THE STATE OF LAKE SUPERIOR IN 2000

Great Lakes Fishery Commission

SPECIAL PUBLICATION 07-02

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July 2007

THE STATE OF LAKE SUPERIOR IN 2000

Edited by

Mark P. Ebener
Inter-Tribal Fisheries and Assessment Program
Chippewa Ottawa Resource Authority
179 W. Three Mile Road
Saul Ste. Marie, Michigan, U.S.A., 49783

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EXECUTIVE SUMMARY

Fish-community objectives (FCOs) to guide lakewide, coordinated fishery management in Lake Superior were developed in 1990 in accordance with A Joint Strategic Plan for Management of Great Lakes Fisheries (Great Lakes Fishery Commission 1997). These objectives were revised in 2000 to recognize the importance of habitat to achieving objectives, and because knowledge regarding fish community function and structure has changed in the last decade (Horns et al. 2003). The revised objectives emphasize natural reproduction of indigenous species, habitat protection, and prevention of additional introductions. Specific objectives have been developed for habitat, prey species, lake trout, lake whitefish, walleye, lake sturgeon, brook trout, non-native salmonines (Pacific salmon, rainbow trout, Atlantic salmon, brown trout, and splake), sea lamprey, nuisance species, and species diversity. This state-of–the-lake report describes the status of the fish community inhabiting Lake Superior in 2000, changes since the last state-of-the-lake report in 1992 (Hansen et al. 1994), and progress toward achieving FCOs, which are provided at the start of the applicable chapters.

Habitat

Management agencies have been successful in promoting more natural and stable flows in dammed tributaries that will benefit walleye, brook trout, and lake sturgeon. There are large-scale watershed restoration projects in both Minnesota and Michigan that will produce positive benefits for brook trout. A GIS-based, fish-habitat inventory for Lake Superior's Minnesota waters and a GIS-based atlas of Lake Superior fish spawning and nursery areas have also been created. Reductions in effluents from Canadian pulp mills have been achieved. However, much is yet to be done regarding habitat inventory and restoration, and many impediments to achieving the habitat objectives remain. Habitat restoration at Remedial Action Plan sites has not progressed beyond identifying and implementing remedial measures. Fish consumption advisories remain common as toxic chemicals continue to enter the lake and accumulate in fish. Other threats to habitat include water-level regulation, development of wetlands, changes in forest composition, forest fragmentation, pollution and nutrient loading, sedimentation, exotic species, dams and water diversion, and dredging. Proper watershed-level land-use planning and reducing the impacts of shoreline development are critical for restoring habitat productivity within the Lake Superior basin.

Prey Species

The prey-fish community of Lake Superior is diverse and made up of primarily self-sustaining populations of indigenous species. Lake herring, deepwater ciscoes, trout-perch, sculpins, and ninespine stickleback are the common indigenous prey fish, and rainbow smelt is the major non-indigenous species. Total prey-fish abundance in 2000 was less than during the early 1990s. Lake herring abundance has recovered from the low levels observed in the 1970s, but it is still less than prior to 1960.

Lake Trout

Lean lake trout populations are self-sustaining throughout most of the lake, and wild lean lake trout may be more abundant in some areas than during the pre-1940s reference period. Lean lake trout populations in far eastern (Whitefish Bay) and far western (Superior/Duluth) waters remain dependent upon stocking of hatchery fish to sustain the populations at levels necessary to sustain sport and commercial fisheries. Siscowets are the most-abundant form of lake trout in the lake, particularly in offshore waters, and humpers, another deepwater form, are abundant in eastern offshore waters and around Isle Royale. Fishery yields of all three forms of lake trout have declined since the early 1990s, and average total mortality rates during 1993-2000 in all jurisdictions were below the 45% target maximum and below the rates prior to 1993. Sea lamprey marking rates on adult lean lake trout throughout Lake Superior remain well above the target rate of 5 marks$\cdot$$100^{-1}$ fish, particularly in northwestern Ontario waters.

Lake Whitefish

Lake whitefish are self-sustaining and made up of many spatially segregated stocks in Lake Superior. The commercial fishery produced high and relatively stable yields that averaged 1.5 million kg during the 1990s. These high yields were due primarily to increased abundance driven by increased recruitment during the early and mid-1990s. Mortality rates for most whitefish stocks are less than maximum values established for sustainable yield.

Walleye

Walleye are known to spawn in 32 areas around Lake Superior, primarily in tributaries. Walleye abundance in Ontario and Michigan is presently less than observed prior to the 1960s, but abundance is near pre-1960 levels in Chequamegon Bay and the St. Louis River estuary. Rehabilitation has been slowed by degraded habitat, fish-passage problems at hydroelectric barriers, highly variable recruitment, and slow growth. Sampling spawning areas in remote locations is difficult and biological data are limited for many locations. Stocking supports many walleye populations.

Lake Sturgeon

At least 21 tributaries historically supported spawning lake sturgeon populations, but lake sturgeon currently reproduce in only 11. Abundances of adults in these 11 spawning populations are lower than that deemed necessary to meet rehabilitation goals. Stocking fingerling lake sturgeon has succeeded in increasing the abundance of juveniles in the St. Louis River and western Lake Superior. Hydropower facilities are situated on 12 of the 21 historical spawning tributaries, and they are the single greatest impediment to rehabilitation in those streams. These hydropower facilities limit lake sturgeon production by dewatering spawning and rearing habitat, altering flow and temperature regimes, and blocking access to historical spawning areas.

Brook Trout

Brook trout populations had formerly been associated with at least 118 tributaries on Lake Superior. Stocking has been undertaken using strains from Lake Nipigon (Ontario) and Isle Royale (Michigan), and more-stringent angling regulations have been enacted. Achieving the FCO for brook trout will require actions to restore tributary habitat, regulate control of harvest, and stocking of appropriate genotypes.

Pacific Salmon, Rainbow Trout, and Brown Trout

Chinook, coho, and pink salmon, rainbow trout (steelhead), and brown trout all reproduce successfully in Lake Superior tributaries, and these naturalized fish vastly outnumber hatchery fish in the recreational harvest. Non-indigenous salmonines play a relatively minor role in the Lake Superior fish community, but they potentially could impact populations of indigenous species, especially brook trout.

Sea Lamprey

Sea lamprey abundance has increased since 1994, and sea lampreys continue to be a significant source of mortality on adult lean lake trout—well above the 5% target maximum rate. Actions taken to reduce sea lamprey abundance in the 1990s have not yet achieved this FCO.

Nuisance Species

At least 39 non-indigenous aquatic species have entered Lake Superior since 1970, and three have entered since 1992 (round goby, tubenose goby, Asian clam) via ship ballast discharge, bio-fouling of ship hulls, or water diversions and canals constructed to facilitate shipping. The majority of the introductions originated in the St. Louis River estuary because it includes the Duluth-Superior Harbor, which is the busiest port on the lake. Governments in the U.S. and Canada have taken very little action to prevent future introductions of non-indigenous species or the spread of those already here.

Species Diversity

Many of the species originally present in Lake Superior still inhabit the lake and contribute to the diversity and stability of the fish community. Recreational species such as yellow perch, northern pike, and smallmouth bass are common members of the fish community in embayments and tributaries and their estuaries. Indigenous cyprinids such as emerald, spottail, and sand shiners are the most-abundant species in very shallow water of embayments and tributaries. No direct action has been taken to achieve this FCO.

Major Recommendations

- Quantify the relationship between habitat supply and fish production
- The Lake Superior Committee should collaborate with township and municipal planning boards, nongovernmental organizations, and state and federal agencies to rehabilitate and protect tributary habitat
- Reduce sea lamprey abundance and balance their control actions with rehabilitation of indigenous species such as lake sturgeon, brook trout, and walleye
- Stop the entry of invasive species into Lake Superior by establishing effective ballast-water management
- Develop models of important populations such as lake trout, lake whitefish, lake herring, and siscowets to track stock dynamics and estimate sustainable harvests levels and predator consumption
- Develop lakewide databases for harvest and effort, survey, diet, and fish stocking

HISTORY

Mark P. Ebener
Inter-Tribal Fisheries and Assessment Program
Chippewa Ottawa Resource Authority
179 W. Three Mile Road
Sault Ste. Marie, Michigan, U.S.A., 49783

The Great Lakes Fishery Commission (GLFC) in cooperation with federal, state, provincial, and tribal natural resource agencies, adopted a Joint Strategic Plan for Management of Great Lakes Fisheries (Joint Plan) in 1981 as an explicit statement for cooperative fishery management on the Great Lakes (Great Lakes Fishery Commission 1992). The Joint Plan was revised in 1994 as a practical tool for coordinating inter-jurisdictional management efforts to provide mutual benefits and protection of the Great Lakes (Great Lakes Fishery Commission 1997). Lake committees and attendant technical committees are the action arms for implementing the Joint Plan and for developing plans for managing the aquatic resources of each Great Lake. The Lake Superior Committee (LSC) is composed of one fishery manager each from the Michigan Department of Natural Resources, Ontario Ministry of Natural Resources, Minnesota Department of Natural Resources, Wisconsin Department of Natural Resources, Chippewa Ottawa Resource Authority, and Great Lakes Indian Fish and Wildlife Commission.

In response to the Joint Plan, fish-community objectives (FCOs) were developed and adopted by the LSC to define objectives for the structure of the fish community and to develop means for measuring progress toward their achievement (Busiahn 1990). These FCOs were recently revised by the LSC (Horns et al. 2003) to reflect a better understanding of the past and present dynamics and processes currently working to shape the structure of the fish community and to make the FCOs fit within the broader context of environmental objectives for Lake Superior being developed under the Binational Program (Lake Superior Work Group 2000, 2002). The Binational Program represents a partnership of federal, state, provincial, and tribal/First Nation governments cooperatively working to protect, restore, and maintain the Lake Superior ecosystem (Lake Superior Work Group 2002).

The overall goal on Lake Superior is:

> …*to rehabilitate and maintain a diverse, healthy, and self-regulating fish community, dominated by indigenous species and supporting sustainable fisheries.*

This goal recognizes the desirability of a mainly self-sustaining, indigenous fish community.

The Lake Superior Technical Committee (LSTC) is charged by the LSC with producing a state of the lake report every five years to assess how effectively FCOs are being met and to identify new and emerging issues that will affect future management.

A description of the limnology, geography, and characteristics of Lake Superior and its drainage basin can be found in Lawrie and Rahrer (1973), Assel (1985), Hansen and Schorfhaar (1994), Schneider et al. (1993), Edsall and Gannon (1993), Horns et al. (2003), and Bronte et al. (2003). The figure below (Fig. 1) shows fishery-management units on Lake Superior as well as important sites and tributaries identified in this report. The common and scientific names of fishes mentioned in this report are presented in Table 1.

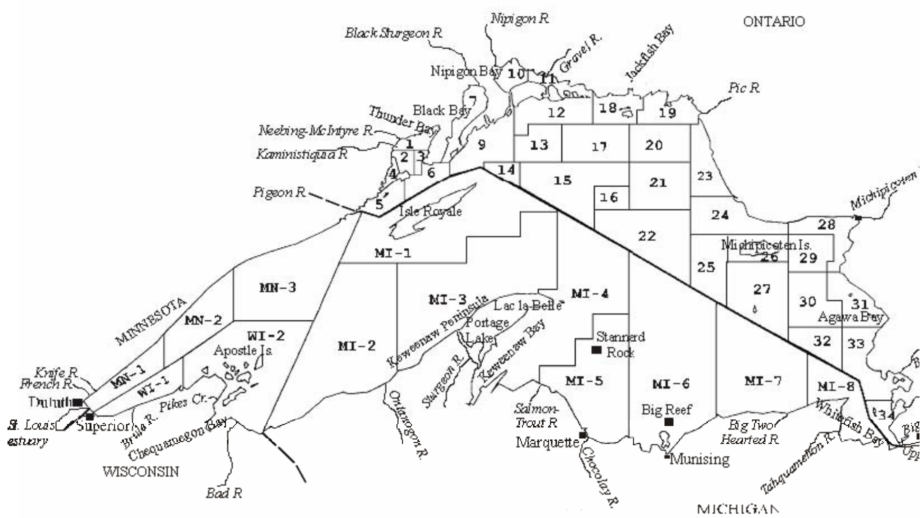

Fig. 1. The Lake Superior basin, including management units and major tributaries (italics).

Table 1. Common and scientific names of fishes referenced in this report.

Common Name	Scientific Name
Indigenous species:	
bloater	*Coregonus hoyi*
bluntnose minnow	*Pimephales notatus*
bowfin	*Amia calva*
brook trout	*Salvelinus fontinalis*
brown bullhead	*Ameiurus nebulosus*
burbot	*Lota lota*
creek chub	*Semotilus atromaculatus*
deepwater ciscoes	*Coregonus* spp.
deepwater sculpin	*Myoxocephalus thompsoni*
emerald shiner	*Notropis atherinoides*
finescale dace	*Phoxinus neogaeus*
lake chub	*Couesius plumbeus*
lake herring	*Coregonus artedi*
lake sturgeon	*Acipenser fulvescens*
lake trout (lean, siscowet, humper)	*Salvelinus namaycush*
lake whitefish	*Coregonus clupeaformis*
logperch	*Percina caprodes*
mimic shiner	*Notropis volucellus*
minnows	Cyprinidae
ninespine stickleback	*Pungitius pungitius*
northern pike	*Esox lucius*
rock bass	*Ambloplites rupestris*
sand shiner	*Notropis stramineus*
sculpins	*Cottus* spp.
slimy sculpin	*Cottus cognatus*
smallmouth bass	*Micropterus dolomieu*
splake	*Salvelinus fontinalis* x *S. namaycus*
spoonhead sculpin	*Cottus ricei*
spottail shiner	*Notropis hudsonius*

Table 1, continued

Common Name	Scientific Name
Indigenous species, continued	
suckers	*Catostomus* spp.
trout-perch	*Percopis omiscomaycus*
walleye	*Zander vitreus*
white sucker	*Catostomus commersoni*
yellow perch	*Perca flavescens*
Non-indigenous species:	
alewife	*Alosa pseudoharengus*
Atlantic salmon	*Salmo salar*
brown trout	*Salmo trutta*
Chinook salmon	*Oncorhynchus tshawytscha*
coho salmon	*Oncorhynchus kisutch*
fourspine stickleback	*Apeltes quadracus*
Pacific salmon	*Oncorhynchus* spp.
pink salmon	*Oncorhynchus gorbuscha*
rainbow smelt	*Osmerus mordax*
rainbow/steelhead trout	*Oncorhynchus mykiss*
round goby	*Neogobius melanostomus*
ruffe	*Gymnocephalus cernua*
sea lamprey	*Petromyzon marinus*
threespine stickleback	*Gasterosteus aculeatus*
tubenose goby	*Proterorhinus marmoratus*
white bass	*Morone chrysops*
white perch	*Morone americana*

Fishery management agencies responded to the collapse of many indigenous fish populations in Lake Superior during 1950-1970 (Lawrie and Rahrer 1972, 1973) by severely restricting or closing commercial fisheries, implementing sea lamprey control, and stocking hatchery-reared fish (Pycha and King 1975; Lawrie and Rahrer 1972; Hansen et al. 1995). These activities produced substantial increases in abundance and a recovery of lake trout, lake whitefish (hereafter, whitefish as a common name), and lake herring (MacCallum and Selgeby 1984; Hansen et al. 1995; Schreiner and Schram 1997; Wilberg et al. 2003; Bronte et al. 2003). Favorable environmental conditions for reproduction and reductions in abundance of rainbow smelt also helped stimulate the recovery of these species (Ebener 1997; Bronte et al. 2003; Cox and Kitchell 2004). Stocking of hatchery-reared salmonines peaked in 1991 at 6.3 million fish, of which 51% were lake trout (Table 2). By 2000, the number of salmonines stocked into the lake had declined to 3.6 million, and only 27% were lake trout, while 40% were Chinook salmon. The reductions in stocking of lake trout were made in response to declines in survival of hatchery-reared fish (Hansen et al. 1994) and to the recognition that lake trout reproduction was sufficient to support many of the populations (Schreiner and Schram 1997). Chinook salmon, coho salmon, brown trout, and rainbow trout were established in Lake Superior as a consequence of stocking (Peck 1970; Peck et al. 1994; Peck et al. 1999).

Table 2. Number of trout and salmon stocked into Lake Superior, 1970-2000.

Year	Lake Trout	Chinook Salmon	Coho Salmon	Rainbow Trout
1970	2,356,500	175,000	618,000	226,825
1971	1,790,941	252,000	590,000	238,600
1972	1,848,772	371,000	297,000	309,700
1973	1,653,570	395,000	135,000	290,000
1974	2,062,042	523,000	529,000	155,400
1975	1,304,117	253,000	275,000	313,100
1976	2,355,234	492,000	400,000	398,400
1977	1,730,274	253,000	627,000	276,700
1978	2,321,646	478,000	141,000	209,000
1979	1,833,711	498,000	192,000	90,000
1980	1,801,749	627,000	350,000	167,800
1981	2,247,267	728,088	288,000	331,539
1982	2,415,477	882,000	238,090	204,373
1983	2,578,827	853,000	325,000	399,665
1984	2,122,911	787,124	275,000	401,588
1985	3,045,804	726,081	302,000	309,772
1986	3,092,059	1,309,536	288,000	379,109
1987	2,755,391	1,193,272	275,000	348,660
1988	2,339,711	1,381,074	334,000	304,630
1989	1,853,562	1,731,253	325,000	341,104
1990	2,229,448	1,842,702	220,000	366,620
1991	3,224,151	1,854,672	195,000	541,628
1992	2,594,524	1,295,119	178,000	414,570
1993	1,955,553	1,215,904	180,000	326,244
1994	2,029,049	1,442,066	—	283,095
1995	1,810,532	1,439,604	—	376,416
1996	1,433,044	1,103,347	87,700	453,116
1997	1,222,673	1,262,935	—	225,294
1998	1,171,600	627,037	—	313,449
1999	1,038,940	1,078,197	—	353,162
2000	959,139	1,432,738	—	388,650

Table 2, continued.

Year	Brown Trout	Splake	Brook Trout	Atlantic Salmon
1970	104,300	—	17,000	—
1971	140,000	13,200	25,000	—
1972	144,500	—	58,000	20,000
1973	147,000	4,000		20,000
1974	137,300	166,600	21,000	—
1975	276,500	15,000	150,000	—
1976	112,500	18,000	17,600	9,100
1977	117,900	—	150,000	200
1978	111,100	82,000	275,500	37,000
1979	114,400	194,000	237,900	—
1980	93,000	173,000	139,000	—
1981	75,200	218,000	80,000	—
1982	102,475	176,000	43,000	17,952
1983	98,500	178,000	59,000	11,025
1984	108,794	168,400	50,000	11,866
1985	98,070	224,600	119,400	25,154
1986	108,770	247,000	168,624	42,041
1987	156,296	356,000	148,694	72,258
1988	159,510	210,000	58,710	49,093
1989	247,855	—	—	31,251
1990	264,545	170,400	41,957	105,747
1991	230,683	122,495	89,185	51,666
1992	285,900	297,960	67,440	97,529
1993	336,140	215,188	64,306	—
1994	149,600	277,200	49,630	—
1995	237,400	148,300	81,700	—
1996	197,635	320,733	69,000	—
1997	203,055	129,425	21,700	—
1998	169,955	210,076	10,000	—
1999	99,831	232,230	103,540	—
2000	349,887	196,204	262,777	—

Although fishery-management agencies on Lake Superior should be proud of the recovery of some indigenous species, invasive species and human activities continue to suppress abundance of other indigenous fish species (Lake Superior Work Group 2002; Horns et al. 2003). Abundances of brook trout, walleye, and lake sturgeon are impacted, although these species are still common members of the fish community in portions of the lake. The LSC and LSTC have produced recovery plans for these three species (Auer 2003; Newman et al. 2003; Hoff 2003), and many agencies are now actively engaged in either monitoring their status or have embarked on a process of promoting their recovery. New invasive species have established in Lake Superior since the last state of the lake report (Bronte et al. 2003), and their effects on indigenous species remain unknown. Some progress has been made to reduce critical pollutants—mercury, PCBs, and DDT. Wetland restoration and maintaining run-of-the-river flows on important tributaries such as the Nipigon River have improved habitat conditions. Degradation of physical habitat continues, and the level of toxaphene, one of nine designated critical pollutants, has not declined in lake trout (Lake Superior Work Group 2002).

FISHERIES OF LAKE SUPERIOR

Mark P. Ebener[1] and Donald R. Schreiner[2]

State-, provincial-, or tribal-licensed commercial, sport, and subsistence fisheries operate throughout Lake Superior (Brown et al. 1999). Subsistence fisheries are licensed exclusively by U.S. Native American and Canadian First Nation tribes. The commercial fishery harvests primarily whitefish and lesser amounts of lake herring, lake trout, siscowet, and deepwater ciscoes, whereas sport fisheries target lake trout, Pacific salmon, brown trout, splake, and, to a lesser extent, walleye and yellow perch. The subsistence fishery harvests both sport and commercial species.

Commercial Fishery

The size and scope of the commercial fishery on Lake Superior declined substantially since the mid-1900s. There were thousands of commercial fishing operations during the 1930-1950s (Legault et al. 1978; Brown et al. 1999) that harvested lake herring, lake trout, whitefish, deepwater ciscoes, and walleyes. Today, there are fewer than 200 commercial operations. The changes that occurred in the commercial fishery have been due to changes in fish stocks, attrition in the fishery, changes in state and provincial regulatory policies, and development of a treaty-reserved fishery. The collapse of lake trout populations and low abundance of lake herring and whitefish during the 1950s to early 1970s (Lawrie and Rahrer 1972; Pycha and King 1975; Jensen 1976; Selgeby 1982) left few fishing opportunities for commercial fishermen. During the 1960s, fisheries agencies in Minnesota, Wisconsin, and Michigan changed their management emphasis from commercial fisheries to recreational (sport) fisheries (Brown et al. 1999; Kocik and Jones 1999).

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[1]Chippewa Ottawa Resource Authority, 179 W. Three Mile Road, Sault Ste. Marie, Michigan, U.S.A. 49783

[2]Minnesota Department of Natural Resources, Lake Superior Area Fisheries Program, 5351 North Shore Drive, Duluth, Minnesota, U.S.A. 55804

This change in management philosophy and lakewide efforts to restore lake trout populations and protect depressed populations of other important fish species resulted in restrictions on the number of commercial licenses, areas where the fisheries could operate, and the types and amount of fishing gear (Christie 1978; Brege and Kevern 1978; MacCallum and Selgeby 1984; Brown et al. 1999). The implementation of treaty-reserved fishing rights by U.S. Native American and Canadian First Nation tribes has allocated a larger share of the fishery resource to tribal fisheries since 1970 (Doherty 1990; Brown et al. 1999).

