1994	199^{a}	18	178^{yz}	401
	1995	391^{a}	21	42^{z}	1111
	1996	1214^{a}	355	2266^{x}	1022
	1997	780^{a}	1113	1200^{xy}	27
		$P=0.0350$		$P=0.0049$	
Ranch					
	1994	23^{b}	18	9^{y}	43
	1995	223^{a}	89	356^{x}	222
	1996	112^{ab}	178	27^{y}	133
	1997	430^{a}	178	44^{y}	1067
		$P=0.0004$		$P=0.0001$	
Stammer					
	1994	118^{b}	89	133^{y}	133
	1995	355^{a}	489	444^{x}	133
	1996	1200^{a}	2711	356^{xy}	533
	1997	415^{a}	267	622^{x}	356
		$P=0.0001$		$P=0.0317$	

Table 11. Cladocera and copepods per liter of water sampled from study ponds.

Pond		May		June		July		Sept.	
		Cladocera	Copepods	Cladocera	Copepods	Cladocera	Copepods	Cladocera	Copepods
Hagstrom									
	1994	15	51	0	21	0	86	0	99
	1995	140	383	4	45	0	20	0	38
	1996	4	40	0	72	0	32	0	0
	1997	10	33	0	48	0	24	0	32
Steinlick									
	1994	165	247	390	69	3	3	0	19
	1995	11	56	0	21	0	38	0	12
	1996	0	80	0	56	0	45	7	21
	1997	2	17	4	21	9	11	6	3
Mavis									
	1994	25	293	9	20	39	999	5	276
	1995	3	34	100	86	10	593	0	95
	1996	0	46	0	278	0	328	0	910
	1997	0	25	103	217	0	342	4	179
Morrison									
	1994	262	205	124	178	37	36	4	448
	1995	20	427	229	526	10	267	8	187
	1996	60	323	6	1974	10	756	0	29
	1997	46	24	229	60	496	178	1047	186
Ranch									
	1994	257	520	175	1430	215	1001	32	162
	1995	98	177	171	45	71	282	34	204
	1996	255	435	149	135	161	126	11	866
	1997	180	206	206	168	469	279	161	212
Stammer									
	1994	160	1356	76	420	16	275	0	70
	1995	43	185	11	403	0	80	0	97
	1996	0	69	7	119	51	488	0	533
	1997	11	27	19	376	0	259	15	425

Unlike Steinlick and Stammer ponds, however, cladocerans in Morrison Pond recovered to the highest abundances observed for any pond during late summer 1997. Cladoceran numbers were low in Hagstrom Pond throughout the study. In Ranch Pond, cladoceran numbers appeared to be stable to slightly increasing over the four years of the study. Copepod numbers declined in Steinlick Pond and fluctuated greatly in Stammer Pond.

Aquatic Plants and Limnological Characteristics

With the exception of Ranch Pond, few changes in the aquatic plant communities were observed. There was a substantial increase in the abundance of clasping-leaf pondweed *Potamogeton richardsonii* in Ranch Pond (Table 12).

Water clarity was generally low in Mavis and Stammer ponds throughout the study and declined in Steinlick Pond (Table 13). A dramatic increase in water clarity was observed in Morrison Pond in 1997.

Substantial increases in total phosphorus levels were noted for all ponds in 1997 (Table 14). With the exception of Chlorophyll-*a* levels in Ranch Pond, which increased in 1997, all other water chemistry variables remained stable over the course of the study.

DISCUSSION

None of the numerous significant changes in macroinvertebrate numbers could clearly be attributed to the presence of walleye, particularly in Morrison and Stammer ponds. Changes in macroinvertebrate communities were as likely to occur in reference ponds as in treatment ponds. The only pattern of abundances that was consistent among ponds was that the highest catches in activity traps and of zooplankton occurred during 1994. This may have been a response to a substantial increase in water levels in 1993 and 1994 that flooded previously dry areas and increased pond productivity. These changes, however, were not observed in the benthic communities.

There was some indication that walleye affected the invertebrate community in Ranch Pond. Significant declines in the pooled mean densities of chironomids and dipterans and total activity trap catches occurred in 1996, after the introduction of walleye. Several of these taxa, particularly the dipterans, were found in the walleye diets, and since the walleye had no fish to prey on, it is reasonable to assume that predation by walleye played some role in the decline of these taxa. However, while catches of amphipods and total activity trap catches did decline significantly following walleye introduction, both had been declining since the inception of the study in 1994. In fact the greatest decline in both groups occurred between 1994 and 1995, both pretreatment years. Furthermore, despite their presence in the walleye diets in 1997, two groups, dipterans and chironomids increased significantly during the second year of treatment. Therefore, while walleye may have contributed to the decline of certain invertebrates, it is questionable whether or not they were the leading cause.