Characteristics of the Fishery

Most commercial fishing operations on Lake Superior use either gillnets or trapnets. Some trawling was done for lake herring in Ontario waters. The trapnet fishery targets whitefish, operates only in U.S. waters, and typically begins in late April or May and extends through early November, depending upon political jurisdiction. Fisheries using gillnets occur throughout the lake, target several species, and operate most months. Most gillnets are fished from boats, but some are fished through ice cover. Gillnet fishing with small boats (<8 m long) makes up the largest proportion of total commercial fishing operations and accounts for roughly 75-80% of the Native American commercial fishery. These small boats fish nearshore waters, and nets are often lifted by hand. Other gillnet operations employ larger boats, usually a Great Lakes gillnetter design (11-17-m long), that use a mechanical lifter to pull nets from the lake. The gillnet ice fishery is least common because suitable ice cover occurs only in Whitefish Bay, lower Keweenaw Bay, Thunder Bay, the Apostle Islands, Black Bay, and Nipigon Bay. This fishery usually operates within several kilometers of the main shoreline, and snowmobiles are used to transport fishermen, gear, and fish on and off the ice.

Changes in Fishing Efficiency and Catchability

A gillnet mesh conversion from multifilament nylon to monofilament nylon and a switch to deeper nets increased catchability of whitefish. Collins (1979) reported that monofilament gillnets, introduced in the 1960s, were about 1.8 times more efficient than multifilament for catching whitefish. Depth of monofilament gillnets increased from 28-36 meshes deep to 50 meshes deep, and, after 1978, this deeper net became the standard gear, especially for the large-boat gillnet fishery. The 50-mesh-deep gillnet was 1.7 times more efficient for capturing whitefish (Collins 1987). Further increases in catchability resulted from reductions in the diameter of the mesh filament and the introduction of the 75-mesh-deep gillnet (Brown et al. 1999).

Fishing Effort

Gillnet effort on Lake Superior during the last few decades has declined over different time spans for fisheries employing different mesh sizes. Small-mesh (64- to 89-mm stretched mesh) effort has declined since 1976 (Fig. 2) due primarily to decreased abundance of lake herring and deepwater ciscoes. Large-mesh (114 mm and larger stretched mesh) effort targeted at whitefish and lake trout increased in every jurisdiction from the late 1970s to 1990 but has declined every year since 1990. The number of trapnet lifts increased annually from 1973 to 1981 primarily because of increased effort in Whitefish Bay and the Apostle Islands but has fluctuated without trend since 1981. Native American trapnet fisheries made up only 2% of the total lakewide trapnet effort in 1982, but the proportion increased to 41% by 1997.

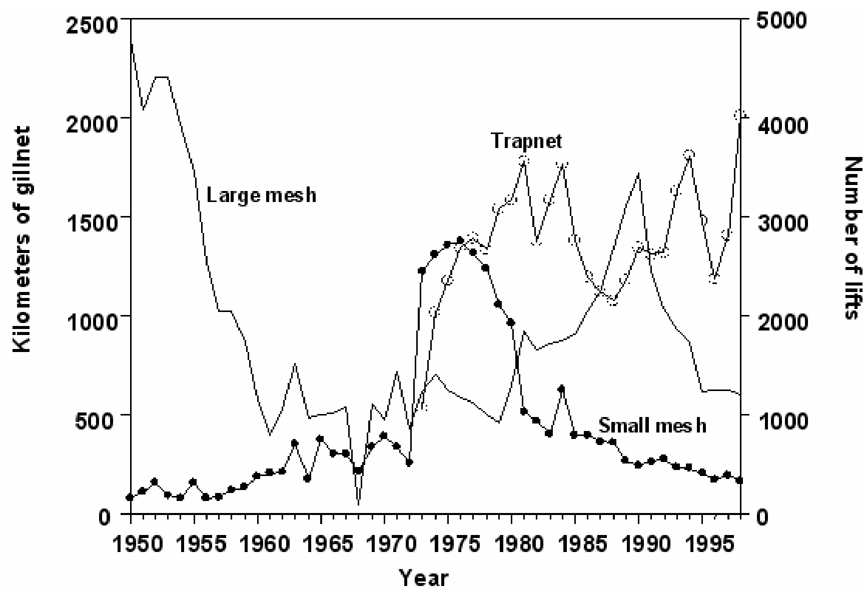

Fig. 2. Large-mesh and small-mesh gillnet effort (km) and trapnet effort (lifts) in the commercial fishery (all species) on Lake Superior during 1950-1998. Large- and small-mesh gillnet effort during 1950-1973 for only the Ontario commercial fishery.

Commercial Yield

The annual commercial yield from all fisheries peaked at 11.5 million kg in 1941 (Lawrie and Rahrer 1972), declined to 2.0 million kg in 1999, and averaged about 2.2 million kg annually since 1980 (Baldwin et al. 2002). Whitefish have been the primary fish targeted by the commercial fishery since the late 1970s (Fig. 3) mainly because their abundance has increased and harvest of other fishes has been severely restricted. Whitefish made up only 2% of the total lakewide commercial harvest in 1960, but their contribution increased to 57% in 1990. In 1999, the commercial harvest consisted of 53% whitefish, 31% lake herring, 8% lake trout, 2% deepwater ciscoes, and 6% other miscellaneous species.

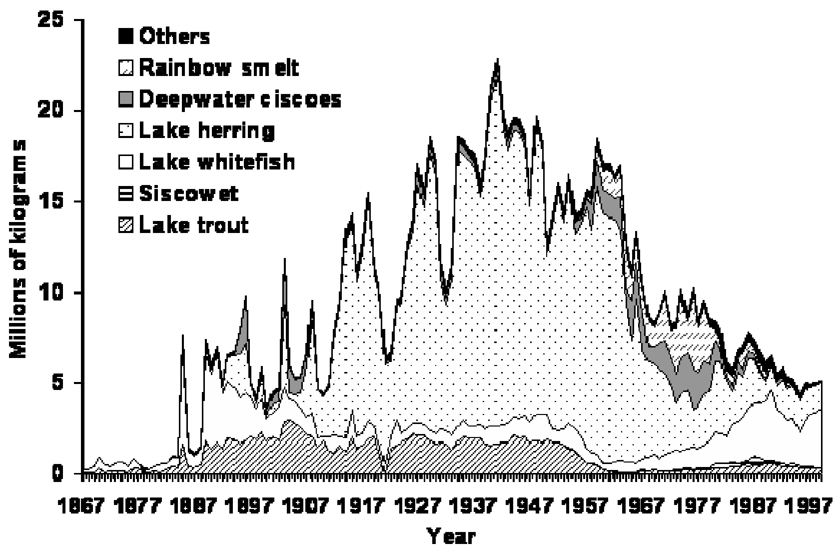

Fig. 3. Commercial harvest (millions of kg) of major fishes from Lake Superior, 1867-1999. There were no harvest data for 1921 (Baldwin et al. 1979, 2002).

Recreational Fishery

Commercial-fishing families started some of the earliest resorts and lodging establishments on Lake Superior that catered to those wealthy enough to travel by boat and rail to the remote areas of the shoreline to fish for lake trout. Fly-fishing on tributaries and from shore was probably the first method employed by many of these early anglers (Roosevelt 1865). Participation in the sport fishery increased during the early 1900s as access to the shoreline became easier. These anglers targeted lake trout and brook trout. Participation in the sport fishery began to decline starting in the 1940s due to declining abundance of lake trout and a preoccupation with World War II. Stream angling still provided a viable fishery, but most anglers were now targeting introduced rainbow and brown trout rather than the native brook and lake trout.

Various factors led to a resurgence of sport fishing during the 1970s, including the recovery of lake trout, introduction of Pacific salmon, increased access, creation of safe harbors and marinas, and the availability of safer and larger sport-fishing vessels. During the same period, anglers discovered and began exploiting walleye, yellow perch, and northern pike in some of the bays and estuaries, including Chequamegon Bay, Black Bay, Keweenaw Bay, St. Louis River estuary, and upper St. Marys River.

Angler Surveys

Little survey information existed to document the catch in the recreational fishery prior to 1970, but state fishery agencies began to conduct more-standardized creel surveys in the early 1970s. Most of these early survey efforts targeted the summer lake fishery. Some harvest information was collected from spring and fall fisheries for spawning trout and salmon in tributaries, but these surveys were conducted intermittently, were not well standardized within agencies, and were not standardized at all among agencies. Ontario conducts angler surveys only in areas of special concern on an intermittent basis, because it is too costly to survey their extensive jurisdiction on a regular basis. Because these infrequent, non-standardized surveys preclude factual comparisons, only information on the summer (May-September) lake fishery in U.S. waters is presented here.

Fishing Effort, Catch, and Catch Rate

In U.S. waters, summer fishing effort increased significantly from the early 1970s to the mid-1980s, then declined, and appears to have now stabilized at about 450,000 angler-hours (Fig. 4). The proportion of fishing effort in each state was 40% in Minnesota, 32% in Wisconsin, and 28% in Michigan. Although no effort estimates are available from Ontario angler surveys, Bence and Smith (1999) reported that approximately 30% of the total Lake Superior fishing effort was derived from Ontario, thus making the lakewide recreational fishing effort about 650,000 angler-hours. These estimates do not include any winter or tributary fisheries, which, in some years, may contribute an additional 25-50% to the overall angling effort.

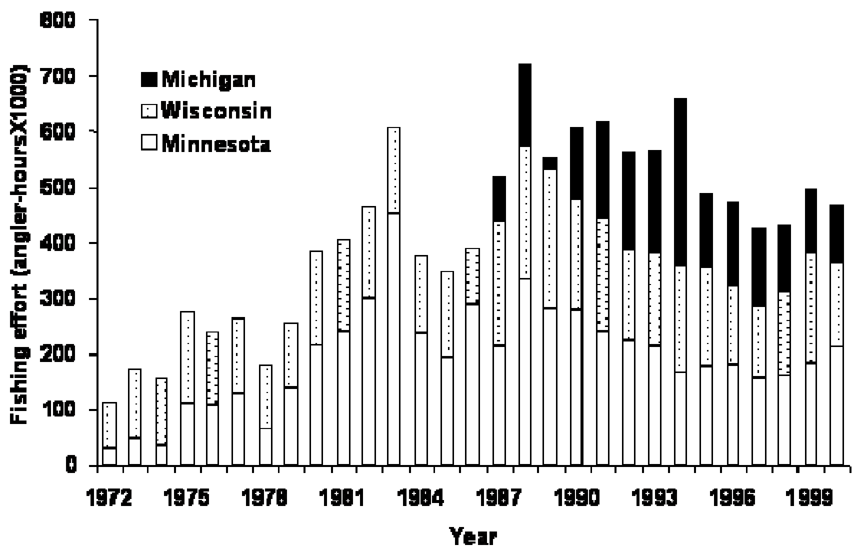

Fig. 4. Summer (May-September) recreational fishing effort (angler-hours) in U.S. waters of Lake Superior, 1972-2000.

Total catch of trout and salmon in the summer lake fishery in U.S. waters averaged 90,000 fish during 1990-2000 (Fig 5). Large catches of coho salmon increased lakewide recreational harvests substantially in 1983, 1988, and 1994. Composition of the catch from U.S. waters during 1990-2000 was dominated by lake trout (72%) with lesser amounts of coho salmon (17%), Chinook salmon (8%), brown trout (2%), and rainbow trout (1%). In Ontario, the catch composition in fishing contests during 1987-1999 was lake trout (78%), Chinook salmon (12%), coho salmon (5%), and rainbow trout (5%).

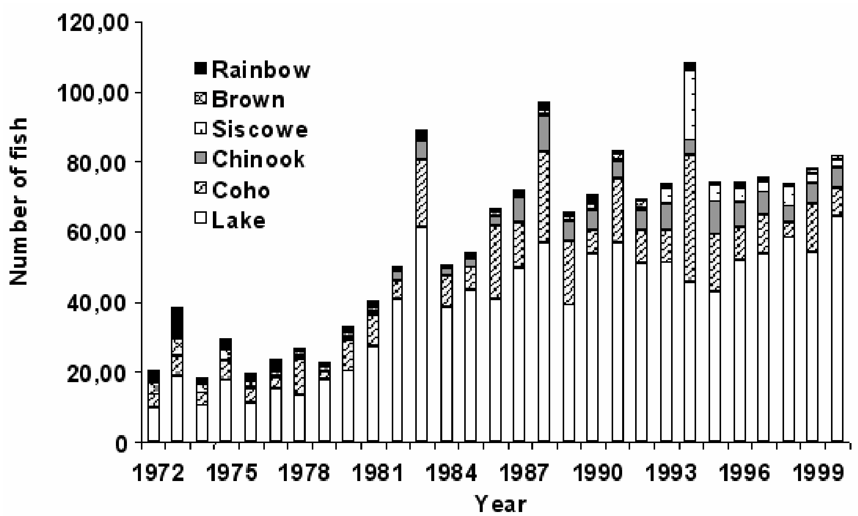

Fig. 5. Summer (May-September) recreational fish harvest of salmonines by agency for U.S. waters of Lake Superior, 1972-2000. State of Michigan harvest not estimated until 1987.

Overall catch rate in the summer lake fishery generally ranged from 0.10 to 0.20 fish·h^{-1} during 1972-2000. The rate increased steadily and averaged 0.15 fish·h^{-1} during 1990-2000, mainly due to an increase in lake trout catch rate. Fishing effort declined during this period of increasing catch rate, which is uncharacteristic for a recreational fishery. The highest catch rate was for lake trout, and the next highest were for coho salmon and Chinook salmon.

Species composition and distribution among jurisdictions in the recreational fishery likely will change little over the next ten years. The contribution of wild fish to angler catch will continue to increase and the contribution of hatchery fish will decrease. If the present trend continues, angling effort will probably be more influenced by economic and social changes than by catch rates.

Subsistence Fishery

Subsistence fishing permits allow the Native American or First Nations' licensee to harvest a limited amount of fish solely for personal consumption with a very limited amount of gear. The gillnet is the primary gear, but dipnets and spears are also used. Subsistence fisheries typically have no closed season or license requirements other than being a member of a tribe or First Nation. There are no more than 200 subsistence fisheries operating annually on Lake Superior. Most subsistence fisheries are required to report their catches to their management authority. Lake trout, whitefish, Pacific salmon, and suckers are the primary targets of the fishery.

PHYSICAL HABITAT

Thomas C. Pratt
Fisheries and Oceans Canada
Great Lakes Laboratory for Fisheries and Aquatic Sciences
1 Canal Drive
Sault Ste. Marie, Ontario, Canada P6A 6W4

> *…achieve no net loss of the productive capacity of habitat supporting Lake Superior fishes; where feasible, restore habitats that have been degraded and have lost their capacity for fish production; reduce contaminants so that fish are safe to eat; and develop comprehensive and detailed inventories of fish habitats.*

The above objectives (Horns et al. 2003) are based on the principle that healthy fish communities require abundant and diverse physical habitats and clean water. The resurgent interest in habitat identification, protection, and remediation, in combination with developing spatial-research tools, provides a window of opportunity for substantial advances in understanding of habitat function and importance basinwide.

Aquatic habitat in Lake Superior is classified into offshore (>80 m deep), nearshore (0-80 m), and embayments (harbors, estuaries, and bays subject to seiches), although tributaries and inland lakes that drain directly into the lake are also important (Lake Superior Work Group 2000; Horns et al. 2003). The physical habitats of the Lake Superior ecosystem are the least impacted by human activity of any of the Great Lakes, but they are variously threatened by stressors that include shoreline development, hydroelectric facilities, barrier dams, industrial effluents, mining waste, loss of wetlands, atmospheric deposition, point-source pollution and nutrient loading, agricultural practices, water-level regulation, dredging, and timber-harvest practices (Lake Superior Work Group 2000, 2002). In general, offshore and nearshore physical habitats are relatively undisturbed, and the associated fish communities are not habitat limited (Lake Superior Work Group 2000). However, many of the stressors identified above continue to affect embayments, tributaries, and connected inland lakes, and fish communities dependent on these habitats are limited by habitat in such a way that the achievement of FCOs is impaired.

In recent years, there have been 100-150 projects implemented and actions taken to restore, protect, and identify habitat in the Lake Superior basin (Lake Superior Work Group 2000, 2002). In U.S. tributaries, hydroelectric facilities have been relicensed requiring run-of-the-river flows as opposed to the previous peaking operations**.** The Nipigon River Management Plan was developed and implemented to produce more-stable flows in Lake Superior's largest tributary and should help increase reproduction of brook trout. Currently, there are several watershed and wetland restoration projects on the Salmon-Trout River in Michigan, which should benefit brook trout and wetland restoration projects at Sugarloaf Cove in Minnesota and at several locations in Michigan's Upper Peninsula. A multilayered GIS platform for Lake Superior's Minnesota waters has been created as well as a GIS-based atlas of Lake Superior fish spawning and nursery areas (E. Chiriboga, Great Lakes Indian Fish and Wildlife Commission, One Point Place, Madison, WI, 53719-2089, unpubl. data). Lastly, there have been reductions in effluents from Canadian pulp mills.

There remain many challenges to protecting and restoring habitat in the Lake Superior basin. None of the seven Remedial Action Plan sites located in Areas of Concern (AOC) in the basin have progressed past Stage 2 (Stage 2 involves identifying and implementing remedial measures required to restore beneficial uses, while Stage 3 documents uses). Habitats most critical to restoring the productive capacity of the Lake Superior ecosystem include wetlands, rivers, streams, and their associated spawning grounds. Recently completed rehabilitation plans for walleye (Hoff 2003), lake sturgeon (Auer 2003), and brook trout (Newman et al. 2003) rely on improved habitat conditions. As such, watershed-level land-use planning aimed at reducing the impacts of shoreline development is also critical for restoring habitat productivity.

Recommendations and Opportunities

There are a number of approaches that management agencies are taking or should be considering to ensure that the no-net-loss objective is achieved. The Lake Huron GIS project, which incorporates a number of valuable data layers (spawning areas, cool-water streams, and barriers to fish passage) into a basinwide GIS platform, would be an excellent model for Lake Superior to follow (http://www.glfc.org/lakehurongis/main.htm). A basinwide GIS tool is necessary if the goal of developing quantifiable habitat-based environmental objectives for Lake Superior is to be met. The Province of Ontario is currently updating watershed management plans throughout the province, providing an opportunity for the modification of flow regimes to minimize impacts on rivers with hydroelectric facilities. The establishment of a Lake Superior National Marine Conservation Area over a wide portion of the Canadian shoreline may also provide additional habitat protection and resources to a little-studied area of the basin. Specific recommendations are to develop long-term monitoring in AOCs and to implement the habitat recommendations in the brook trout (Newman et al. 2003), lake sturgeon (Auer 2003), and walleye rehabilitation plans (Hoff 2003).

CHEMICAL CONTAMINANTS IN FISH

Kory J. Groetsch[1], D. Mike Whittle[2], Melanie Neilson[3], and Patricia J. McCann[4]

…reduce contaminants so that all fish are safe to eat.

Reaching the above goal (Horns et al. 2003) will be a slow process because the lake has a long water retention time, slow sedimentation rate, and a cold temperature. A wide variety of persistent, bioaccumulative, and toxic chemicals have contaminated Lake Superior during the past three decades. Canada and the U.S. currently consider the concentrations of mercury, dioxins/furans, PCBs, toxaphene, and chlordane in fish flesh to be high enough to warrant consumption advisories.

Three additional groups of chemicals may be emerging as chemicals of concern in Lake Superior: polybrominated diphenyl ethers (PBDEs), polybrominated biphenyls (PBB), and polychlorinated naphthalenes (PCNs). PBDEs and PBB have been or are being used as flame retardants. PBBs have been banned in the U.S. since 1974 (Luross et al. 2002); however, PBDEs continue to be widely used in products such as paints, textiles, plastics, and electronics (Luross et al. 2002). PCNs were first synthesized in the mid-1800s and have been used as capacitor dielectrics, cutting oils, engine oil additives, ship insulation, and preservatives in wood, paper, and textiles (Kannan et al. 2000).

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[1]Michigan Department of Community Health, Toxicology and Responses Section, 3423 N. MLK Jr. Blvd., Lansing, Michigan, U.S.A., 48909

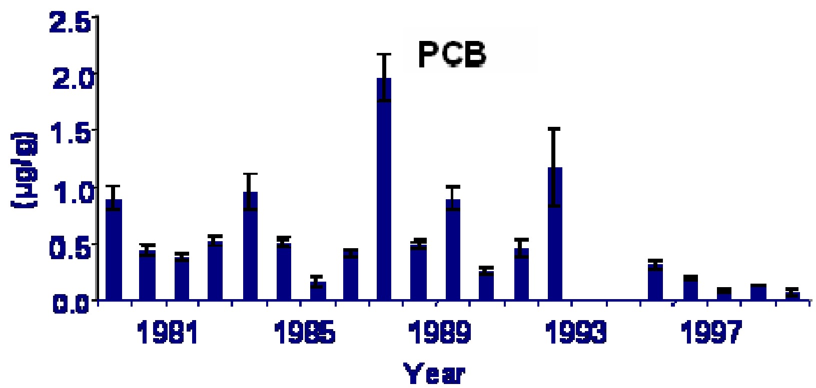
[2]Department of Fisheries and Oceans, Great Lakes Laboratory for Fisheries and Aquatic Sciences, 867 Lakeshore Road, P.O. Box 5050, Burlington, Ontario, Canada, L7R 4A6

[3]Environment Canada, Ecosystem Health Division, Environmental Conservation Branch, Ontario Region, 867 Lakeshore Road, P.O. Box 5050, Burlington, Ontario, Canada, L7R 4A6

[4]Minnesota Department of Health, 121 East Seventh Place, Suite 360, St. Paul, Minnesota, U.S.A., 55101

They are also produced from waste incineration and industrial activity such as chlor-alkali processes (Kannan et al. 2000). PCN production ended in the U.S. in 1980, but PCNs are still found in electrical equipment. PCNs were detected in two whole lean lake trout collected from the south shore of Lake Superior.

Canada's Department of Fisheries and Oceans has compared concentrations of PCBs, DDT, and mercury over time in 4-year-old whole lake trout. Mean PCB concentrations fluctuated between 0.1 and 2.0 $\mu g \cdot g^{-1}$ wet weight during 1980-2000 (Fig. 6) and were consistently lower during 1996-2000 than prior to 1994. Concentrations of DDT and its metabolites DDD and DDE were low during 1996-2000 compared to 1980-1982 (Fig. 6). Mean mercury concentrations were variable but declined during 1981-1999 (Fig. 6). Regulatory actions taken by the Canadian Government in the late 1970's reduced mercury inputs from pulp and paper mills and may be responsible for the decline observed in lake trout.



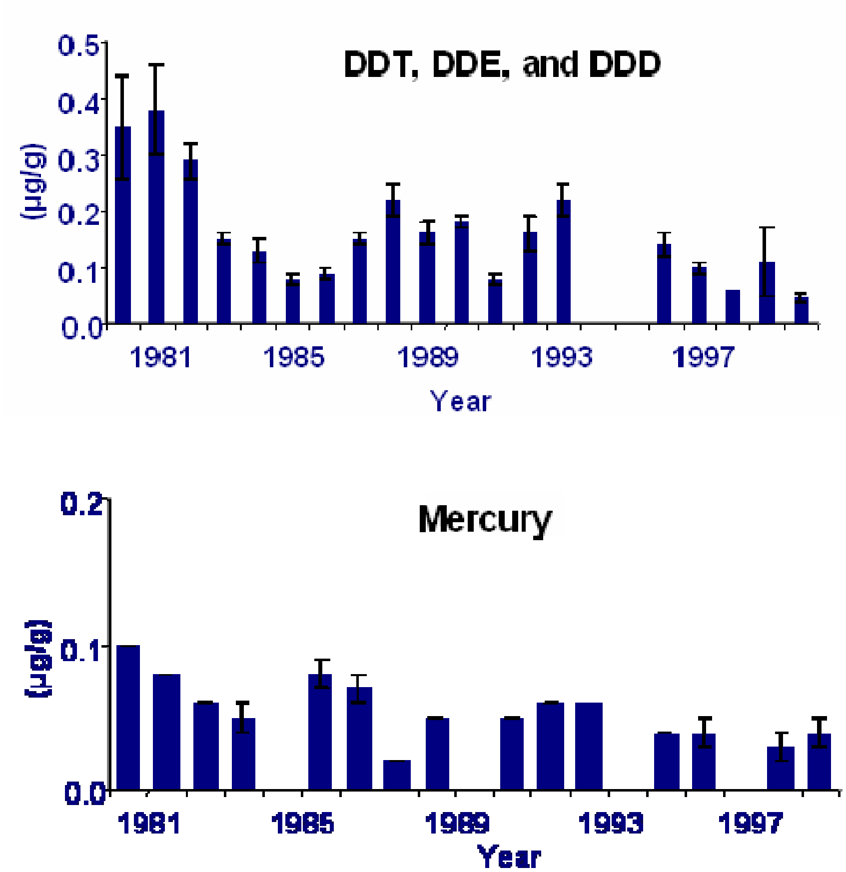


Fig. 6. Mean concentration (wet weight) of total polychlorinated biphenyls (PCB), total DDT, DDE, and DDD, and total mercury in 4-year-old whole lake trout from Lake Superior, 1980-2000. Error bars are ± 1 standard error.

Concentrations of toxaphene in Lake Superior lake trout have not declined like other chemicals (Bronte et al. 2003). Glassmeyer et al. (1997) reported that toxaphene concentrations in whole lake trout in 1982 and 1992 were similar. Whittle et al. (2000) found an apparent decreasing trend of toxaphene concentrations in lake trout during 1980-1986 followed by increasing concentrations during 1986-1998. Toxaphene concentrations in lake trout have declined during the same period in the other Great Lakes.

That toxaphene concentrations have not declined in Lake Superior lake trout has been in part attributed to colder water temperatures and longer water retention times (Swackhamer et al. 1998; James et al. 2001) and potential changes in lake trout prey (Whittle et al. 2000).

Analyzing different species can lead to different conclusions when comparing contaminant concentrations among the Great Lakes. MacEachen et al. (2000) found no significant differences in the concentrations of mercury in whole lake trout from Lakes Erie, Huron, Ontario, and Superior, but they did find that mean concentrations of mercury in adult sea lampreys from Lake Superior (2.28 ppm) differed significantly from those in Lake Huron (0.79 ppm), Lake Erie (0.43 ppm), and Lake Ontario (1.35 ppm). The bioaccumulation of PBT chemicals in sea lamprey is similar to that in lake trout, but sea lamprey accumulate much higher levels of mercury; thus, sea lamprey may be a useful species for monitoring basinwide trends in contaminants.

Most of the persistent chlorinated chemicals that were regulated out of use during the 1970s have declined substantially in Lake Superior fish. Toxaphene concentrations in lake trout are an exception. The source of toxaphene is from outside the Lake Superior basin (James et al. 2001; Swackhamer et al. 1998). The greatest declines for most of the toxic chemicals in fish have already occurred, and the amount of reduction each year is slowing as these chemicals reach equilibrium in Lake Superior and in its aquatic life. Despite these declines, restrictions on fish consumption for most species will remain for the foreseeable future.

PHYTOPLANKTON, ZOOPLANKTON, AND BENTHOS

Richard P. Barbiero[1], Owen T. Gorman[2], Michael H. Hoff[2], Lori M. Evrard[2], and Marc L. Tuchman[3]

There are no FCOs for phytoplankton, zooplankton, or benthos, but other FCOs recognize that the productivity of Lake Superior is limited (Horns et al. 2003). If population levels of prey fish are supposed to be abundant enough to support both predators and commercial fisheries, then there must be sufficient production at lower trophic levels to support the prey fish.

The open-water plankton communities of Lake Superior are probably the least studied of the Laurentian Great Lakes. Historically, the lake's relative isolation, its enormous size, and the shortage of suitable research vessels have limited the ability to obtain accurate and meaningful data on its offshore phytoplankton and zooplankton communities. Allied with these obstacles are the nature and complexity of the plankton communities themselves: their erratic horizontal distribution, seasonal (phytoplankton) and diurnal (zooplankton) changes in distribution, and the unusually large number of phytoplankton species.

[1]DynCorp Environmental, 1359 W. Elmdale Ave. Suite #2, Chicago, Illinois, U.S.A., 60660

[2]U.S. Geological Survey, Great Lakes Science Center, Lake Superior Biological Station, 2800 Lake Shore Drive East, Ashland, Wisconsin, U.S.A., 54806

[3]United States Environmental Protection Agency, Great Lakes National Program Office, 77 W. Jackson Boulevard, Chicago, Illinois, U.S.A., 60604

Offshore Waters

The Great Lakes National Program Office (GLNPO) of the United States Environmental Protection Agency (USEPA) commenced regular surveillance monitoring of Lake Superior in 1993 and regular biological sampling of benthos in 1997 and plankton in 1998. As with the other lakes, the GLNPO's monitoring effort in Lake Superior is focused on whole-lake responses to changes in loadings of anthropogenic substances, so sampling was restricted to the relatively homogeneous offshore waters. The aim of the offshore monitoring program is to characterize the lake during two well-defined temporally limited periods: the spring isothermal period and the summer stable-stratified period. In Lake Superior, the seasonal development of the offshore communities is not completely known, and this information gap should be considered when interpreting the data presented here.

In 1999, the latest year for which data are available, phytoplankton and zooplankton communities were sampled at 19 widely distributed offshore sites during spring (3-7 May) and summer (18-24 August), and benthos were collected from 11 of these sites during the summer survey. Sampling methods for phytoplankton and zooplankton are described in Barbiero and Tuchman (2001) and Barbiero et al. (2001), respectively.

Phytoplankton

Phytoplankton communities in Lake Superior are characteristically species rich. Nearly 160 phytoplankton taxa were found in May 1999, a number similar to both Lakes Huron and Michigan. Phytoplankton biomass in early May was uniformly low across the lake (Fig. 7). The median of 0.065 $g{\cdot}m^{-3}$ was substantially lower than that of Lake Huron (0.44 $g{\cdot}m^{-3}$) and nearly an order of magnitude less than that seen in Lake Michigan (0.62 $g{\cdot}m^{-3}$). Diatoms were the most diverse and the dominant group, contributing about 40% of the taxa and just under half of total phytoplankton biomass. *Aulacoseira islandica* was dominant, making up nearly 20% of the spring biomass, and the pennates *Asterionella formosa* and *Tabellaria flocculosa* and various species of cryptophytes also contributed considerable biomass to the spring community.

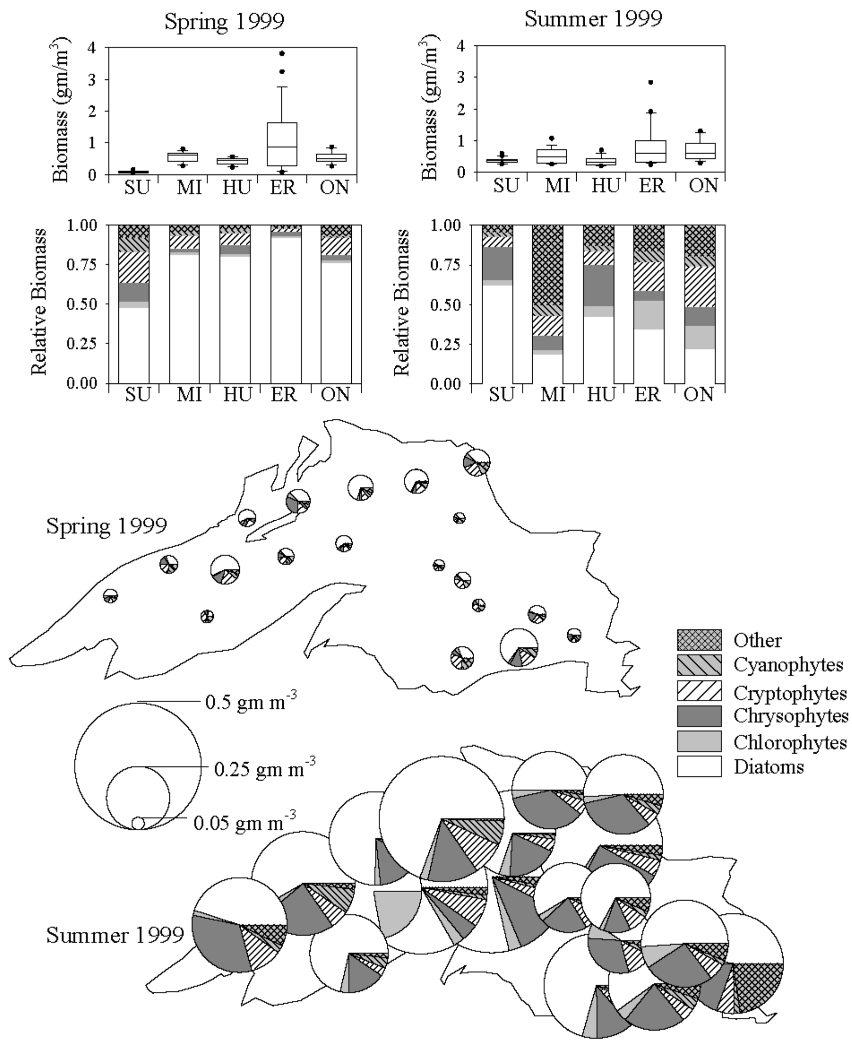


Fig. 7. Upper panel: box plots of phytoplankton biomass across the Great Lakes in spring (May) and summer (August) 1999. Boxes denote 25th and 75th percentiles; lines denote median; whiskers denote 10th and 90th percentiles; individual points denote outliers. Middle panel: whole-lake average relative biomass of major phytoplankton groups for spring and summer 1999. Lower panel: biomass of major phytoplankton groups at each site for spring and summer 1999.

Taxa richness was nearly identical in the summer, although the number of diatom taxa decreased while those of the chrysophytes increased. Phytoplankton biomass in summer was greater than in spring and quite uniform across the lake (Fig. 7). The median biomass of phytoplankton was 0.39 $g \cdot m^{-3}$ and was similar to that seen in Lake Huron (0.34 $g \cdot m^{-3}$), although it was still lower than the 0.58 $g \cdot m^{-3}$ seen in Lake Michigan. Most biomass was concentrated in the centric diatom *Cyclotella*, the pennate diatoms *Fragilaria crotonensis* and *Tabellaria flocculosa*, and chrysophytes of the genus *Dinobryon*.

In 1999, as in 1998 (Barbiero and Tuchman 2001), Lake Superior was distinguished from the other lakes by a markedly lower biomass and smaller diatom community in the spring and by an increase in both overall biomass and diatom dominance in the summer. Relatively smaller spring diatom populations are due to colder water, which tends to delay phytoplankton community development, whereas the increase in diatoms in summer is due to a much greater silica reserve, which is not exhausted during the summer as it is in other Great Lakes (Barbiero et al. 2002). Although little long-term quantitative data exist for the phytoplankton community in the lake, our results are consistent with earlier reports indicating the importance of the diatom genera *Fragilaria*, *Asterionella*, *Tabellaria*, *Melosira* (= *Aulacoseira*), and *Urosolenia* (Davis 1966), as well as the summer dominance of the chrysophyte genus *Dinobryon* (Davis 1966) and the centric diatom *Cyclotella* (Holland 1965; Schelske et al. 1972; Fahnenstiel and Glime 1983). The overall size and composition of the phytoplankton community is indicative of extreme low productivity that has changed little in the past 50 years.

Crustacean Zooplankton

Open-water crustacean communities in Lake Superior in spring (May) and summer (August) are characterized by low species richness, a preponderance of large, deep-living calanoid copepods and their immature forms, and relatively small summer cladoceran populations. Areal biomass of crustaceans (excluding nauplii) in spring 1999 ranged from 0.38-1.45 g $DW \cdot m^{-2}$. The median biomass of 0.62 g $DW \cdot m^{-2}$ was similar to the median biomass in Lake Ontario and northern Lake Michigan and substantially higher than in Lake Erie (Fig. 8). Species diversity was extremely low; only nine taxa were found among the 19 sites and no more than five taxa were found at any one site. The large, deep-living calanoid copepods *Limnocalanus macrurus* and *Leptodiaptomus sicilis*, along with the cyclopoid *Diacyclops thomasi*, accounted for most of the biomass.

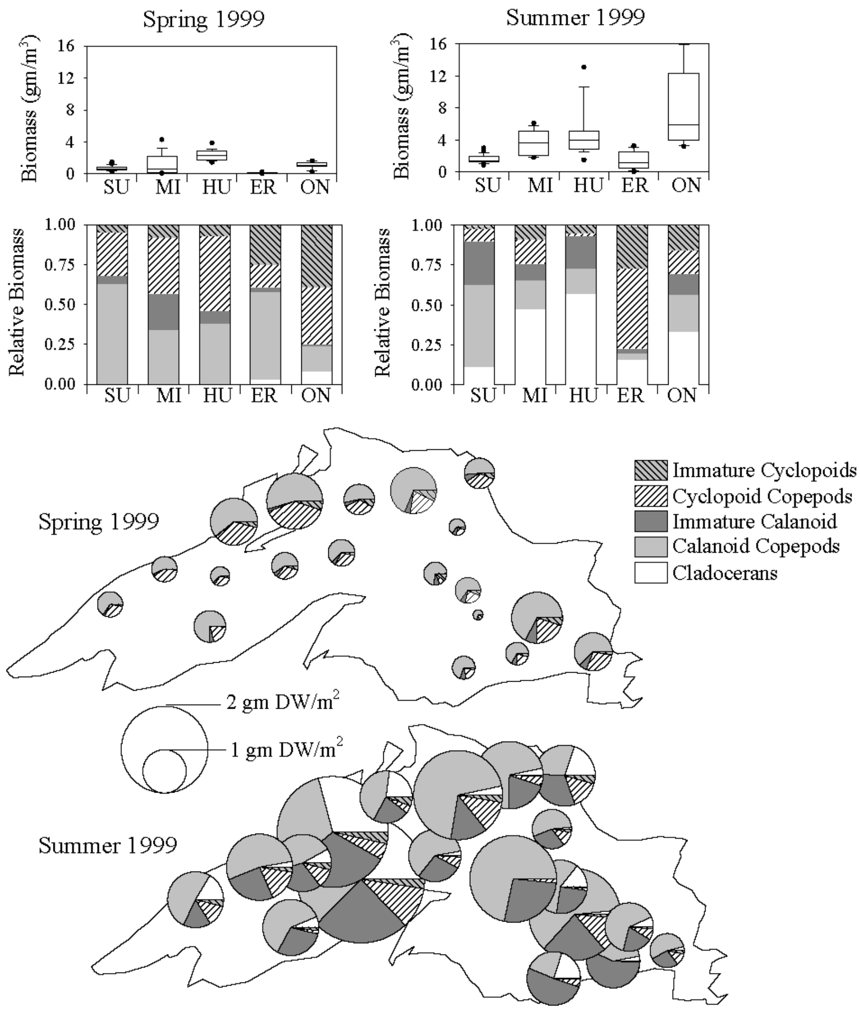


Fig. 8. Upper panel: box plots of areal zooplankton biomass across the Great Lakes in spring (May) and summer (August), 1999. Boxes denote 25th and 75th percentiles; lines denote median; whiskers denote 10th and 90th percentiles; individual points denote outliers. Middle panel: whole-lake average relative biomass of major zooplankton groups for spring and summer 1999. Lower panel: areal biomass of major zooplankton groups at each site for spring and summer 1999.

Median biomass of crustacean zooplankton nearly doubled to 1.32 g DW·m^{-2} from spring to summer (Fig. 8). Diversity increased in the summer due to the appearance of cladocerans. Dominant species included the three copepod species found in spring and the cladocerans *Daphnia galeata mendotae* and *Holopedium gibberum*, which contributed about equally to summer biomass. The non-native predatory cladoceran *Bythotrephes longimanus* was found in modest numbers at most sites, but, in spite of its larger size, accounted for <0.5% of total crustacean biomass. *Leptodora kindtii*, a native predatory cladoceran, was not found in 1999. The average contribution of cladocerans to summer biomass was about 6%, notably lower than in the other Great Lakes.

Benthos

Studies of the benthic fauna in Lake Superior go back over 100 years, and a number of surveys were conducted in the late 1960s and 1970s, mostly in the western basin (Adams and Kregear 1969; Hiltunen 1969; Schelske and Roth 1973; Cook 1975; Dermott 1978). The offshore benthic community in 1999 at depths from 56 to 228 m consisted primarily of the amphipod *Diporeia* spp. (hereafter, diporeia as a common name), the oligochaetes *Stylodrilus heringianus*, unidentified members of Enchytraeidae, the chironomid *Heterotrissocladius*, and members of Sphaeriidae (not identified further, but most probably the deep-living *Pisidium conventus*) (Henson and Herrington 1965). This community has been previously reported in Lake Superior and other Great Lakes (Adams and Kregear 1969; Cook 1975; Reynoldson and Day 1998). Aside from these taxa, very few other benthos have been found in Lake Superior. A total of ten taxa were found among all 11 sites—five or fewer at most sites but nine at a relatively shallow (65 m) site in Whitefish Bay.

In 1999, total benthos density, either deeper or shallower than 90 m, in Lake Superior was lower than in either Lakes Huron or Michigan (Table 3), suggesting that low productivity was limiting benthic populations in Lake Superior. The abundance of diporeia, although low, has not declined as it has in the other four lakes. Abundance of diporeia during 1997-1999 was similar or slightly higher than historical reports (Hiltunen 1969; Schelske and Roth 1973; Cook 1975). In addition, diporeia abundance in all sites but one in 1999 either met or exceeded State of the Lakes Ecosystem Conference criteria.

Table 3. Comparison of mean (+ SE) densities (number·m^{-2}) of the major macroinvertebrate groups in Lakes Superior, Huron, and Michigan in 1999. Numbers in parentheses indicate the number of sites (all sampled in triplicate) in each depth interval for each lake.

Depth (m)	Group	Superior	Huron	Michigan
50-90	*Diporeia* spp.	1,546 ± 149	1,788 ± 177	2,617 ± 212
	Oligochaeta	717 ± 42	864 ± 106	2,743 ± 176
	Sphaeridae	140 ± 23	209 ± 26	906 ± 103
	Chironomidae	29 ± 11	25 ± 9	102 ± 12
	Total	2,435 ± 156	2890 ± 280	6,431 ± 380
		(2)	(4)	(5)
>90	*Diporeia* spp.	377 ± 21	1,793 ± 145	1,707 ± 62
	Oligochaeta	251 ± 35	625 ± 91	825 ± 55
	Sphaeridae	73 ± 8	19 ± 9	46 ± 8
	Chironomidae	10 ± 2	25 ± 8	0 ± 0
	Total	724 ± 51	2,468 ± 207	2,655 ± 64
		(9)	(5)	(4)

Nearshore Waters

Crustacean Zooplankton

The U.S. Geological Survey surveyed the status and trends of crustacean zooplankton communities in nearshore areas of Lake Superior in four ecoregions during mid-May to mid-June in 1989-2000 in conjunction with forage-fish assessments. The four ecoregions are Minnesota North Shore (MNNS) between Two Harbors and the Canadian border (8 stations), Wisconsin Apostle Islands (APIS) (9 stations), Michigan eastern Keweenaw Bay (EKEW) between Sand Bay and Bete Grise (5 stations), and Michigan Whitefish Bay (WFBY) (6 stations) (Fig. 1). Whitefish Bay was sampled only during 1994-2000. Zooplankton were sampled with a conical plankton net towed once vertically from approximately 1 m off the bottom to the surface at stations <5 km from shore at depths of from 30 to <140 m.

During 1989-2000, 14 species of crustacean zooplankton were identified from 244 vertical tows among the four ecoregions. The mean density of the total adult zooplankton over all ecoregions varied annually between approximately 100 and 500 plankters·m^{-3}. Of the four ecoregions surveyed, APIS showed the least interannual variation (approximately 100-350 plankters·m^{-3}). Mean annual total density was highest in MNNS (347·m^{-3}) and the lowest in APIS (226·m^{-3}).

The adult copepods *Leptodiaptomus sicilis*, *Limnocalanus macrurus*, and *Diacyclops thomasi* dominated the zooplankton assemblages across the four ecoregions (Table 4). *L. sicilis* was the most-abundant species in all ecoregions, but its biomass was second to *L. macrurus* in EKEW. Relative abundance of the three species was most even in APIS. *L. macrurus* was the largest and *D. thomasi* the smallest in size (Table 4). For the three species, mean total length (TL) was smallest in APIS and largest in EKEW and WFBY. Data from the spring forage-fish assessments indicate that APIS has the highest relative abundance of planktivorous fishes, and this abundance may be sufficient to suppress the zooplankton abundance and size composition. The domination of spring crustacean zooplankton communities in nearshore areas by the large calanoid copepods *L. sicilis* and *L. macrurus* is typical of cold, oligotrophic lakes, and their domination in Lake Superior, along with the abundance of the smaller cyclopoid *D. thomasi*, has been previously reported (Patalas 1972; Conway et al. 1973; Selgeby 1975; Watson and Wilson 1978).