Walleye consumed macroinvertebrates throughout the summer and into the fall in all three treatment ponds. Walleye ate far more zooplankton, primarily cladocerans, throughout the year than expected. In other studies of walleye diets in similar habitats, they readily consumed zooplankton until they reached lengths of 62 mm; shortly after walleye reach that length they usually became piscivorus (Walker and Applegate 1976; Dobie 1958). Many of the diet items eaten by walleye are also preferred by nesting and migrating waterfowl (Swanson 1984; Afton et al. 1991), therefore, the potential for competition for food between waterfowl and walleye may exist.

Despite the fact that walleye consumed zooplankton further into the growing season and at larger sizes than expected, corresponding declines in the zooplankton populations were not observed. Since invertebrate populations did not decline significantly in Morrison or Stammer ponds in the presence of walleye, it is unlikely that walleye reduced the availabil-

Table 12. Mean aquatic plant density ratings of select species from the study ponds collected 1994 through 1997. Density ratings are based upon methodology developed by Jesson and Lound (1962) where: 5=dense; 4=heavy; 3=moderate; 2=scattered; 1=sparse.

	Hagstrom		Steinlick		Mavis		Morrison		Ranch		Stammer	
	June	Aug.	June	Aug.	June	Aug.	June	Aug.	June	Aug.	June	Aug.
Myriophyllum sibiricum												
1994	3.7	2.8	3.2	3.8	0.7	1.2	0.2	2	0.3	0.3	0.3	1.3
1995	1.3	3.5	0	0.3	0.7	2.3	2.5	2.5	0.7	0.2	0	0
1996	0.2	0.5	2	0	2.5	4	3.3	2	1.7	0.7	0.2	0.5
1997	0	2.3	0.7	0	1.7	2.8	0.2	2.3	0	0	0.7	4
Potamogeton pectinatus												
1994	3	1.8	0	0	0	0	0	1.7	0	0.5	0	1.2
1995	3.2	4.5	0	0	0.2	1.2	0.7	1.2	0	0.7	0	0
1996	3.5	2.3	0	0	0	0.3	1.3	1	0	0.2	1.2	2.3
1997	1.7	3.2	0	0.2	0.7	0.7	1.3	0.5	0	0.2	0	0.2
Potamogeton richardsonii												
1994					0.5	0.8	0.3	0	0.5	1		
1995					1.3	1	0.5	0.2	2.3	3.2		
1996					2.2	1	1	0.5	3.7	3.5		
1997					1.3	1	0.7	0.2	4.2	3.5		
Utricularia vulgaris												
1994	2.3	2.2	1.8	1.5								
1995	3.3	0.2	1.2	0.7								
1996	0.2	0.8	1.2	1								
1997	0.2	0.8	1.2	1								
Chara spp.												
1994	0	0	0	0.3								
1995	0.2	1	0	0								
1996	2	1.8	0	0.2								
1997	4	1.2	0	0								

Table 13. Secchi disk measurements (in centimeters) from all study ponds, May through September, 1994-1997.

Pond		May	June	July	August	September
Hagstrom						
	1994	152*	152*	129	173*	173*
	1995	182*	170*	170*	167*	102
	1996	148	138	127	131	113
	1997	180*	162*	142	138	132
Steinlick						
	1994	305*	244*	244*	170	104
	1995	134	162	135	152	132
	1996	147	95	145	115	78
	1997	113	141	112	121	94
Mavis						
	1994	31	56	71	53	43
	1995	71	79	71	69	73
	1996	69	93	72	62	63
	1997	181	247	123	64	68
Morrison						
	1994	104	147	79	46	64
	1995	91	44	38	41	55
	1996	294	49	37	29	34
	1997	287*	247*	227	187	115
Ranch						
	1994	74	114	66	46	26
	1995	107	216*	139	214*	136
	1996	274*	175	177	146	49
	1997	145	223	97	140	134
Stammer						
	1994	213*	213*	89	66	109
	1995	94	91	63	48	59
	1996	81	102	49	38	40
	1997	156	81	44	37	52

(*Indicates that secchi disk could be seen at the bottom of the pond)

Table 14. Comparison of select water chemistry variables collected in July, 1994 through 1997.