Table 4. Zooplankton community attributes during 1989-2000 by Lake Superior ecoregion: Minnesota North Shore (MNNS); Wisconsin Apostle Islands (APIS); Michigan Eastern Keweenaw Bay (EKEW), and Michigan Whitefish Bay (WFBY).

Taxa	Ecoregion			
	MNNS	**APIS**	**EKEW**	**WFBY**
Mean density ($n \cdot m^{-3}$)				
Cladocera	0.15	3.18	0.74	2.48
Copepoda (immature)	1.46	2.82	2.96	0.11
Diacyclops thomasi	81.55	70.89	63.64	68.78
Leptodiaptomus sicilis	178.65	80.95	147.66	178.50
Limnocalanus macrurus	85.31	67.88	64.81	64.05
Total	347.02	225.55	279.65	313.73
Relative proportion				
Cladocera	0.00	0.01	0.00	0.01
Copepoda (immature)	0.00	0.01	0.01	0.00
Diacyclops thomasi	0.24	0.32	0.24	0.21
Leptodiaptomus sicilis	0.52	0.36	0.53	0.59
Limnocalanus macrurus	0.24	0.30	0.22	0.19
Mean total length (mm)				
Diacyclops thomasi	1.18	1.07	1.44	1.34
Leptodiaptomus sicilis	1.60	1.57	2.06	1.71
Limnocalanus macrurus	1.98	1.85	2.32	2.41
Mean biomass ($mg \cdot m^{-2}$)				
Diacyclops thomasi	15.87	10.94	10.98	20.41
Leptodiaptomus sicilis	172.14	78.50	181.84	219.37
Limnocalanus macrurus	83.08	44.98	207.64	188.53
Total	271.09	134.43	400.46	428.31

PREY FISHES

Michael H. Hoff and Owen T. Gorman
U.S. Geological Survey
Great Lakes Science Center
Lake Superior Biological Station
2800 Lake Shore Drive East
Ashland, Wisconsin, U.S.A. 54806

> *…a self-sustaining assemblage of prey dominated by indigenous species at population levels capable of supporting desired populations of predators and a managed commercial fishery.*

In this section, we report progress on achieving the above goal (Horns et al. 2003) by describing the dynamics of the major prey-fish populations during 1978-2000. The data in this report were obtained during spring (May and June) bottom-trawl surveys conducted by the U.S. Geological Survey at 33-53 stations in U.S. waters and 26-35 stations in Ontario waters at depths of 13-85 m. These data represent indexes of densities or biomass rather than estimates of absolute abundance.

Lake Herring

During the historical reference period of 1916-1940, commercial yield of lake herring was greater than that for all other species combined (Baldwin et al. 1979). After 1941, lake herring yield (Fig. 3) and populations declined, and that decline has been largely attributed to overharvest (Peck et al. 1974; Selgeby 1982), although Anderson and Smith (1971a) indicated that competition and predation by rainbow smelt likely were factors in the decline. The LSTC recommended in 1986 that lake herring harvest by fisheries be limited to no more than the average for 1974-1983 until either populations recover or better methods are developed to estimate surplus production. In 1992, the LSTC recommended that spawning-season closures, quotas, or gear limits should be implemented to manage commercial harvest until the populations recover.

Lake herring population dynamics have been driven more by variations in recruitment than by either mortality or growth. Lake herring mean biomass in U.S. waters during 1978-2000 was higher than that for any other prey species. Biomass was low during 1978-1984, increased during the mid-1980s and peaked in 1990 as the large 1984 year class recruited to the population, declined during 1991-1997, and then increased thereafter (Fig. 5b in Bronte et al. 2003). Mean biomass during 1998-2000 was 20% lower than that measured during the previous 20 years. Annual estimates of lake herring biomass in Ontario waters have been lower than in U.S. waters, except in 1996, but population trends have been similar. Mean biomass in Ontario waters during 1998-2000 was 46% lower than that measured during the previous nine years.

Dynamics of lake herring biomass in U.S. waters differed across jurisdictions. In Minnesota waters, biomass estimates have been lower than in any other jurisdiction, possibly because there is little suitable substrate for bottom trawling. In Wisconsin waters, survival and growth of the 1984 and 1988-1990 year classes resulted in a 12-fold increase in lake herring mean biomass from 1978-1984 to 1985-1995. In Michigan waters, lake herring recovery followed the Wisconsin pattern through the early 1990s, but biomass declined more sharply in Michigan during 1990-1997. Although biomass in Michigan waters has increased substantially since 1997, mean biomass during 1998-2000 was 56% lower than the mean for the previous 20 years.

Abundance of the 1977-1998 lake herring year classes in U.S. waters fluctuated by a factor of 562. Strong year classes occurred consecutively only during 1988-1990, and the only strong year class that followed those was produced in 1998 (Fig. 5a in Bronte et al. 2003). The 1999 year class was the eleventh most-abundant year class. Most adult lake herring present in the lake in 2000 are of the 1988-1990 and 1998 year classes.

The inferred overharvest of lake herring in the 1960s and 1970s caused declines in the abundance of Wisconsin and Michigan spawning aggregations, which were probably discrete stocks (Peck et al. 1974; Selgeby 1982). The reaction to these declines by fishery-management agencies was to set a lakewide strategic objective of rehabilitating lake herring stocks to historical levels of abundance to support lake trout rehabilitation, production of other predators, and commercial harvest (Busiahn 1990). Information at the time indicated that at least seven stocks existed in Michigan waters (Peck and Wright 1972), at least six stocks existed in Wisconsin waters (Selgeby 1982), and two existed in Minnesota waters (Hassinger and Kuechenmeister 1972). This information was based on differences in gillnet CPUE, mean and ranges of length, body color, flesh characteristics such as ease of bruising and oiliness, sex ratios, year-class strengths, growth rates, and distances between adjacent spawning grounds. Recent attempts to discriminate lake herring stocks using genetics, life-history parameters (growth, mortality, diet, and parasites), and trace elements in otoliths suggest that discrete stocks exist, but these methods have not been successful in discriminating stocks reliably (Todd 1981; Fields and Philip 1995; Hoff et al. 1995; Link et al. 1995; Bronte et al. 1996; Hoff et al. 1997a; Hoff et al. 1997b). Hoff (2004) evaluated the accuracy of discriminating among lake herring spawning aggregations using whole-body morphometrics and correctly classified 53% of all fish from all spawning aggregations. The morphometrics study and other evidence suggest that discrete stocks of lake herring exist at Grand Marais and Two Harbors, Minnesota; Duluth-Superior, Cornucopia, Sand Island, and Stockton Island, Wisconsin; Eagle Harbor, Keweenaw Bay, Munising, and Whitefish Bay, Michigan; and Thunder Bay and Black Bay, Ontario.

Rainbow Smelt

Rainbow smelt were first documented in Lake Superior in 1930 (Becker 1983) and became abundant enough to harvest commercially in Michigan waters in 1938. In Minnesota waters, commercial yield of rainbow smelt fluctuated at relatively high levels from the mid-1960s to the late 1970s, declined severely during 1979-1981, and has remained relatively low through 1998 (Fig. 3a in Bronte et al. 2003).

Rainbow smelt mean biomass during 1978-2000 ranked second among species behind lake herring. In U.S. waters, biomass declined by >85% between 1978 and 1981, increased to 80% of the 1978 level by 1986, decreased between 1986 and 1994, peaked moderately in 1995, and then declined thereafter (Fig. 3b in Bronte et al. 2003). Biomass trends were similar among state jurisdictions, and mean biomass in U.S. waters during 1998-2000 was 52% lower than the mean for the previous 20 years. Mean biomass in Ontario waters was 125% greater than in U. S. waters during 1989-2000, but in 2000 it was lower than in any previous year.

In U.S. waters, the strength of the 1977, 1979, 1980, and 1981 rainbow smelt year classes was low (Fig. 3c in Bronte et al. 2003), which accounts for much of the decline in biomass during the early 1980s. Trends in biomass of rainbow smelt were also affected by mortality rates. The total annual mortality rate of rainbow smelt ages 1-4 increased from about 37% in 1978 to >75% by 1981 and, coupled with poor recruitment in 1977 and 1979-1981, caused the density of large rainbow smelt (>200 mm TL, mostly ages ≥4) to decline sharply in 1980 and again in 1981. Densities of large rainbow smelt have remained low since 1981 and will probably not increase unless mortality declines (Fig. 3d in Bronte et al. 2003). High mortality and declines in density and biomass of rainbow smelt since the early 1980s have been attributed largely to recovery of lean and siscowet lake trout populations (Bronte et al. 2003).

Bloater

Bloater mean biomass in U.S. waters during 1978-2000 ranked third among species behind lake herring and rainbow smelt. Biomass was low during the 1980s, increased between 1991 and 1994, and then declined irregularly during 1995-1999 (Fig. 6c in Bronte et al. 2003). Biomass in 1998-2000 was 13% higher than the mean for the previous 20 years. Biomass has tended to be higher in U.S. waters than in Ontario waters, but trends have been similar in both jurisdictions. Mean biomass in Ontario waters during 1998-2000 was 36% higher than the mean for the previous nine years. Among U.S. jurisdictions, bloater biomass was slightly higher in Michigan waters than in Wisconsin waters. Bloater biomass estimates for Minnesota waters were considerably lower than in the two other jurisdictions, but this may be due to a lack of suitable trawling grounds.

Ninespine Stickleback

Mean biomass of ninespine stickleback ranked fourth among prey fish in U.S. waters during 1978-2000. Biomass in U.S. waters increased slightly during the 1980s and then declined during the 1990s (Fig. 6g in Bronte et al. 2003). Biomass declined 66% between 1999 and 2000 to its lowest level for the period, and the mean during 1998-2000 was 25% lower than that for the previous 20 years. Biomass of ninespine stickleback was higher in Ontario waters than in U.S. waters, but it also decreased during the late 1990s. Mean biomass in Ontario waters during 1998-2000 was 58% lower than the mean for the previous nine years, and biomass in 2000 was the lowest measured during the 12-year assessment. Among U.S. jurisdictions, biomass of ninespine stickleback during 1978-2000 was highest in Wisconsin waters and practically zero in Minnesota waters. The highest annual biomass in Wisconsin and Michigan waters was in 1979.

Trout-Perch

Mean biomass of trout-perch in U.S. waters during 1978-2000 was identical to that of slimy sculpin and lower than that of the other major prey-fish species. Biomass during 1978-1982 was relatively low, increased during 1983-1989, decreased between 1989 and 1992, and has fluctuated without trend since then (Fig. 9). In Ontario waters, trout-perch annual biomass estimates were similar to those for U.S. waters except that biomass declined after 1997 to the lowest recorded during 1989-2000 (Fig. 9). Mean biomass in Ontario waters in 1998-2000 was 86% lower than that for the previous nine years and 88% lower than in U.S. waters. Among U.S. jurisdictions, biomass of trout-perch tended to be higher in Michigan than in Wisconsin waters and extremely low in Minnesota waters (Fig. 9).

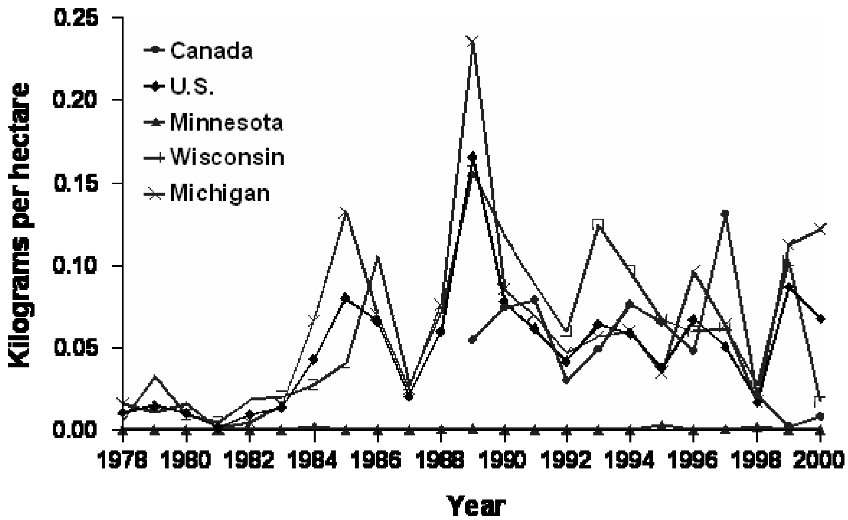


Fig. 9. Biomass (kg/ha) of trout-perch in Lake Superior estimated from spring (May-June) bottom-trawl surveys, 1978-2000.

Sculpins

In U.S. waters, slimy sculpin biomass decreased 82% from 1979 to 1980 and remained low thereafter (Fig. 6d in Bronte et al. 2003). Mean biomass during 1998-2000 was 67% lower than the mean for the previous twenty years. In Ontario waters, mean biomass was 40% higher than the mean for U.S. waters during 1998-2000, but biomass declined in the late 1990s, and the 1999-2000 mean was 38% lower than the mean for the previous nine years. Among U.S. jurisdictions, biomass of slimy sculpin tended to be higher in Michigan than in Wisconsin or Minnesota waters. Biomass of deepwater and spoonhead sculpins also declined during 1978-1999 (Figs. 6h, 6f in Bronte et al. 2003).

Recommendations

- Develop lake herring population models to track stock dynamics and to estimate sustainable harvests and predator consumption
- Initiate long-term, lakewide studies of the dynamics of prey fishes in the offshore waters deeper than 80 m where very little sampling has been done
- Evaluate the spring bottom-trawl survey sampling design and modify, as appropriate, to accurately assess the entire nearshore prey-fish community
- Design and implement a lakewide acoustic sampling program to estimate abundance of pelagic fishes
- Determine the effects of siscowet predation on the deepwater prey assemblage

LAKE TROUT

Shawn P. Sitar[1], Charles R. Bronte[2], Mark P. Ebener[3], William P. Mattes[4], Michael Petzold[5], Stephen T. Schram[6], and Donald R. Schreiner[7]

> *…achieve and maintain genetically diverse self-sustaining populations of lake trout that are similar to those found in the lake prior to 1940, with lean lake trout being the dominant form in nearshore waters, siscowet lake trout being the dominant form in offshore waters, and humper lake trout being a common form in eastern waters and around Isle Royale.*

Each of the three forms recognized in the above goal (Horns et al. 2003) are long-lived and late-maturing (Schram and Fabrizio 1998), yet they are unique in that they have different morphological characteristics (Burnham-Curtis 1993; Moore and Bronte 2001), occupy different bathymetric habitats in the lake, have different reproductive life histories, and have different energetic requirements (Henderson and Anderson 2002). Siscowets occupy the highest trophic level in Lake Superior followed by burbot, then lake trout (Harvey and Kitchell 2000).

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[1]Michigan Department of Natural Resources, 484 Cherry Creek Road, Marquette, Michigan, U.S.A., 49855

[2]U.S. Fish and Wildlife Service, 2661 Scott Tower Drive, New Franken, Wisconsin, U.S.A., 54229

[3]Chippewa Ottawa Resource Authority, 179 W. Three Mile Road, Sault Ste. Marie, Michigan, U.S.A., 49783

[4]Great Lakes Indian Fish and Wildlife Commission, P.O. Box 9, Odanah, Wisconsin, U.S.A., 54861

[5]Ontario Ministry of Natural Resources, 1235 Queen Street East, Sault Sainte Marie, Ontario, Canada, P6A 2E5

[6]Wisconsin Department of Natural Resources, 141 South 3rd Street, Bayfield, Wisconsin, U.S.A., 54814

[7]Minnesota Department of Natural Resources, 5351 North Shore Drive, Duluth, Minnesota, U.S.A., 55804

Humpers probably occupy an even lower trophic level given that their diet is primarily *Mysis relicta* and sculpins (MPE, unpubl. data). Recent survey data and modeling results indicate that siscowets are the dominant form in offshore waters (Bronte et al. 2003), but there are indications of increasing siscowet abundance in nearshore areas dominated typically by lean lake trout (hereafter, lake trout). However, there currently is little trophic overlap between siscowets and lake trout except at smaller sizes (Harvey et al. 2003).

Lake trout rehabilitation efforts over the last 20 years have resulted in establishment of self-sustaining populations in most nearshore waters, which support varying levels of fishery harvest. Based on criteria defined in the lake trout restoration plan for Lake Superior (Hansen 1996), stocking hatchery-reared lake trout ceased in 1997 in most of Michigan's waters. Offshore lake trout populations at Isle Royale and Stannard Rock have been described, and they are currently presumed to be healthy. Furthermore, there are other offshore lake trout populations, such as Big Reef (32 km north of Munising (Fig. 1)), that have not been assessed. These offshore populations have recovered rapidly from remnant wild fish and provide a unique comparison for nearshore lake trout rehabilitation efforts. There is not much recent information for most humper populations, but humpers were abundant during the mid- to late 1990s in August-September gillnet surveys on a reef in eastern waters (Fig. 1, MI-7; M. Holey, U.S. Fish and Wildlife Service, 2661 Scott Tower Drive, New Franken, WI, 54229, unpubl. data), and they are believed to be abundant on other historical grounds because their offshore location and small size minimizes exposure to fisheries and sea lamprey predation.

Recent research evaluating stock-recruitment dynamics and comparisons of historical and modern lake trout abundance indicate that abundance in most of Michigan's waters has increased since the 1970s and, despite a decline during the 1990s, is at or exceeds historical levels (Wilberg et al. 2003; Richards et al. 2004). Concurrent with the increases in abundance, lake trout growth rates have declined progressively, indicating that lake trout populations may be near or at density-dependent levels. Other factors supporting attainment of density-dependant levels include declining abundance of lake trout prey fishes and encroachment of siscowet populations into lake trout habitat (nearshore). Lake trout population-density trends in other areas of Lake Superior are following the pattern in Michigan waters.

Stocking

Stocking hatchery lake trout, along with sea lamprey control and restrictions on fishing, was the primary mechanism for rebuilding lake trout populations. During the 1990s, the total numbers of lake trout stocked lakewide declined from a peak of over 3.2 million fish in 1991 to <1 million fish in 2000 (Table 2), reflecting primarily the reestablishment of self-sustaining populations. Predation by the increased numbers of wild fish is believed responsible for decreased survival of stocked fish in recent years (Hansen et al. 1994). Stocking of lake trout continues in some areas of Minnesota, Wisconsin, and Ontario and in one Michigan unit (MI-4).

Lake Trout Abundance

Indices of lake trout abundance are based on data collected from long-term annual standardized spring (April-June) lake trout surveys with gillnets at multiple stations in nearshore management units (Fig. 1). The gillnetting was done by either agency teams or by contracted commercial fishers (Hansen et al. 1995; Hansen et al. 1998). The relative abundance index is calculated as the geometric mean CPUE and expressed as the number of lake trout caught per kilometer of net fished adjusted for the number of nights fished and the number of fish in the net (Hansen et al. 1998). Relative abundance data were not available for all management units in all years, especially in many of the Ontario management units. Since there were some differences in the sampling designs across agencies, the trends in relative abundance should be examined within jurisdictions and not compared quantitatively across jurisdictions.

In U.S. jurisdictions, where surveys have been ongoing since the 1960s, mean relative abundance of wild and hatchery lake trout during 1993-2000 was compared to the mean for 1983-1992 (Fig. 10). In Minnesota waters, the mean CPUE of wild lake trout during 1993-2000 exceeded the mean during 1983-1992, but hatchery lake trout CPUE was lower during 1993-2000 than during 1983-1992. Hatchery CPUE was higher than that for wild lake trout in units MN-1 and MN-2 and CPUE of both wild and hatchery lake trout was highest in MN-1. In Wisconsin waters, wild lake trout CPUE during 1993-2000 was higher than during 1983-1992, but hatchery lake trout CPUE was lower during 1993-2000 than during 1983-1992. CPUE of both wild and hatchery lake trout was higher in WI-2 than in WI-1. In Michigan waters, wild lake trout mean CPUE for 1993-2000 was lower than the 1983-1992 mean in units MI-2, MI-3, MI-4, and MI-5, higher in MI-6, and similar in MI-7 and MI-8 (Fig. 10). Hatchery lake trout CPUE during 1993-2000 was highest in MI-2 and MI-8, and the means for all units were lower than during 1983-1992, except in MI-2. Wild lake trout mean CPUE during 1993-2000 was highest in MI-2, MI-5, and MI-6; lowest in MI-3 and MI-8; and higher than that for hatchery fish in all units except MI-8.

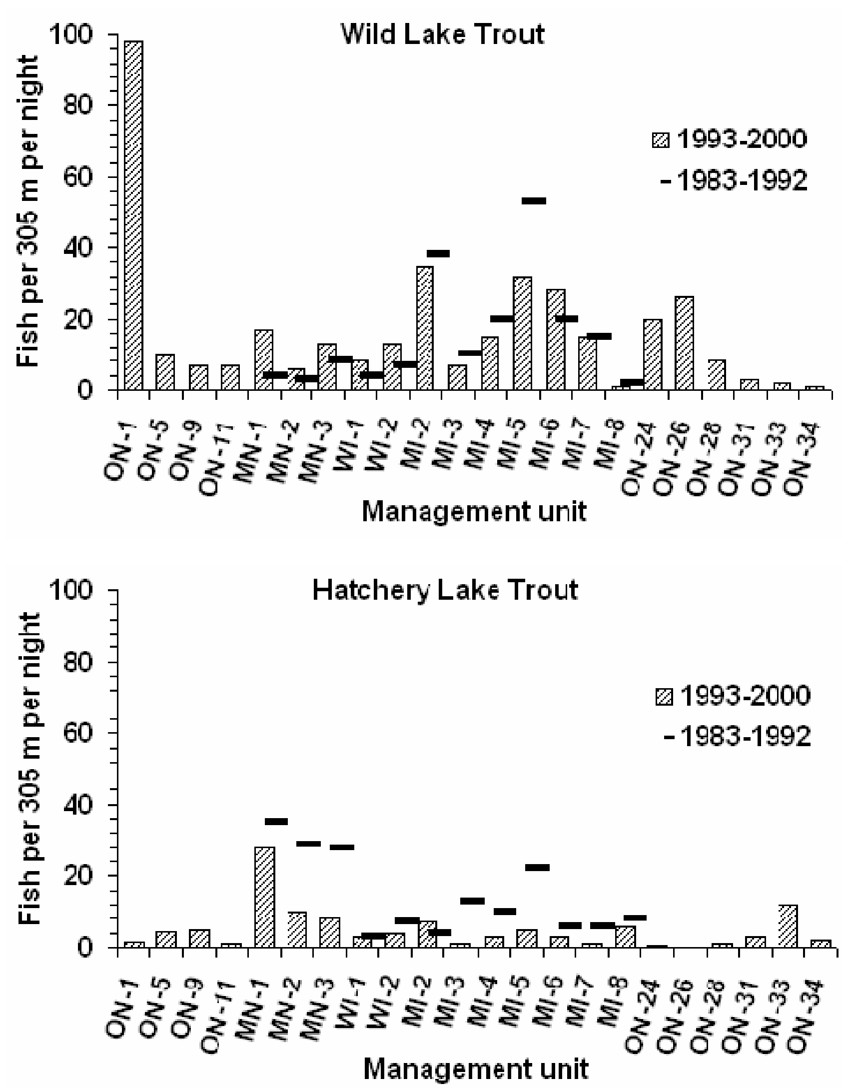

Fig. 10. Mean relative abundance (geometric mean number per 305 m of gillnet per night) of wild and hatchery lake trout in Lake Superior management units during 1993-2000 (bars) and 1983-1992 (horizontal lines) based on spring (April-June) large-mesh gillnet surveys.

In Ontario waters, lake trout relative-abundance data were collected from commercial fishery monitoring and catch reports, which did not distinguish between lake trout and siscowets. The ratio of siscowet to lake trout from the LSTC lakewide siscowet survey was used to estimate the proportion of lake trout. Relative abundance of wild lake trout was highest in ON-1, ON-24, and ON-26, and the highest abundance of hatchery lake trout was in ON-33 (Fig. 10).

Siscowet Abundance

The status of siscowet populations was not consistently evaluated until the LSTC began coordinated and standardized assessments of abundance and depth distribution in all jurisdictions in 1996, 1997, and 2000. Other siscowet data were available previously from commercial fishery monitoring and agency summer (July-August) graded-mesh gillnet surveys.

Siscowet abundance has increased lakewide since the 1950s—the catch rate in commercial fisheries increased from about 20 $kg \cdot km^{-1}$ in 1952 to about 300 $kg \cdot km^{-1}$ in 1997 (Fig. 13 in Bronte et al. 2003). In Michigan waters, the CPUE of siscowets in summer surveys during 1993-2000 was almost twice as high as during 1985-1992 (Fig. 11), and this increase was also observed in the spring lake trout surveys.

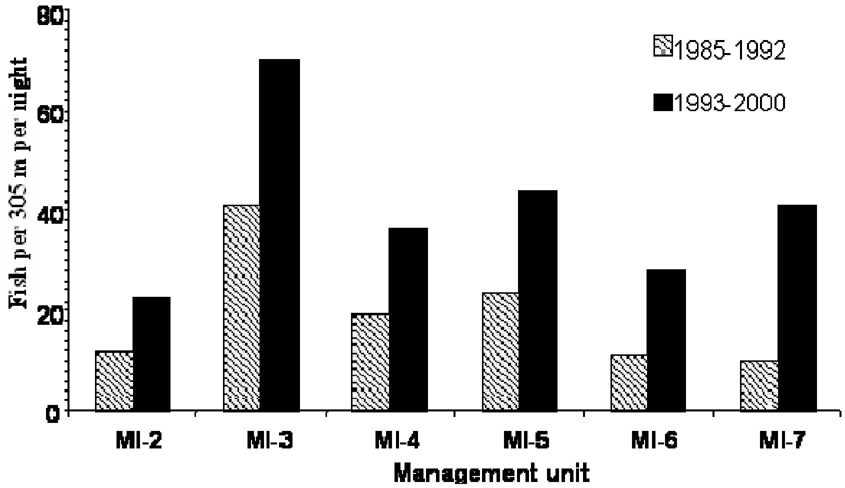

Fig. 11. Mean relative abundance (geometric mean number per 305 m of gillnet per night) of siscowets in Michigan waters of Lake Superior during 1985-1992 and 1993-2000 based on summer (July-August) graded-mesh gillnet surveys.

Lakewide relative abundance of siscowets in U.S. waters during 1996-1997 and 2000 was substantially greater than that of lake trout in all but the shallowest depth stratum (<37 m) (Fig. 12). The ratio of siscowets to lake trout increased with depth, and the lakewide average ratio for all depths was 14:1. The highest siscowet densities were at depths >146 m. Lake trout densities were highest at depths <74 m.

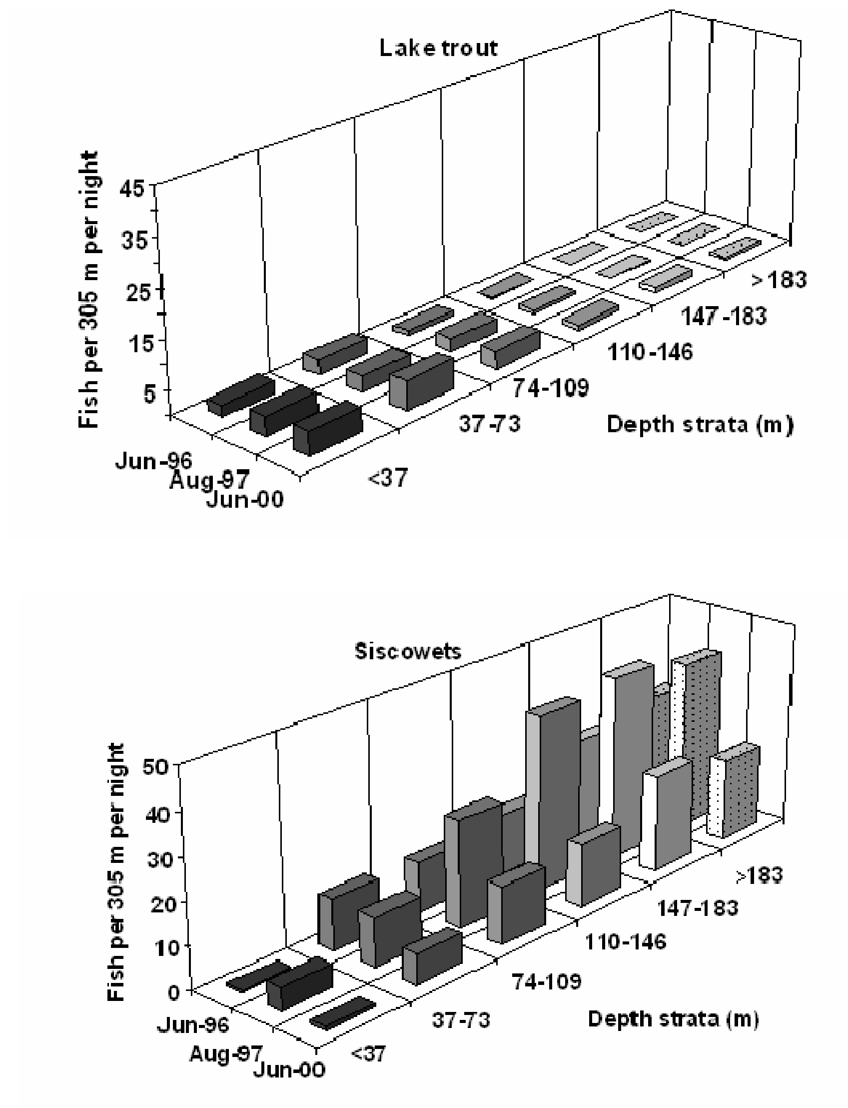

Fig. 12. Mean relative abundance (geometric mean number caught per 305 m of gillnet per night) of siscowet and lake trout in 36-meter depth strata of Lake Superior management units in June 1996, August 1997, and June 2000 graded-mesh gillnet surveys.

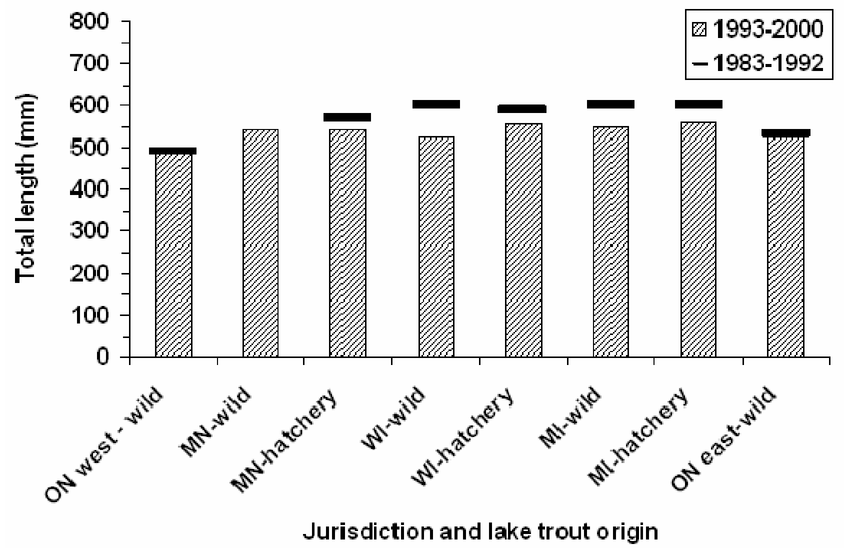

Fig. 13. Mean total length of age-7 wild and hatchery lake trout from Minnesota, Wisconsin, Michigan, and Ontario waters of Lake Superior during 1993-2000 (bars) and 1983-1992 (horizontal line).

Mortality

The target maximum total annual mortality rate for lake trout rehabilitation in Lake Superior was established at 45% (Hansen 1996). Total annual mortality rates were calculated for lake trout using catch-curve analysis (Ricker 1975) from spring gillnet surveys and were adjusted for a 20% overestimation bias due to gillnet selectivity (Hansen et al. 1997). Management units in this analysis were all nearshore U.S. units and Ontario units 1, 14, 24, 28, 31, and 33 (Fig. 1). Mean total mortality rates during 1993-2000 (adjusted for gillnet selectivity bias) ranged from 16% in MN-1 to 55% in MI-8 and were below the 45% target maximum in all management units except MI-8, ON-14, and ON-24.

Growth

Growth of lake trout was tracked by calculating the mean TL at age 7 for fish collected during spring gillnet surveys in U.S. waters and for fish collected from commercial fisheries in Ontario waters. Age determinations were based on scale or otolith analysis. In Minnesota, age data were only available for hatchery fish. In U.S. waters, mean TL at age 7 was lower during 1993-2000 than during 1983-1992, which indicated a declining growth rate, whereas mean TL for the two periods was similar in Ontario waters (Fig. 13). The declining growth in U.S. waters is likely due to increasing lake trout and siscowet abundances and declining abundance of prey fishes.

Sea Lamprey Predation

The FCO for sea lamprey (Horns et al. 2003) is to:

> *…suppress sea lampreys to population levels that cause only insignificant mortality on adult lake trout.*

Insignificant mortality was defined as <5% (Horns et al. 2003). The level of sea lamprey predation was indexed by the number of Type A, Stages I, II, and III sea lamprey marks (King 1980) per 100 lake trout in spring surveys in U.S. waters and in commercial catches in Ontario waters. In U.S. waters, sea lamprey marking rates on lake trout declined to low levels in 1994 and then increased steadily. Marking rates on all sizes of lean lake trout increased from 2.5 marks$\cdot$100^{-1} fish in 1994 to 6.0 marks$\cdot$100^{-1} fish in 2000, and rates on lake trout $\geq$635-mm long increased from 5 marks$\cdot$100^{-1} fish to 20 marks$\cdot$100^{-1} fish. The sea lamprey-induced mortality rate on adult lake trout averaged 16% in 2000, well above the target of <5%. Wounding rates in U.S. and Ontario waters were different for most years during 1986-2000, but they were similar in 2000 (Fig. 14).

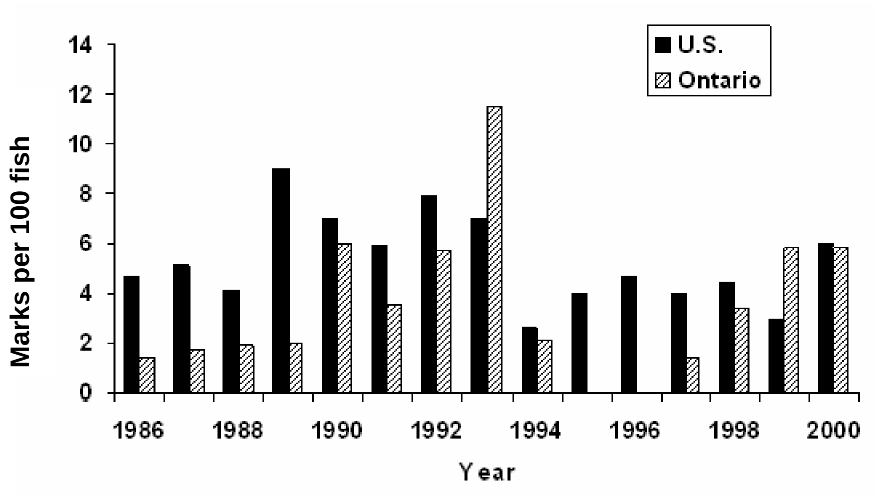

Fig. 14. Mean sea lamprey marking rate (number of Type A, Stages I, II, and III marks per 100 fish) for lake trout in U.S. and Ontario waters of Lake Superior during 1986-2000. No Ontario data were available in 1995 and 1996.

Fishery Harvest

Annual recreational harvest of lake trout in U.S. waters during 1993-1999 averaged about 52,000 fish. In recent years, commercial harvest of lake trout averaged 288,000 kg·y^{-1} and was about 15% of the historical (1929-1943) average (Fig. 3). The average commercial harvest during 1993-2000 was lower than during 1983-1992 (Fig. 3). The highest harvest was in Michigan waters and averaged 137,000 kg·yr^{-1} during 1993-2000.

Research and Modeling

Historical fishery yields (1929-1943) that occurred prior to the collapse of lake trout populations in the 1950s have been compared to current yields to gauge the progress of lake trout rehabilitation. The previous FCO for lake trout was to achieve a population that was capable of sustaining an annual harvest of 1.8 million kg, which was the average yield during 1929-1943 (Busiahn 1990). Wilberg et al. (2003) reexamined historical and current lake trout-abundance indices (density) and commercial-fishery dynamics for

Michigan waters and assessed the soundness of the 1929-1943 reference period. Lake trout densities during 1984-1998 were at or higher than the mean CPUE during 1929-1943 in MI-2, MI-3, MI-4, MI-5, and MI-6. It is believed now that lake trout were overexploited in most Michigan waters during the reference period, and using the yield from this period may be unrealistic (Wilberg et al. 2003).

Richards et al. (2004) used spring survey CPUE to evaluate lake trout stock-recruitment dynamics in Michigan units MI-3, MI-4, MI-5, MI-6, and MI-7. Stock-recruitment relationships were different among units, and peak recruitment was achieved during 1970-1998, except in MI-4. These results support the general notion that most lake trout populations in Michigan waters are at or near density-dependent levels.

Statistical catch-at-age (SCAA) models were developed for wild lake trout in MI-5, M-6, and MI-7 (Bence and Ebener 2002). The models were used to assess the allocation of lake trout mortality among its causes and to evaluate mortality rates in relation to reproductive potential. Natural mortality was the leading cause of death among lake trout age 8 and older in MI-5, MI-6, and MI-7 followed in decreasing order by sea lamprey-induced mortality, commercial-fishing mortality, and recreational-fishing mortality (Fig. 15).

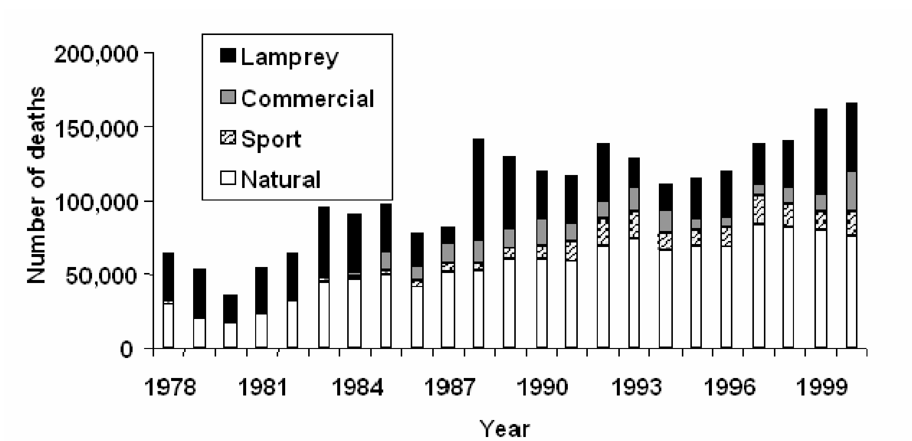

Fig. 15. Allocation of mortality for age-8 and older wild lake trout in Michigan's Lake Superior management units MI-5, MI-6, and MI-7 during 1978-2000 estimated from statistical catch-at-age modeling.

Lake trout biomass estimated from the SCAA models was used in an equilibrium yield model to project spawning-stock production under a given set of mortality rates in MI-5, MI-6, and MI-7. Spawning-stock-biomass-produced-per-recruit (SSBR) values were calculated from the equilibrium model and defined as the amount of female biomass produced per recruit under the current or target mortality rates. Current SSBRs were based on the average mortality during 1998-2000 and were compared to the SSBR for the target total annual mortality of 45% (Bence 2002). The current SSBRs in MI-5, MI-6, and MI-7 were above the target minimum values, indicating mortality rates below 45% and reproductive potential above the minimum necessary for increasing abundance.

Recommendations

- Develop and evaluate statistical population models for lake trout in each management unit where the necessary data exist
- Implement a lakewide mark and recapture study of lake trout spawning stocks to determine movement, survival, growth, and mortality
- Compare population parameters of current lake trout populations to parameters of historical (1929-1943) populations in Wisconsin, Minnesota, and Ontario waters
- Collect age, growth, and mortality data and develop stock-assessment models for siscowet populations
- Develop bioenergetics models to determine the biomass of lake trout and other predators that can be supported by existing prey-fish populations
- Map and quantify lake trout habitat important for reproduction and juvenile survival
- Assess the status of important offshore (e.g., Isle Royale, Stannard Rock, Big Reef, Superior Shoal, and Michipicoten Island) lake trout populations

LAKE WHITEFISH

Michael Petzold
Ontario Ministry of Natural Resources
1235 Queen Street East
Sault Ste. Marie, Ontario, Canada P6A 2E5

…maintain self-sustaining populations of lake whitefish within the range of abundance observed during 1990-1999.

Abundance of whitefish has increased dramatically over the past 40 years (Ebener 1997), but whether it stays high as specified in the above goal (Horns et al. 2003) remains to be seen. The larger spawning and nursery areas in Lake Superior include the Apostle Islands, Isle Royale, the northern and eastern Keweenaw Peninsula, Whitefish Bay, Agawa Bay, and Thunder Bay. Spawning populations formerly existed in the St. Louis River and may still exist in the Kaministiquia and Michipicoten Rivers. Hydroelectric development of these rivers may have impeded restoration of whitefish, but the overall contribution of river populations to total whitefish biomass is likely small. Gravel extraction in the Ontario waters of Lake Superior may have resulted in some loss of spawning habitat, but extraction activities have ceased.

Management

The current guidelines of a 60-65% maximum total annual mortality and a maximum harvest of 0.11 $kg \cdot ha^{-1}$ were established in 1990 (Busiahn 1990). The harvest guideline, when converted to nearshore habitat (<80 m) only, is approximately 0.51 $kg \cdot ha^{-1}$. Harvests in Whitefish Bay, Thunder Bay, Black Bay, and the south Pukaskwa region have exceeded the guideline, but these high harvests were justified in Thunder and Black Bays because of high abundance, as indicated by a CPUE exceeding 100 $kg \cdot km^{-1}$. In Whitefish Bay, however, harvest is based not on the guidelines but on total allowable catch calculated from SCAA modeling, which estimates biomass and recruitment of adult whitefish using a maximum total annual mortality rate of 65% and a minimum spawning-potential reduction from an unfished state of 0.2 (Bence and Ebener 2002).

Commercial Yield and Abundance

The highest reported catch on record (2,200 t) occurred in 1889 (Fig. 16), the first year that harvest records from all jurisdictions were recorded (Baldwin et al. 1979). Thereafter, yield declined continuously until the mid-1920s where it leveled off at around 300 t. The initial decrease in yield was mainly due to exploitation of climax populations and the destruction of habitat via deposition of woody debris from extensive logging (Lawrie and Rahrer 1972, 1973). The yield of whitefish increased gradually to 700 t by 1950, declined sharply after 1950 as agencies acted to greatly reduce large-mesh gillnet effort following collapse of lake trout populations, and then increased since the 1960s to current levels (approximately 1,500 t) (Fig. 16). The commercial harvest is currently taken with large-mesh gillnets in Ontario waters and with trapnets and gillnets in U.S. waters. Angling harvest amounts to <1% of the commercial harvest.

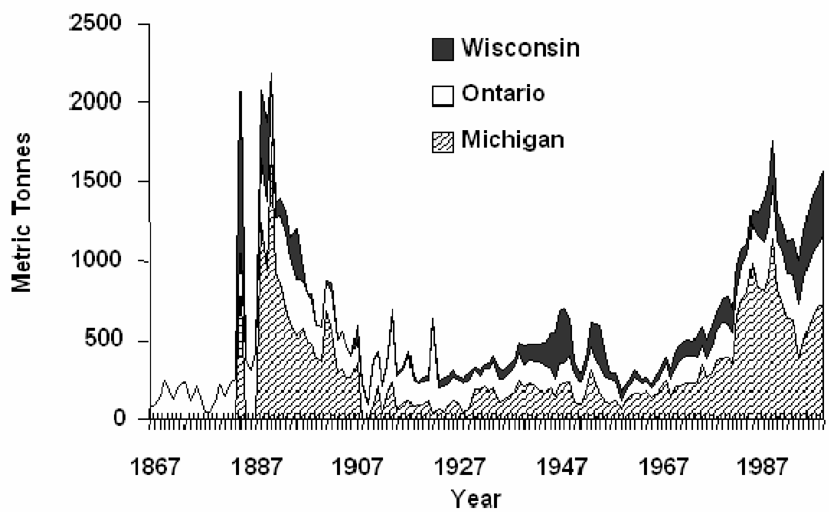

Fig. 16. Annual commercial fishery yield (metric tonnes) of lake whitefish from Ontario, Michigan, and Wisconsin waters of Lake Superior during 1889–1999 (Baldwin et al. 1979, 2002).