	Total Dissolved Solids (ppm)	Chlorophyll-a (ppb)	Total Phosphorus (ppm)	Total Alkalinity (ppm)
Hagstrom				
1994	448	9.6	0.024	258
1995	498	17.1	0.044	237
1996	476	9.0	0.021	241
1997	358	22.2	0.272	217
Stienlick				
1994	508	11.6	0.043	285
1995	364	19.3	0.044	298
1996	590	24.9	0.046	286
1997	388	31.1	0.094	252
Mavis				
1994	474	44.3	0.099	375
1995	416	44.0	0.049	329
1996	528	27.7	0.050	337
1997	492	19.6	0.422	315
Morrison				
1994	572	41.7	0.120	437
1995	512	79.8	0.086	396
1996	432	57.6	0.130	397
1997	620	51.5	0.224	402
Ranch				
1994	440	27.2	0.080	314
1995	402	23.0	0.033	290
1996	492	12.8	0.024	282
1997	428	62.8	0.136	280
Stammer				
1994	804	53.1	0.130	464
1995	680	62.5	0.048	400
1996	508	15.2	0.063	343
1997	564	51.2	0.575	305

ity of food for waterfowl anymore than existing populations of fathead minnows.

Walleye are known to be quite piscivorus in ponds when they reach larger sizes (Dobie 1958; Walker and Applegate 1976). Therefore the lack of fish, especially fathead minnows and brook sticklebacks, in thewalleye diets was somewhat surprising. The anoxic conditions in winter 1996-1997 may have reduced fathead minnow populations in Morrison and Stammer ponds to levels so low that walleye could rarely prey on them. Despite the absence of fish in diet samples, we question whether walleye could have maintained the growth observed on a diet strictly of invertebrates. Walleye may have suppressed fathead minnow populations in Morrison and Stammer ponds, but small sample sizes and the lack of fathead minnows in walleye stomachs examined make this speculative.

Fish and wildlife managers have suggested that wetlands already degraded by high density fathead minnow populations could be rehabilitated through biomanipulation. Walker and Applegate (1976) found that introduction of walleye into a wetland created trophic cascades that improved water clarity and invertebrate abundance. Biomanipulation has been used successfully in larger water bodies as well (Shapiro et al. 1975; Benndorf 1987).

Fathead minnow numbers declined significantly in Morrison Pond followed by immediate increases in water clarity and cladoceran numbers. It is unknown if walleye predation could have created a similar response without the anoxic conditions that reduced fathead minnow numbers prior to the introduction of walleye. Declines in the fathead minnow population in Stammer Pond did not result in improved water clarity, invertebrate numbers, or macrophyte abundance. Successful biomanipulation and the resulting trophic cascades may be as dependent on catastrophic events such as winter anoxia as they are on predatory influences.

The initial introduction of walleye fry into Morrison and Stammer ponds in 1996 failed. Fathead minnow abundances were high and minnows may have preyed on the fry or competed with them for food. Managers attempting to control fathead minnows and improve water quality through biomanipulation should consider stocking a life stage larger than fry to successfully introduce walleye into situations with high densities of fathead minnows. This management strategy was used successfully in a South Dakota pond (Walker and Applegate 1976).

MANAGEMENT RECOMMENDATIONS

Despite diet overlaps with waterfowl, walleye had no apparent effect on invertebrate populations in ponds with existing fathead minnow populations. Therefore, it is recommended that fishery managers continue with current fish culture practices in wetland ponds.

The only indication that walleye could significantly affect invertebrates needed by waterfowl occurred in Ranch Pond. Since Ranch Pond did not contain fathead minnows or brook sticklebacks at anytime during the study, the lack of an alternative preferred prey resource for walleye may have been a factor in the decline of invertebrate abundances. The use of fishless ponds for walleye production should be discouraged. Furthermore, because of the direct link between high fathead minnow abundance, low invertebrate numbers, and the inferior conditions that result for waterfowl production (Hanson and Riggs 1995), fishery managers should refrain from stocking fathead minnows in a wetland to make it more suitable for walleye production.

Stocking walleye fry into ponds to control or reduce high density fathead minnow populations may be unsuccessful unless accompanied by some catastrophic event such as winterkill. We suggest that wetland managers wishing to reduce endemic fish populations manage these system to maximize the likelihood of winterkill. Introduction of walleye fry following winterkill may slow the recovery of fathead minnow populations.

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ACKNOWLEDGMENTS

We would like to thank student interns Kris Klimstra and Jody Nemcek for field and laboratory assistance. Jeanne Reed also assisted with various aspects of field work over the course of the study. Mark Hanson (Minnesota D.N.R. - Wildlife Research) and Richard Lillie (Wisconsin D.N.R.) provided helpful comments and advice on the initial study design. We also thank Rod Pierce, Tim Cross, Cindy Tomcko, and Mark Hanson for their helpful reviews of the manuscript. Staff from the Morris and Fergus Falls Wetland Management District Offices (U.S.F.W.S.) provided assistance in locating study sites.

SH 328 .I58 no. 477
Reed, Jeffrey R.
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