The increase in whitefish abundance during 1970-1999 in Ontario, Michigan, and Wisconsin waters was associated with an increased CPUE in large-mesh gillnets (Fig. 17). Adult biomass in Michigan waters increased steadily after 1980 in western management units but decreased in the easternmost unit (Whitefish Bay) (Bence and Ebener 2002). In Ontario waters of Whitefish Bay (ON-33, ON-34), CPUE peaked above 100 $kg \cdot km^{-1}$ in the latter half of the 1980s but declined thereafter. In western Ontario waters (ON-1) (Fig. 1), the spring-summer CPUE increased after 1990 and leveled off above 200 $kg \cdot km^{-1}$. These changes in abundance were also detected in surveys. Estimates of whitefish biomass from bottom-trawl surveys increased from 1.0 $kg \cdot km^{-1}$ during 1978-1983 to 2.5 $kg \cdot km^{-1}$ during 1984-1988 and to 6.5 $kg \cdot km^{-1}$ during 1989-1994 (Bronte et al. 2003).

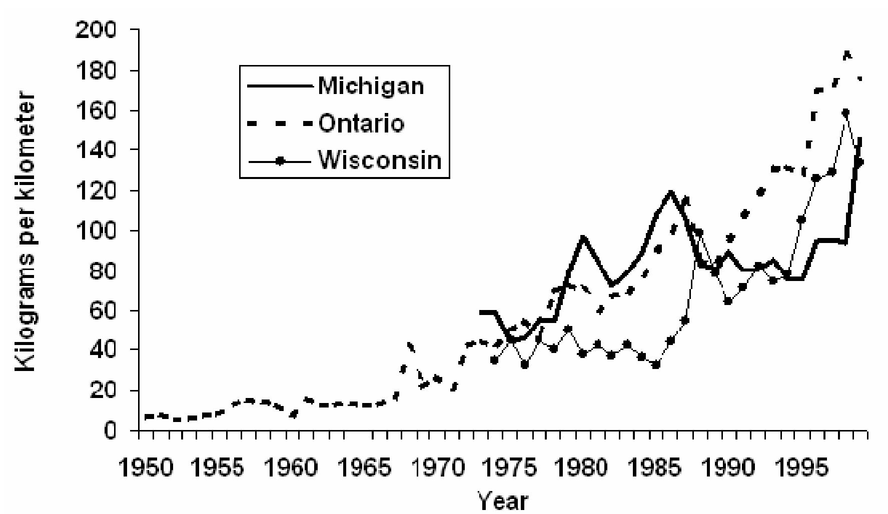

Fig. 17. Catch per unit effort of lake whitefish expressed as kg of fish per km of large-mesh equivalent gillnet effort in Ontario, Michigan, and Wisconsin waters of Lake Superior, 1950-1999.

Mortality and Growth

Total annual mortality, estimated from catch curves for whitefish caught in commercial large-mesh gillnets in Ontario and Michigan waters during 1998-2000, varied from a low of 30% in MI-7 to a high of 75% in MI-5 (Fig. 1; Table 5). Mortality rates in all management units, with the exception of MI-5 and MI-3, were below the 65% guideline even though they were likely biased high due to gillnet selectivity. Whitefish tend to be resilient to exploitation and can sustain annual mortality rates as high as 65% (Healey 1975). Compensation for higher mortality occurs primarily through a decrease in the average age-at-maturity (Healey 1980; Jensen 1985), increased fecundity and growth (Healey 1978, 1980; Liu and Jensen 1993), and increased recruitment (Healey 1980; Jensen 1985).

Table 5. Total annual mortality and mean age of lake whitefish caught in large-mesh commercial gillnets (>114 mm) from Michigan and Ontario management units in Lake Superior. Mortality estimates were calculated by linear regression of catch at age for six ages following the modal age. Estimates were only included where precision was within 10% of the 95% confidence interval (CI).

Unit	Year	Number Aged	Range of Ages	Mean Age	Annual Mortality (%)	95% CI
MI-2	1998	293	6-16	9.9	55	55-56
	1999	177	5-20	9.6	52	50-54
	2000	850	5-18	8.5	57	56-58
MI-3	1998	870	5-13	8.7	65	63-66
	1999	1,422	5-15	8.9	57	55-58
	2000	913	5-20	8.6	68	67-69
MI-4	1998	797	5-13	7.6	59	58-60
	1999	669	4-14	8.0	53	53-54
	2000	940	5-13	8.1	63	62-63
MI-5	1998	326	5-19	8.0	52	51-53
	1999	288	5-23	7.9	52	50-55
	2000	178	4-11	7.5	75	73-77
MI-6	2000	256	4-18	7.7	35	33-37
MI-7	1998	857	4-18	7.6	39	37-40
	1999	557	4-17	7.0	45	43-46
	2000	294	3-17	7.5	30	27-32
MI-8	1999	400	4-15	7.7	45	43-47
	2000	230	4-13	7.3	49	46-52
ON-31	1998	395	5-15	7.5	50	48-52
	1999	323	4-19	8.0	59	55-62
ON-33	1998	350	4-17	8.0	49	49-49
ON-34	1998	351	4-15	7.9	61	58-63
	1999	229	4-18	8.5	50	47-53

The modal age of whitefish in the lakewide commercial harvest has increased since 1975. Generally, higher population-density areas, as indicated by CPUE or biomass, contain older-aged fish than those with lower CPUE or biomass. Mean age varied from a low of 6.8 years in ON-7 to a high of 11.5 years in ON-1 during 1998-2000 (Table 5). The age at full vulnerability to commercial nets is between ages 7 and 8 for most of the lake. Mean TL of age-7 whitefish decreased from 1974 to 1990 and increased during the 1990s, in response to changes in density (Bence and Ebener 2002).

Recommendations

- Develop population simulation models that estimate harvest rates
- Map spawning and nursery areas to better protect them from future encroachments
- Restore river-spawning populations by ameliorating the impacts of hydroelectric-peaking operations
- Establish fishery-independent surveys

WALLEYE

Stephen T. Schram
Wisconsin Department of Natural Resources
141 South 3rd Street, Box 589
Bayfield, Wisconsin, U.S.A. 54814

…maintain, enhance, and rehabilitate self-sustaining populations of walleye and their habitat over their historical range.

In response to the above objective (Horns et al. 2003), a rehabilitation plan that describes objectives, issues, and strategies was developed (Hoff 2003). Historically, the walleye populations of Lake Superior have been relatively small and widely scattered, primarily because shallow, cool-water habitat is limited and patchy (Schneider and Leach 1979). Fisheries for walleye are associated with large bays, estuaries, and rivers, and overfishing has been identified as the primary reason for lower walleye abundance (Schram et al. 1991). Maximum commercial harvests were 56,000 kg from Minnesota waters in 1885 and 170,000 kg from Ontario waters in 1966 (Baldwin et al. 1979). The Ontario fishery has declined drastically due to overexploitation in the Black and Nipigon Bays (Schram et al. 1991). The state-licensed commercial fishery for walleye has been closed in U.S. waters since 1955, but Native Americans in the U.S. and First Nations in Canada are allowed by treaty to harvest walleye commercially.

Walleye currently spawn in 32 areas around Lake Superior, although biological data for many of these spawning populations are nil or scant. Spawning areas are primarily in tributaries, although estuaries and large bays are also used. Low numbers of walleye are found in association with river mouths along the entire shore. Historically, the largest populations were found in Ontario's Black Bay and the St. Louis River. Populations in Black Bay are currently low with small resident populations remaining only in tributaries. These resident populations may be important sources of remnant genotypes for rehabilitation efforts. Adult walleye were transferred from Ontario inland lakes to Nipigon and Black Bays to increase reproduction, but the results were inconclusive. Current strategies to enhance natural reproduction include no harvest of walleye in Nipigon Bay and the north end of Black Bay and construction of a spawning reef in Thunder Bay at the mouth of the Current River. In Wisconsin, self-sustaining populations have been rehabilitated in the St. Louis River (Schram et al. 1992), the Kakagon

Slough, and the Bad River. About 100,000 walleye fry are stocked annually into the Kakagon Slough and Bad River, but the contribution of these fish to these populations is unknown. In Michigan, some populations are maintained by stocking fingerlings—annually in Huron and Waishkey Bays; in alternate years in Portage Lake, Lac La Belle, and the Tahquamenon River; and sporadically in the Ontonagon River.

Walleye abundance is still below historical levels in Ontario and Michigan waters but is near historical levels in some Wisconsin waters. Rehabilitation has been impeded by loss or degradation of habitat due to sedimentation caused by human encroachment along streams, estuaries, coastal wetlands (e.g., commercial and residential development, logging, and agricultural practices), and hydroelectric dams. Initiating rehabilitation has been impeded by a lack of biological data for many populations. The available data indicate highly variable recruitment, slow growth, late maturation, and an age structure skewed toward older individuals. These characteristics make Lake Superior populations less able than other Great Lakes populations to withstand high levels of exploitation. Overexploitation was blamed when the abundance of walleye in Black Bay declined during the 1960s, and the fishery eventually collapsed (Colby and Nepszy 1981; Schram et al. 1991). However, habitat loss also could have been a factor, because a hydroelectric facility was constructed on a major tributary to Black Bay just prior to the population collapse. Strong year classes in Lake Superior are often separated by as much as ten years. This highly variable recruitment was observed in the St. Louis River in 1981 where one year class represented 50% of the spawning population, fish were as old as age 22, some 63% of the mature males and 78% of the mature females were age ten or older, and total annual mortality for both sexes was 42% (Schram et al. 1992). There had been little or no fishery exploitation of this population for at least 80 years, because industrial discharges tainted the fish.

Recommendations

- Collect biological data necessary for rehabilitation of important populations
- Within watersheds, implement forestry and agricultural practices that protect walleye habitat

LAKE STURGEON

Henry R. Quinlan
U.S. Fish and Wildlife Service
Ashland Fishery Resources Office
2800 Lake Shore Drive East
Ashland, Wisconsin, U.S.A. 54806

…rehabilitate and maintain spawning populations of lake sturgeon that are self-sustaining throughout their native range.

Lake sturgeon rehabilitation efforts reflect the above goal (Horns et al. 2003) and are guided by a Lake Sturgeon Rehabilitation Plan (Auer 2003). Historical and current information indicates that at least 21 Lake Superior tributaries supported spawning populations of lake sturgeon (Harkness and Dymond 1961; Auer 1999). Lake sturgeon currently reproduce in ten tributaries that include the Sturgeon River, Michigan; Bad River, Wisconsin; and the Kaministiquia, Black Sturgeon, Nipigon, Gravel, Pic, Michipicoten, Batchawana, and Goulais Rivers in Ontario (Fig. 1). Lake sturgeon also inhabit Lake Nipigon, Ontario, and the upper St. Louis River, Minnesota, but hydroelectric facilities block upstream access from Lake Superior.

Targets for rehabilitation of lake sturgeon in the 21 historical spawning tributaries were developed in part from information on persistent spawning populations in the Sturgeon and Bad Rivers, which have been assessed since the 1980s. These targets are to achieve spawning populations of at least 1,500 adult fish that are made up of 20 or more year classes, exhibit a male to female sex ratio of 1:1, and produce young every year resulting in recruits of ages 0-5. Sex ratios in spawning runs in the Sturgeon and Bad Rivers in recent years were 1.25-2.7:1 and 2:1, respectively (Auer 1999). Quantitative information on suitable and available spawning, rearing, and nursery habitat currently is not available for most of the 21 tributaries, but mean annual discharge in most of them is greater than in the Sturgeon and Bad Rivers, indicating that they are likely capable of supporting spawning populations >1,500 fish.

In the ten tributaries that currently support lake sturgeon populations, abundance is below historical levels and none meet all rehabilitation targets. The number of adults in annual spawning runs was 200-375 in the Sturgeon River (Hay-Chmielewski and Whelan 1997), 200-350 in the Bad River in 1997 and 1998 (HRQ, unpubl. data), and 140 in the Kaministiquia River

(Stephenson 1998). Hay-Chmielewski and Whelan (1997), using harvest data from Baldwin et al. (1979), estimated that abundance in Lake Superior in the 1880s was roughly 870,000 individuals of all ages. Even if the rehabilitation target of 1,500 adults was met in all 21 historical tributaries, the contemporary population of adult fish would only number 31,500.

The abundance of juvenile lake sturgeon during June and September near the mouth of the Bad River, indexed as CPUE in gillnets, increased more than fivefold between 1991 and 1999 (W.P. Mattes, Great Lakes Indian Fish and Wildlife Commission, P.O. Box 9, Odanah, WI, 54861, unpubl. data). Radio telemetry studies suggest that a resident population inhabits the Kaministiquia River (M. Friday, Ontario Ministry of Natural Resources, Upper Great Lakes Management Unit—Lake Superior, 435 James St. South, Thunder Bay, Ontario, Canada, P7E 6S8, personal communication).

Lake sturgeon populations in each of the upper Great Lakes are genetically distinct. Based on a higher degree of gene diversity within than between the Sturgeon River and Bad River populations, DeHaan (2003) concluded that the two populations are distinct. That the two populations are genetically distinct is corroborated by the presence of a haplotype unique to the Bad River population and a high fidelity of adults to spawning populations in each tributary—only two of 1,200 fish tagged in one of these two tributaries were recaptured in another stream.

Lake sturgeon have been stocked in the lower St. Louis River (approximately 780,000 fry and 142,700 fingerlings during 1983-2000) and Ontonagon River (approximately 16,500 fingerlings during 1998-2000) to rehabilitate extirpated populations (Schram et al. 1999; J. Lindgren, Minnesota Department of Natural Resources, 5351 Northshore Drive, Duluth, MN, 55804, personal communication). To supplement existing populations, 24,000 fry and 17,800 fingerlings were stocked in the Bad River in 1988-2000, and 700 fingerlings were stocked in the Sturgeon River in 1998.

Hydroelectric facilities that block stream access are the single greatest impediment to lake sturgeon rehabilitation in Lake Superior, and 12 of the 21 historically used spawning tributaries are blocked. These facilities limit lake sturgeon production by dewatering spawning and rearing habitat, altering flow and water-temperature regimes, and blocking access to spawning areas. On the Kaministiquia River, Ontario, water is frequently diverted (via bypasses) around lake sturgeon-spawning habitat to turbines during the spawning run, causing mortality of adult sturgeon and their eggs (M. Friday, Ontario Ministry of Natural Resources, Upper Great Lakes

Management Unit—Lake Superior, 435 James St. South, Thunder Bay, Ontario, P7E 6S8, personal communication). Were dewatering minimized or eliminated with a run-of-the-river flow, the number and size of lake sturgeon in spawning runs would likely increase (Auer 1996).

Commercial fishing for lake sturgeon is prohibited in Lake Superior by most agencies, and recreational and subsistence harvest is managed on an agency-by-agency basis. Harvest data from Lake Winnebago, Wisconsin, suggests that exploitation rates should not exceed 5% to maintain a self-sustaining population (Bruch 1999). Until specific data are available for Lake Superior populations, agencies have expressed support for harvest regulations that maintain annual exploitation rates below 5% (Auer 2003).

Recommendations

- Describe and quantify lake sturgeon spawning and juvenile nursery habitat
- Identify tributary-specific flow regimes that are compatible with lake sturgeon rehabilitation, and develop agreements with hydroelectric facilities to maintain required flows during the spawning season
- Monitor harvest to determine population-specific exploitation rates
- Conduct standardized surveys to monitor relative abundance and life history parameters

BROOK TROUT

Dennis M. Pratt
Wisconsin Department of Natural Resources
1401 Tower Avenue
Superior, Wisconsin, U.S.A. 54880

> *…maintain widely distributed, self-sustaining populations in as many of the historical habitats as is practical.*

In response to the above objective (Horns et al. 2003), a rehabilitation plan detailing objectives, issues, and strategies was developed (Newman et al. 2003). The plan benefited from a prior status report by Newman and Dubois (1997). Brook trout were common historically in many of Lake Superior's spring-fed tributaries (Moore and Braem 1965), and larger brook trout inhabited the rocky coastlines. Brook trout that spend at least a portion of their life in the nearshore lake environment grow faster and reach larger sizes and are commonly referred to as coasters (Becker 1983). Coaster brook trout were formerly associated with at least 118 tributary streams, but few coaster populations are currently present in the lake (Newman and Dubois 1997; Newman et al. 2003). Spawning locations are primarily associated with tributaries, although a U.S. Fish and Wildlife Service study documented a shoal-spawning population in Tobin Harbor of Isle Royale (HRQ, unpubl. data). The most well-known coaster brook trout population is found in the Nipigon Bay region of Ontario (Fig. 1), and it remains one of the most-robust populations. Other significant coaster brook trout populations are found at Isle Royale and in the Salmon-Trout River, Michigan. Observations of coaster brook trout have been reported rarely in Wisconsin and Minnesota waters (Schreiner 1995; D. Pratt, unpubl. management report). It is uncertain whether coasters are a unique strain of brook trout with migratory tendencies and reproductively isolated from stream-resident brook trout, or whether they are progeny of stream-resident brook trout that migrate to the lake, either by chance or because of high in-stream population densities, and upon maturing, breed freely with stream-resident fish. D'Amelio (2002) suggested that coaster brook trout are produced by stream-resident brook trout populations and may be an ecological variant rather than a distinct genetic group.

Agencies around Lake Superior have long been concerned about brook trout populations. Responses to declining abundance of both stream-resident and coaster brook trout, caused by overharvest and stream habitat destruction in the 1870s and 1880s, included enacting commercial and recreational fishery regulations and stocking. Much of the recent interest in rehabilitation of brook trout populations has been directed toward the coaster life-history variant. To protect coaster brook trout, many fishery agencies have enacted more-stringent angling regulations, including restricting harvest to a single fish 50 cm (20 inches) and larger or no harvest in some areas. Stocking to reestablish coaster populations has been undertaken using strains that originated from Lake Nipigon, Ontario, and Isle Royale, Michigan, but results are pending.

Recommendations

- Protect existing habitats and rehabilitate impaired tributary habitats
- Stock or introduce only strains of brook trout originating from within the basin
- Monitor abundance, growth, recruitment, and harvest from each population
- Focus research on identifying impediments to rehabilitation, including critical habitats for each life stage; on genetic, behavioral, and morphological traits; and on community interactions
- Establish populations composed of at least six year classes of which two contain spawning females

NON-INDIGENOUS SALMONINES

Donald R. Schreiner[1], Stephen T. Schram[2], Shawn Sitar[3], and Mike Petzold[4]

> *…manage populations of Pacific salmon, rainbow trout, and brown trout that are predominately self-sustaining but may be supplemented by stocking that is compatible with restoration and management goals established for indigenous fish species.*

The above objective (Horns et al. 2003) recognizes that most of the introduced salmonines in Lake Superior have naturalized and are a permanent component of the fish community. Furthermore, it acknowledges the potential risk to rehabilitation of indigenous species by continued supplemental stocking of non-indigenous species. The balance between indigenous and non-indigenous species abundance is extensively debated among agencies and between agencies and their constituents. An acceptable balance that considers biological limitations as well as social and economic needs is continually being formulated.

The introduction and widespread naturalization of non-indigenous salmonines have significantly expanded sport-fishing opportunities, especially in tributaries and nearshore waters. Since most of the introduced salmonines are now naturalized, agencies are reevaluating the cost-effectiveness of their stocking programs (Peck 1992; Schreiner 1995; Schreiner and Schram 1997).

[1]Minnesota Department of Natural Resources, Lake Superior Area Fisheries Program, 5351 North Shore Drive, Duluth, Minnesota, U.S.A., 55804

[2]Wisconsin Department of Natural Resources, 141 South 3rd Street, Box 589, Bayfield, Wisconsin, U.S.A., 54814

[3]Michigan Department of Natural Resources, 484 Cherry Creek Road, Marquette, Michigan, U.S.A., 49855

[4]Ontario Ministry of Natural Resources, 1235 Queens Street East, Sault Ste. Marie, Ontario, Canada, P6A 2E5

All agencies now monitor, to varying extents, the relative abundance of non-indigenous salmonines and the contribution of stocked fish to fisheries. Assessing angler catch and catch rate in the recreational fishery (creel survey) is the primary monitoring technique. Angler catch and catch rates presented herein are from annual creel surveys of the summer (May-September) open-water lake fishery. Spawning migrations of non-indigenous salmonines in the Brule River in Wisconsin, the French and Knife Rivers in Minnesota, and the McIntyre River in Ontario are monitored using permanent traps or counting stations. Some electrofishing surveys have been done in tributaries to assess spawning success and juvenile production by naturalized salmonines.

Rainbow Trout (Steelhead)

Migratory rainbow trout, or steelhead, became naturalized soon after introduction in 1895 (MacCrimmon 1971). Since about 1970, additional strains have been introduced to supplement naturalized stocks in areas with intense fisheries or limited spawning habitat (Peck et al. 1994). All agencies have stocked various life stages of rainbow trout, and yearlings are the preferred life stage (Table 2). The percentage of stocked rainbow trout that is caught by anglers has generally averaged 1% or less. However, Close and Hassinger (1981) reported percentage-return to anglers of 3-28% for the Kamloops, Donaldson, and Madison strains stocked in Minnesota waters, and Peck (1992) reported that anglers caught 1-2% of the Siletz strain stocked in a Michigan tributary. The contribution of stocked fish to rainbow trout populations and fisheries has varied widely among streams, but significant contributions to fisheries occur only when the numbers stocked are substantial and the numbers of wild fish are small (Peck 1992). Stocking to supplement wild populations usually is inefficient and may pose genetic risks to wild populations (Krueger and May 1987a; Krueger et al. 1994).

Abundance of rainbow trout in western waters, as monitored in the Brule River, Wisconsin, and French River, Minnesota, declined in the mid-1990s (Fig. 18) due to extremely cold lake temperatures during 1992-1993, which caused poor growth and possibly low survival of the year classes emigrating as juveniles from streams. Abundance increased during the late 1990s, likely because of relatively warm lake temperatures and restrictive harvest regulations.

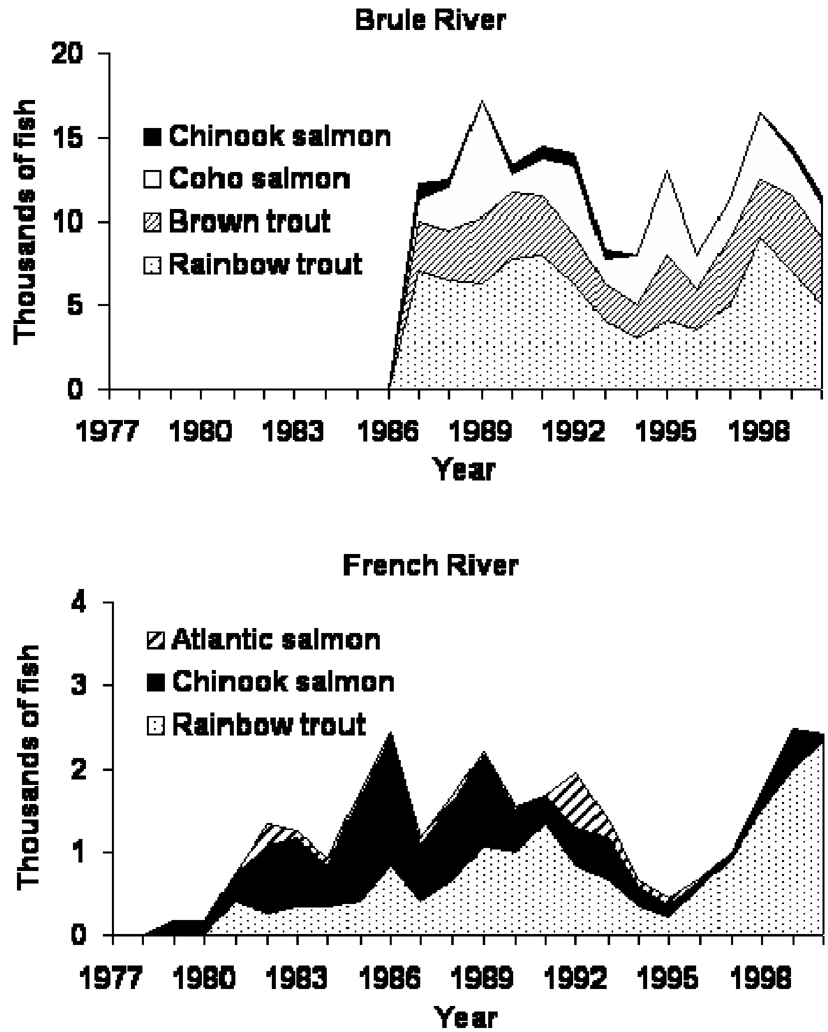

Fig. 18. Number of adult trout and salmon observed in the spawning runs at the Brule River, Wisconsin, fishway (top panel), 1986-2000 and French River, Minnesota, traps (bottom panel), 1977-2000.

Coho Salmon

Coho salmon were stocked by Michigan in 1966 and by Minnesota and Ontario during 1969-1972 (Table 2) and quickly became naturalized throughout Lake Superior. Coho salmon reproduce successfully in most tributaries that are accessible during the spawning period, have suitable substrate, and maintain adequate winter groundwater flows. Michigan discontinued stocking coho salmon in 1994, because stocked fish comprised <10% of the coho salmon catch in the recreational fishery (Peck 1992). Anglers exploit almost entirely age-2 fish, which results in wide harvest fluctuations dictated by variations in year-class strength. In most years, coho salmon is the second most-harvested species, after lake trout, in the U.S. sport fishery. Total harvest since the 1970s has fluctuated greatly in U.S. waters with slightly higher numbers being caught in Michigan than in Wisconsin and Minnesota (Fig. 19). In general, a positive relationship appears to exist between catch rates in the lake and returns to the Brule River, Wisconsin, which is understandable since both the fishery and spawning run depend on a single year class. Since stocking no longer contributes substantially to coho salmon populations, more-restrictive harvest regulations were enacted in most jurisdictions during the 1990s to provide for an adequate number of spawning wild fish.

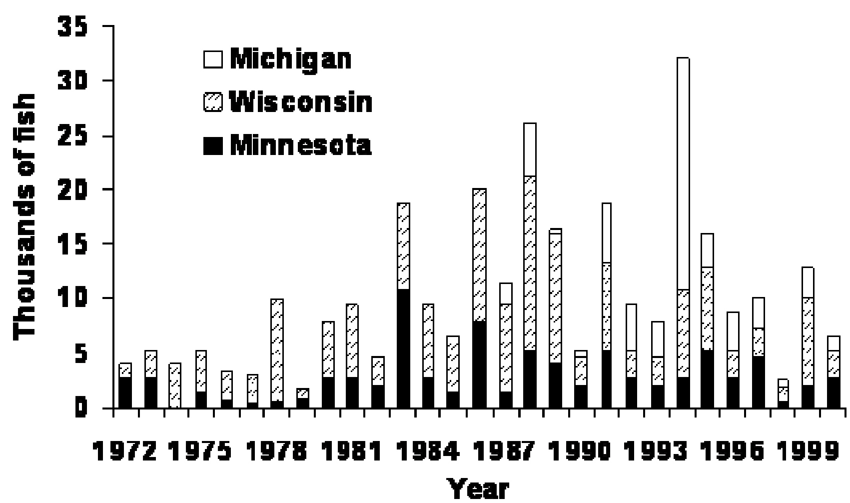

Fig. 19. Number of coho salmon caught annually in the summer (May-September) open-water recreational fishery in Michigan, Wisconsin, and Minnesota waters of Lake Superior, 1972-2000.

Chinook Salmon

Contemporary Chinook salmon stocking into Lake Superior was initiated by Michigan in 1967, followed by Minnesota in 1974, Wisconsin in 1977, and Ontario in 1988 (Table 2). Chinook salmon were stocked previously in 1874-1875, but no naturalized populations were established (Parsons 1973). Presently, all agencies stock Chinook salmon as fingerlings in the spring. The utility of continued stocking is questioned because most Chinook salmon in Lake Superior are naturally reproduced. In a lakewide study of the sport fishery, over 75% of the Chinook salmon harvested were naturally reproduced (Peck et al. 1999). Stocked Chinook salmon made up 57% of the angler harvest in Minnesota, 32% in Wisconsin, 25% in Michigan, and 9% in Ontario during 1990-1994. Chinook salmon stocked in each jurisdiction contributed to the fisheries in all other jurisdictions, indicating that they move considerable distances and have little stocking-site affinity during the summer angling season. Stocking programs by most agencies are being reevaluated to determine if they are cost effective.

Chinook salmon have been observed spawning in numerous streams in the U.S. and in most of the larger rivers in Ontario (Peck et al. 1994). The numbers of naturalized Chinook salmon in spawning runs in the Brule River during 1987-2000 were highest in 1987 and 1990-1992, were low during 1993-1998, then increased during 1998-2000 (Fig. 18). There has been a dramatic decline in the numbers of stocked fish returning to brood-fish collection sites at the French River in Minnesota during the 1990s (Fig. 18) and at Pikes Creek in Wisconsin, but harvest has held up well because it is largely sustained by natural reproduction (Fig. 20). Among jurisdictions, the Chinook salmon harvest is usually greatest in Minnesota waters.

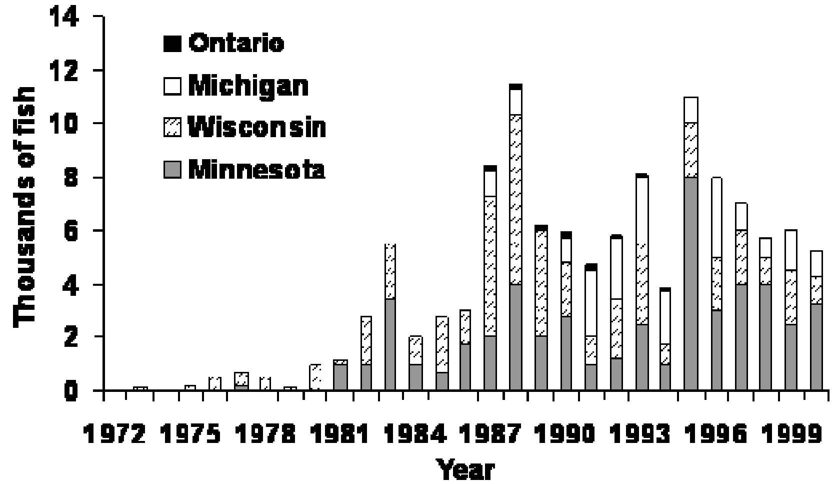

Fig. 20. Number of Chinook salmon caught annually in the summer (May-September) open-water recreational fishery in Michigan, Wisconsin, Minnesota, and Ontario waters of Lake Superior, 1972-2000.

Pink Salmon

In 1956, approximately 21,000 pink salmon fry were accidentally introduced into the Current River in Ontario (Nunan 1967), and pink salmon are now naturalized in Lake Superior. In general, pink salmon abundance increased from the 1960s through the 1970s, then declined to extremely low levels during the late 1980s (Peck et al. 1994). From 1984-1995, the annual catch by lake anglers from each of the three states' waters has averaged <200. However, abundance appears to have increased since the mid-1990s, as indicated by increased harvest and reports of increased numbers of spawning adults in tributaries. In Ontario, the angler catch is unknown, but pink salmon spawn in a number of tributaries and have been harvested by a dipnet fishery in the Michipicoten River (Peck et al. 1994). Spawning runs initially were only in odd years, but delayed maturity resulted in some fish spawning as 3 year olds so even-year runs have established.

Brown Trout

Brown trout were introduced into Lake Superior in the late 1890s and naturalized populations spawn in a number of tributaries, especially in Wisconsin waters (Lawrie and Rahrer 1972). Brown trout are stocked in Wisconsin and Michigan tributaries (Table 2) where they support a few tributary spawning populations and sport fisheries, such as the winter fishery in Chequamegon Bay. Hatchery-reared brown trout made up 50% of the sport-fishery harvest of brown trout in Wisconsin and 40% in Michigan prior to 1994 (Peck et al. 1994). The sport-fishery harvest of brown trout is highest in Wisconsin waters where it has ranged from approximately 5,000 in 1975 to 534 in 1996. The reported harvest of brown trout from all jurisdictions during 1972-2000 averaged about 1,000 fish. The Brule River, Wisconsin, supports the largest known spawning run of brown trout in Lake Superior; the run averaged 3,625 during 1987-1999 (Fig. 18). Significant genetic variation exists among naturalized brown trout populations from different drainages in Wisconsin and between migratory and resident brown trout in the Brule and Sioux Rivers, Wisconsin (Krueger and May 1987b).

Atlantic Salmon

Presently, there is no management program for Atlantic salmon in Lake Superior. Each of the states experimented with stocking Atlantic salmon sometime during 1972-1992 (Table 2). Catch and catch rate of Atlantic salmon in the summer sport fishery was relatively low—the average lakewide catch was 300 fish during 1983-1996, which came mostly from Minnesota and Wisconsin waters. Minnesota, in 1982, developed the most-extensive stocking program but discontinued it in 1992 because of low returns, high cost, and limited angler interest (Schreiner 1995). Wisconsin and Michigan discontinued their Atlantic salmon stocking for the same reasons. There is no evidence that Atlantic salmon have naturalized in Lake Superior.

Splake

Splake are hybrids created in the hatchery from the fertilization of lake trout eggs with brook trout sperm. They were first stocked into Lake Superior by Michigan in 1971, and Wisconsin followed in 1973. No natural reproduction by splake has been documented in Lake Superior, but sexually mature splake have been found in spawning aggregations of lake trout (Peck et al. 1994). Harvest of splake occurs primarily near the stocking sites, and, hence, splake are valued by management agencies for development of localized nearshore fisheries. Wisconsin has developed a major splake fishery in Chequamegon Bay that accounts for much of the harvest in Lake Superior, but, in some years, a substantial number are also caught in Michigan waters near Marquette and Munising. The lakewide recreational harvest of splake during May-September ranged from near zero to 1,500 during 1972-2000.

Non-indigenous salmonines play a relatively minor role in the Lake Superior fish community (Kitchell et al. 2000). Thermal requirements relegate these species to the nearshore and offshore pelagic portions of the lake. However, non-indigenous salmonines have the potential to compete with brook trout for spawning and nursery habitat in the nearshore zone and in tributaries. Because non-indigenous salmonines have naturalized and are of great social and economic importance to the sport fishery, a better understanding of their role in shaping the Lake Superior fish community is important.

Recommendations

- Determine if non-indigenous salmonines are having a negative effect on stream-resident and nearshore populations of brook trout
- Inform the public of the cost effectiveness of all stocking programs
- Apply the Great Lakes Fish Health Committee protocol and standards to all stocking programs
- Develop a database for sport-fish harvests

SEA LAMPREYS

Michael Fodale[1] and Douglas Cuddy[2]

> *…suppress sea lampreys to population levels that cause only insignificant mortality on adult lake trout.*

Insignificant mortality is defined in the above objective as <5% annually (Horns et al. 2003). Previously, in 1992, the FCO was to achieve by the year 2000 a 50% reduction from then-current levels of sea lamprey abundance and by 2010 a 90% reduction (Busiahn 1990). The current level referred to in 1992 was average abundance during 1986-1989. As of 2000, the reduction target of 50% has not been met, and the new objective of insignificant mortality on lake trout is even more distant.

Notable changes have occurred within the sea lamprey control program since 1992, the last year of record in the previous state-of-the-lake report (Hansen 1994). Lake Superior streams needing treatment are now ranked against all other such streams in the Great Lakes basin (Christie et al. 2003). Concentrations of TFM in lampricide applications have been reduced to protect sensitive nontarget species and to achieve the GLFC's goal of reducing lampricide use. During the 1990s, low-head barrier dams and sterile-male release programs were tested and employed as an alternative to lampricide treatment. Determining appropriate locations and construction of low-head barriers has progressed with development of two barriers on Canadian tributaries: a low-head dam with an inflatable crest on the Big Carp River near Sault Ste. Marie and an experimental velocity-barrier on the McIntyre River near Thunder Bay. In 1997, the sterile-male release program was shifted from Lake Superior to the St. Marys River. The sterile-male release program had no measurable effect on Lake Superior sea lamprey populations, because the effort was too small (Twohey et al. 2003), but it did reduce larval populations (Bergstedt et al. 2003).

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[1]U.S. Fish and Wildlife Service, 1924 Industrial Parkway, Marquette, Michigan, U.S.A., 49855

[2]Dept. of Fisheries and Oceans, 1 Canal Drive, Sault Ste. Marie, Ontario, Canada P6A 6W4

The average number of Lake Superior tributaries treated each year declined from 25 treated prior to 1990 to 16 treated during 1990-1999, a decline of 36%. Also, the amount of TFM applied per cubic meter of stream treated declined during this same period. The amount of high-quality larval habitat treated annually with TFM increased during the past decade, despite treatment of fewer streams (Heinrich et al. 2003).

The number of adult spawning-phase sea lampreys in Lake Superior has slowly declined since 1962 (Fig. 21), but their abundance is still too high because the management objective of sea lamprey-induced mortality of <5% is not being met. Changes in the program that likely contributed to the failure to meet this objective include use of lower concentrations of TFM, discontinuation of sterile-male release, lags in construction of new sea lamprey barriers, and regulatory and permitting constraints on lampricide use. The increased concern to protect nontarget and especially threatened species from adverse effects of TFM application has necessitated changes to lampricide-treatment protocols that likely result in increased numbers of sea lamprey larvae surviving treatment. Restricted windows for applications, reductions in the TFM concentrations, and additional application points on streams (to protect lake sturgeon) all contribute to reduced treatment effectiveness.

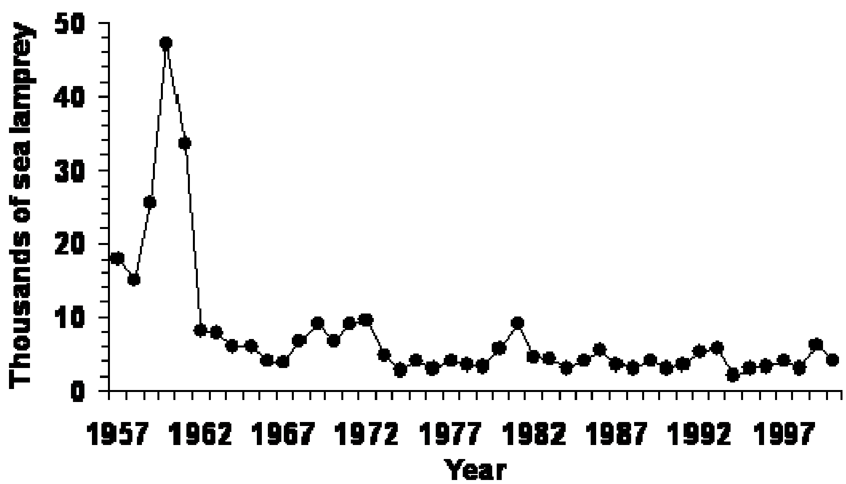

Fig. 21. Estimated abundance of spawning-phase sea lampreys in Lake Superior during 1957-2000 (Heinrich et al. 2003).

Recent assessments indicate that residual larvae that survive treatments contribute more to the adult population in the lake than to larvae produced in untreated lentic areas. Furthermore, a new model predicts that streams just treated can still contain large numbers of residual larvae that can result in more transformers within two years of treatment than the expected number of transformers that would have been produced without treatment (Heinrich et al. 2003). Thus, more-frequent treatment of streams that harbor large numbers of larvae should significantly reduce the adult populations in the lake.

Recommendations

- Increase the frequency of lampricide treatments on streams that consistently produce large numbers of sea lamprey larvae
- Determine how larvae survive treatments and use this knowledge to better eliminate residual larvae
- Construct sea lamprey barriers in all streams that meet barrier location criteria
- Determine the origins of the adult populations of sea lampreys in the lake
- Implement new control technologies such as that based on pheromones

NUISANCE SPECIES

Mark P. Dryer[1], Gary D. Czypinski[1], and Douglas A. Jensen[2]

> *…prevent the introduction of any non-indigenous aquatic species that is not currently established in Lake Superior, prevent or delay the spread of non-indigenous aquatic species, where feasible, and eliminate or reduce populations of non-indigenous nuisance species, where feasible.*

Notwithstanding the above objective (Horns et al. 2003), the rate of introduction has increased in recent years, and at least 39 new non-indigenous aquatic species have entered the lake since 1970 (Bronte et al. 2003). Lake Superior has the highest percentage of non-indigenous to indigenous fish species (24%) of all the Great Lakes (Mills et al. 1993). However, most of the fishes unintentionally introduced into Lake Superior are less common and occupy reduced ranges in comparison to their counterparts in the other Great Lakes. Pathways for accidental introduction of non-indigenous species into Lake Superior and the other Great Lakes (which are pathways) include ballast-water discharge, hull fouling, water diversions and canals, recreational watercraft, live-bait release by anglers, aquarium release, and aquaculture activities. The entry pathway for all of the non-indigenous fishes and three of four non-indigenous invertebrates found in Lake Superior since 1970 has been attributed to inter-lake movement of ocean-going cargo ships (Bronte et al. 2003). Most of these species were discovered in the Duluth-Superior Harbor of the St. Louis River estuary. Duluth-Superior is the busiest inland port in the U.S. with more than 1,000 vessels visiting annually, thus putting Lake Superior continually at risk for introductions of non-indigenous species.

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[1]U.S. Fish and Wildlife Service, 2800 Lake Shore Drive East, Ashland, Wisconsin, U.S.A. 54806

[2]University of Minnesota Sea Grant College Program, 2305 East Fifth Street, Duluth, Minnesota, U.S.A., 55812

The effects of recent aquatic nuisance species on the Lake Superior food web are poorly understood, but these species could pose a threat to the lake's biodiversity, especially in nearshore areas where most introductions originate (Mills et al. 1993; Leach et al. 1999). Moreover, since the rate of introduction is increasing, the threat is growing. The following history and status is presented for six fishes and four invertebrates unintentionally introduced since 1970.

Fourspine Stickleback

The fourspine stickleback is indigenous to Europe and the Atlantic coast and was initially collected in Lake Superior from Thunder Bay Harbor, Ontario, in 1987. It is not known from any other location in the Great Lakes. Threats and impacts from fourspine stickleback are largely unknown. However, the abundance of indigenous sticklebacks has declined concurrent with the fourspine and threespine stickleback infestations in Thunder Bay (Stephenson and Momot 2000).

Round Goby

The round goby is a small bottom-dwelling fish indigenous to the Black and Caspian Seas. The round goby was first detected in the St. Clair River in 1990 and, by 1995, it had spread to four of the five Great Lakes. In Lake Superior, the round goby is found only in the Duluth-Superior Harbor, and mean densities there, estimated from bottom-trawl surveys, had increased to 25 ha^{-1} by 2000 (OTG, personal communication). In the Great Lakes, round gobies are most abundant on nearshore, rocky habitat and feed on insect larvae, zebra mussels, and eggs and fry of indigenous fish. Round gobies often outcompete native sculpins for food and habitat. In Lake Superior, round gobies will likely invade other ports by natural expansion of the Duluth-Superior population along the rocky shoreline or via interlake or intralake ballast-water intake and discharge.

Ruffe

The ruffe is a small perch-like Eurasian fish that was first detected in the St. Louis River estuary at Duluth-Superior Harbor in 1986 (Pratt et al. 1992). Ruffe quickly became abundant in the St. Louis River, peaking at a mean density of approximately $2{,}000 \cdot ha^{-1}$ in 1995 and stabilizing at approximately $1{,}000 \cdot ha^{-1}$ by 2000. By 1999, ruffe had migrated west to the Amnicon, Iron, and Flag Rivers in Wisconsin and the Ontonagon River in Michigan. Populations in these rivers have been monitored since 1995. Ruffe were discovered in Thunder Bay Harbor, Ontario, in 1991 and in Lake Huron at Alpena, Michigan, in 1995, and these populations likely originated from ballast water taken on at Duluth-Superior.

Decreased abundance of indigenous fishes in the St. Louis River estuary concurrent with increased ruffe abundance raised concerns about the ruffe's impact on the fish community (Busiahn 1997), but it was subsequently determined that many of these decreases were natural population fluctuations (Bronte et al. 1998). Likewise, it could not be demonstrated that yellow perch density in the Amnicon, Iron, and Flag Rivers was affected by increased ruffe abundance.

Walleye and northern pike were stocked in the St. Louis estuary from 1989 to 1994 to suppress the ruffe population, but these predators failed to reduce ruffe abundance (Mayo et al. 1998). A ruffe control program featuring population reduction, ballast-water management, fish-community management, public education, and bait-fish management was developed to prevent or delay the further spread of ruffe through the Great Lakes and to prevent their spread to inland lakes and watersheds (Busiahn 1997). This program may have been successful—at least its objectives have been met so far. Surveillance and monitoring to detect new invasions and to establish long-term baseline conditions are integral components of the control program.

Threespine Stickleback

The threespine stickleback, indigenous to Arctic and Atlantic coast drainages, is present in all the Great Lakes and was initially collected in Lake Superior from Thunder Bay Harbor, Ontario, in 1987. It is now found at low densities in other harbors and bays in Lake Superior west of a line from Marquette, Michigan, to Black Bay, Ontario. The threats and impacts of threespine stickleback on the indigenous fish community are unknown.

Tubenose Goby

The tubenose goby, like the round goby, is a small bottom-dwelling fish indigenous to the Black and Caspian Seas. The tubenose goby is present but not abundant in all five Great Lakes. It was first detected in the St. Louis River estuary in 2001 (LME, personal communication), but it has not been observed elsewhere in Lake Superior. In contrast to the round goby, the tubenose goby prefers well-vegetated habitat. The potential impacts of the tubenose goby are unknown.

White Perch

The white perch is indigenous to the Atlantic coast and its tributaries and it has established in all the Great Lakes. It was initially collected in Lake Superior from the St. Louis River estuary in 1986 and has subsequently spread east along the south shore to the Ontonagon River, Michigan. Eggs of other fish are an important diet component of white perch. They are known to overpopulate habitats and to hybridize in Lake Erie with white bass. In Lake Superior, white perch numbers are low and their impact on the fish community is unknown.

Asian Clam

The Asian clam *Corbicula fluminea* is indigenous to eastern Asia and Africa and was discovered in the St. Louis River estuary in 1999 (LME, personal communication). It is found in sandy and mud-bottomed streams, rivers, ponds, lakes, and canals. The Asian clam has an advantage over indigenous clams because it better tolerates anthropogenic activities and has an unusually high reproductive capacity and growth rate, which allows it to quickly adapt to disturbed environments. It can tolerate a wide range of water temperatures (McMahon 1983), but its range expansion is probably limited by Lake Superior's cold water.

Rusty Crayfish

The rusty crayfish *Orconectes rusticus*, indigenous to the Ohio River basin and eastern U.S. rivers, was discovered in the late 1980s in the Thunder Bay area and was found subsequently in the Pigeon River, Ontario (Momot 1996), and in Duluth-Superior Harbor. The rusty crayfish can be extremely abundant in inland lakes and rivers of the Lake Superior basin. It is more aggressive than indigenous crayfish, giving them a competitive advantage. They severely reduce aquatic plant abundance and diversity, which affects fish and other wildlife populations (Lodge et al. 2000).

Spiny Waterflea

The spiny waterflea *Bythotrephes longimanus*, a crustacean indigenous to Great Britain and northern Europe, was discovered initially in Lake Superior in 1987. It is a component of the zooplankton community in all the Great Lakes and reaches its greatest density in Lake Erie and its lowest density in Lake Superior. Its body is <12-mm long and it has a long, barbed tail spine that is an effective defense mechanism against predation. Females can produce up to ten offspring every 14 days during warm summers. Eggs produced under cooler conditions lie dormant until warmer conditions return. The spiny water flea competes with juvenile fish for zooplankton. It can affect angling because adults sometimes collect on fishing lines, downrigger cables, and eyelets of fishing rods. Anglers can spread them to inland waters if fishing equipment is contaminated with egg-laden females. Although females die when out of water, under certain conditions they produce eggs that resist drying and remain viable.

Zebra Mussel

The zebra mussel *Dreissena polymorpha* is indigenous to western Russia near the Caspian Sea and entered the Great Lakes in 1985 or 1986, likely via ballast water. It was found in all the Great Lakes by 1989. Zebra mussels are prolific and very tolerant to a wide range of environmental conditions (Nalepa and Schloesser 1993). They can compete with native species for food, kill native mussels, concentrate contaminants, clog the intake pipes of water facilities, and damage boats and motors. Zebra mussels have been found at nine locations in Lake Superior, but the only reproducing population known is in the St. Louis River estuary, where its abundance increased greatly in 1989. Lake Superior’s cold waters and deficiency of calcium have thus far prevented zebra mussels from becoming as abundant as they are in the other Great Lakes.

Recommendations

- Prevent further introduction of non-indigenous species via Great Lakes shipping through effective ballast-water management, including controls on interlake intake and discharge
- Develop a rapid-response plan that will prevent or contain new aquatic invasions
- Develop public information and education programs that will promote greater awareness of undesirable non-indigenous species and how they were introduced
- Institute a monitoring and surveillance program that leads to the early detection of new introductions and range expansions of existing species

SPECIES DIVERSITY

Stephen T. Schram[1], Theodore N. Halpern[2], and Timothy B. Johnson[3]

> *…protect and sustain the diverse community of indigenous fish species not specifically mentioned earlier (e.g., burbot, minnows, yellow perch, northern pike, and suckers). These species add to the richness of the fish community and should be recognized for their ecological importance, and cultural, social, and economic value.*

Of the so-called minor species at which the above objective (Horns et al. 2003) is directed, only one, the burbot, is routinely studied. Some sixty-odd minor species currently contribute to the diversity and stability of the Lake Superior fish community, but they are rarely assessed and their population status is largely unknown. Many (i.e., suckers and minnows) are captured incidentally during routine assessments for other species (i.e., lake trout and sea lamprey) and the number captured is often recorded. However, these data are seldom summarized or reported, and, when they are, they are secondary in agency databases and in-house reports.

Northern pike, yellow perch, and smallmouth bass have very localized populations in tributaries or embayments, and information for a few of these populations has been collected, summarized, and reported (Bronte et al. 1993; Bronte et al. 1998; Mayo et al. 1998). These three species are integral components of the fish community in places like the St. Louis River estuary, Chequamegon Bay, Black Bay, and Whitefish Bay. In the St. Louis River estuary and Chequamegon Bay, spottail shiners, trout-perch, emerald shiners, and logperch are common indigenous species (Bronte et al. 1998). Brown bullheads are also very common and abundant in the St. Louis River estuary and in a portion of Whitefish Bay (Mayo et al. 1998).

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[1]Wisconsin Department of Natural Resources, 141 South 3rd Street, Box 589, Bayfield, Wisconsin, U.S.A., 54814

[2]Minnesota Department of Natural Resources, 5351 North Shore Drive, Duluth, Minnesota, U.S.A., 55804

[3]Ontario Ministry of Natural Resources, R.R. #2, 320 Milo Road, Wheatley, Ontario, Canada, N0P 2P0

Indigenous shiners (Cyprinidae) may be the most-abundant fish in shallow-water environments (<2-m deep). Spottail and emerald shiners were the most abundant of 32 species of fish captured in daytime, shallow-water seine hauls in lower Whitefish Bay and the upper St. Marys River during 1997 and 1998 (MPE, personal communication). The catch also included white sucker, larval whitefish, bowfin, lake chub, mimic shiner, bluntnose minnow, rock bass, finescale dace, smallmouth bass, and creek chub.

Burbot

Burbot are indigenous Lake Superior piscivores that share similar habitat and food resources with the lake's other indigenous predator, the lake trout. Burbot abundance, like that of lake trout, declined due to sea lamprey predation (Smith 1968; Lawrie and Rahrer 1973; Smith and Tibbles 1980) and then increased subsequent to initiation of sea lamprey control. Spatially, burbot inhabit all waters of Lake Superior from tributaries during spawning (Schram 2000) to depths of at least 366 m (Boyer et al. 1989). Burbot have never been economically important to the Lake Superior fishery, and, consequently, their ecological role as a top predator remains poorly understood, because fishery agencies have focused past research efforts mostly on trout, salmon, and forage species. Lack of information on the food habits of burbot was a concern raised in recent bioenergetics analyses (Negus 1995; MPE, personal communication).

Relative abundance of burbot, as indexed by CPUE in gillnet surveys, declined during 1970-2000 in eastern Wisconsin waters (WI-2), but abundance in western waters (WI-1) was fairly stable during most of the period before starting to decline in the late 1990s (Fig. 22). In eastern Lake Superior, burbot are a diet item for lake trout but are seen infrequently in assessments.

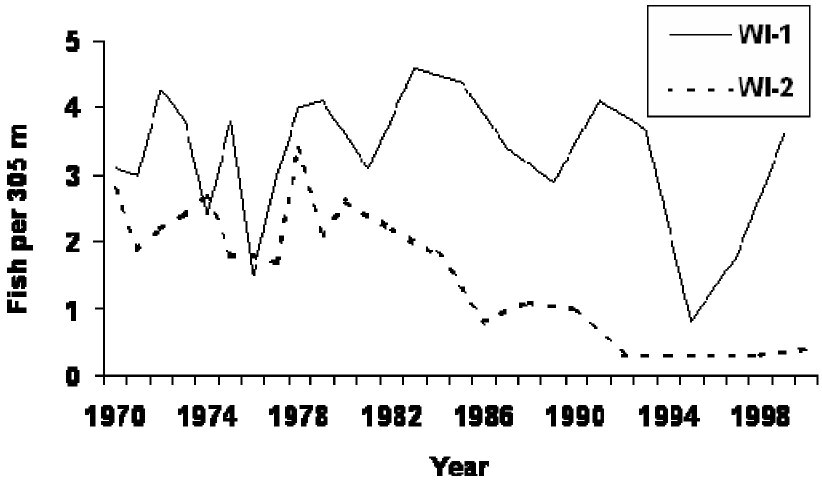

Fig. 22. Relative abundance of burbot as indexed by geometric mean catch (number) per unit effort (305 m of graded-mesh gillnet) in Wisconsin waters of Lake Superior during summer, 1970-2000.

Total annual mortality of burbot in Wisconsin waters was 43% and is attributed mostly to natural causes because the bycatch of burbot in the commercial fishery and harvest in the sport fishery is currently negligible—harvest in the Apostle Islands sport fishery during 1992-1997 averaged 114 fish annually.

Age of burbot collected from Wisconsin and Minnesota waters ranged from 2 to 17 years with a median age of 10 for both sexes. Both sexes exhibited large variations in lengths at a given age. Lake Superior burbot grow slower than other Great Lakes populations (Scott and Crossman 1973; Schram 2000). Slow growth is likely a consequence of the colder temperatures and lower productivity of Lake Superior relative to the other systems.

The decline in burbot abundance in WI-2 waters in recent years has likely occurred elsewhere. The decline is probably related to changes in the fish community that have occurred during the past 20 years and is not related to sea lamprey or fishing mortality. Sea lamprey marking rates on burbot are very low (0-0.1%), and the estimated total annual mortality rate of 43% is indicative of an unexploited population (Clemens 1951; Bailey 1972; Muth and Smith 1974). Increased predation and competition for food resources by the increasingly abundant wild lake trout and siscowet populations are believed to be the primary reasons for the decline in burbot abundance. Burbot appear in the diet of both lake trout and siscowets, and juvenile lake trout, siscowets, and burbot all prey on sculpins and macroinvertebrates. Higher burbot abundance during the 1970s (Fig. 22) was likely due to low biomass of lake trout and siscowets and an abundance of prey species.

Recommendations

- Collect relative abundance and life-history data sufficient to assess age and growth, mortality, spawning areas, diet, and population size of burbot and monitor population trends of other fishes that are subject to sport and commercial fishing (e.g., suckers, yellow perch, northern pike, smallmouth bass, and minnows)
- Monitor population trends and species composition of fish communities in all habitat zones, especially estuary and embayment habitats

THE LAKE SUPERIOR FOOD WEB

Chris J. Harvey[1], Sean P. Cox[2], and James F. Kitchell[3]

The collapse of lake trout populations in the 1950s precipitated sea lamprey control efforts and lakewide stocking of lake trout and Pacific salmon in the 1960s. This stocking continued during the 1970s and 1980s. Naturalized populations of coho salmon and Chinook salmon established in many tributaries, self-sustaining populations of the lean form of lake trout reestablished in much of the lake, and offshore populations of the siscowet and humper lake trout forms increased in abundance. Overall, the abundance of predators in the lake increased steadily during the 1980s and 1990s.

The goal of Lake Superior fisheries management is to rehabilitate and maintain a diverse, healthy fish community dominated by native fishes that support sustainable fisheries. Managers acknowledge that several exotic fishes are now inextricable parts of the community due to ecological mechanisms and societal values (Horns et al. 2003). Managers also recognize that understanding food-web interactions (e.g., predation and competition) among major fish species is essential to achieving rehabilitation goals for depressed native stocks. This summary describes past and present research on Lake Superior food-web dynamics and discusses modeling frameworks through which an understanding of community interactions can be folded into fisheries management.

Food-web structure in Lake Superior has been studied primarily through analysis of stomach contents. Such studies suggest that the food web changed substantially following the perturbations in the 1950s and 1960s. In the ancestral food web, top predators fed primarily on coregonines and cottids (sculpins), which in turn fed on zooplankton and larger invertebrates (*Mysis relicta*, diporeia).

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[1]National Marine Fisheries Service, 2725 Montlake Blvd. E., Seattle, Washington, U.S.A., 98112

[2]Simon Fraser University, 8888 University Drive, Burnaby, British Columbia, Canada, V5A 1S6

[3]University of Wisconsin-Madison, 680 N. Park St., Madison, Wisconsin, U.S.A., 53706

In the 1960s, rainbow smelt replaced coregonines as the major prey of lake trout (Dryer et al. 1965; Anderson and Smith 1971b). In the 1960s and 1970s, rainbow smelt may have caused population declines of lake herring, the major inshore pelagic coregonine, through resource competition or direct predation on larval herring (Lawrie and Rahrer 1973; Selgeby et al. 1978; Selgeby et al. 1994). Rainbow smelt have declined since the late 1970s, but they remained the dominant item in inshore piscivore stomachs, at least through the 1980s and 1990s (Conner et al. 1993; Ray et al. 2007). Because Pacific salmon have higher growth and consumption rates than lake trout, diet similarities could lead to local competition with lake trout for food resources (Negus 1995). Past and recent stomach-content analyses also imply that siscowet, the offshore, deepwater form of lake trout, moved into shallower waters to feed (Thurston 1962; Moore and Bronte 2001), potentially competing with lake trout for prey resources.

Stable-Isotope Analyses

Stable-isotope analysis has recently been employed as a complement to stomach-content analysis to characterize trophic interactions in Lake Superior. In stable-isotope analysis, naturally occurring heavy isotopes (^{15}N, ^{13}C) are used to trace trophic pathways in a food web (Vander Zanden et al. 1998). Stable-isotope analysis essentially follows the "you are what you eat" principle—the isotopic signature of a predator is determined by the isotopic signature of its diet. This method is valuable because a stable-isotope analysis provides dietary information for a much longer time period than a stomach-content analysis, and tissue samples for stable-isotope analysis are much easier and less expensive to collect and analyze than stomach contents, especially considering the temporal and spatial scales associated with determining dietary habits of Lake Superior fishes.

Harvey and Kitchell (2000) examined stable-isotope ratios in the western Lake Superior food web in the late 1990s and found that ratios in lake trout and siscowets were distinct. Siscowets had higher $\delta^{15}N$ (relative trophic position) and lower $\delta^{13}C$ (base of production), implying that siscowets, unlike lake trout, derive most of their growth from different prey (e.g., deepwater coregonines) and are supported by a separate production base (probably deepwater, respired carbon). Harvey and Kitchell (2000) also found that stable-isotope ratios of lake trout and Pacific salmon showed considerable overlap of $\delta^{13}C$, implying a common base of primary production (probably phytoplankton), but their $\delta^{15}N$ values suggested that lake trout occupy a higher trophic position than Pacific salmon. Harvey and Kitchell (2000) concluded that the differences in $\delta^{15}N$ were because adult lake trout probably consume more lake herring and sculpin than Pacific salmon. In eastern U.S. waters, siscowet and lake trout ratios were different in all areas but one located immediately east of the Keweenaw Peninsula (Harvey et al. 2003), where lake trout have recovered to historical levels (Wilberg et al. 2003). The results of these stable-isotope analyses imply that neither siscowets nor Pacific salmon are likely to suppress the recovery of lake trout via competitive interactions.

To better understand spatial foraging patterns, Harvey et al. (2003), using a model that combined the principles of stable-isotope analysis and fish bioenergetics, concluded that, based on their $\delta^{13}C$ signatures, siscowets in the Apostle Islands region derived between 0% and 25% of their growth from nearshore waters (80-m depth and shallower). The $\delta^{15}N$ signatures of lake trout and siscowets differed at all body sizes, indicating little or no direct trophic overlap.

Food Web Models

Given the impacts of exotic invasions, overfishing, and stocking, the Lake Superior ecosystem provides an excellent setting in which to test whether ecosystem models can reveal ecosystem-management conflicts. These factors have been linked to lakewide population collapses of lake trout (overfishing and sea lamprey) and lake herring (overfishing and rainbow smelt) populations. Although most lake trout populations have recovered, most lake herring populations have not, and they continue to exhibit little evidence of compensatory improvements in juvenile survival (Cox and Kitchell 2004). Furthering the work of Kitchell et al. (2000), we developed an ecosystem simulation model of the Lake Superior fish community during 1929-1998 to evaluate the relative impacts of lake trout enhancement programs, fish-community dynamics, and fishing mortality on lake herring. We simulated four alternative hypotheses as potential explanations for lack of compensatory recruitment of juvenile lake herring: strong/weak predation by lake trout and strong/weak predation by rainbow smelt. Strong smelt predation (Fig. 23A) was more consistently related to predicted lake herring biomass than weak smelt predation (Fig. 23B) or either level of lake trout predation. This analysis indicates that lake herring recruitment improved in the 1980s and 1990s because rainbow smelt abundance declined severely. On the other hand, recovery of lake herring populations in Lake Superior began with successful recruitment of the 1977 and 1978 year classes when rainbow smelt were very abundant, and, thus, factors other than rainbow smelt abundance (i.e., favorable environmental factors) may also play a significant role in the recovery of lake herring (Bronte et al. 2003).

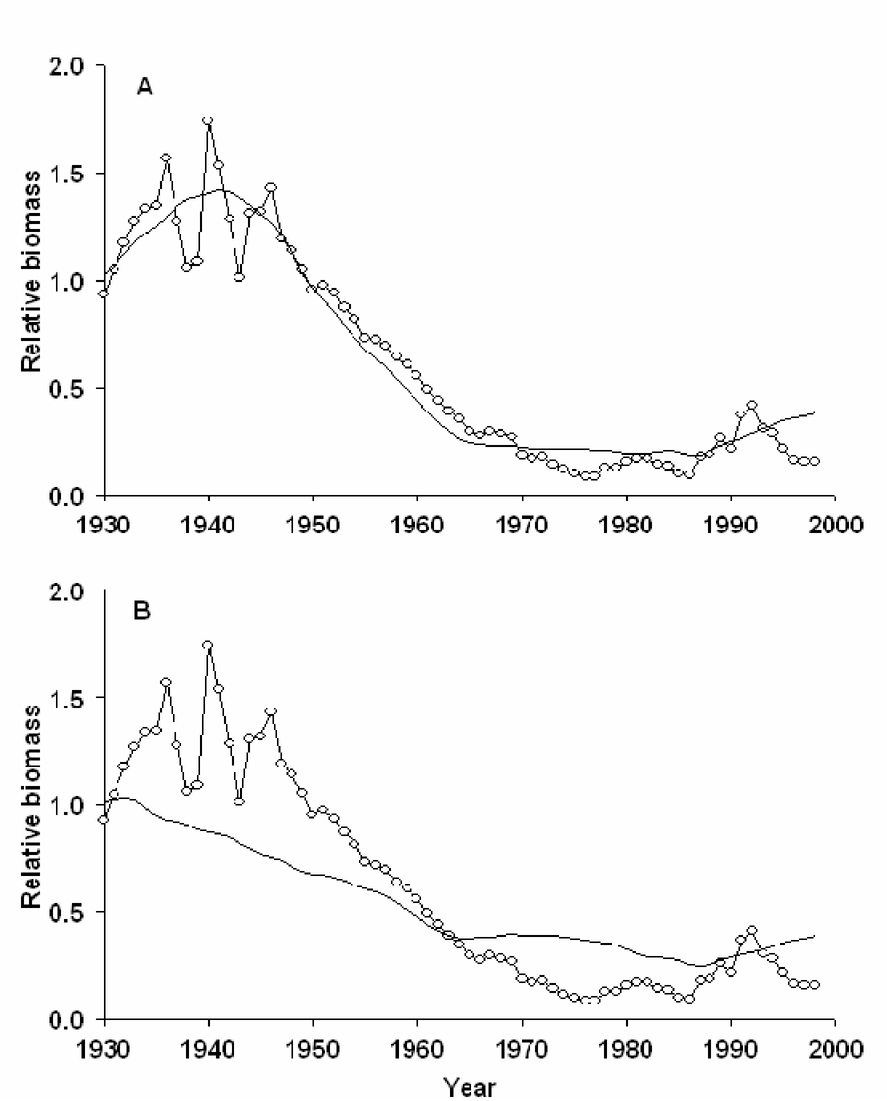

Fig. 23. Comparison of predicted lake herring biomass trajectories in Lake Superior (1930-1998) for the ecosystem model (solid line) and independent single-species assessments (circles) under the strong (A) and weak (B) rainbow smelt predation hypotheses.

Recommendations

- Establish and maintain a database of stomach content information for predator fish with a collection and analysis protocol that is consistent across regions and agencies
- Conduct stable-isotope analyses in both U.S. and Canadian waters
- Develop a spatial ecosystem model that accounts for regional-scale fishing and lake trout stocking impacts on rainbow smelt and lake herring

FUTURE CONSIDERATIONS

Mark P. Ebener
Inter-Tribal Fisheries and Assessment Program
Chippewa Ottawa Resource Authority
179 W. Three Mile Road
Sault Ste. Marie, Michigan, U.S.A. 49783

The habitat and fish communities in Lake Superior are the least degraded of those of the other Great Lakes. Although some tributary and embayment habitats have been impacted by anthropogenic activities, there have been few stresses on offshore and nearshore habitats. Lake herring populations are partially recovered and are a common prey of predators in Lake Superior. The lean, siscowet, and humper forms of lake trout are abundant, and they collectively are the dominant source of predation in the lake. Lake sturgeon, walleye, and brook trout are common members of the fish community, and their abundance is either increasing or stable. Whitefish populations are abundant and support fishery yields that equal or exceed target yields specified in the FCOs. Recent invasive species such as ruffe, gobies, white perch, zebra mussels, and the spiny water flea have proven to be nearly benign in nearshore and offshore waters, and their effects, if any, on the embayment and tributary aquatic communities have been much less than anticipated.

The primary habitat objective is to achieve no net loss of productive capacity and to restore those that have been degraded. While this objective has universal acceptance, it may not be practical. Habitat loss occurs daily on small spatial and extended temporal scales despite the best attempts to halt losses and despite the existence of governmental organizations whose primary responsibility is to protect and restore habitat. A more-practical approach may be to set objectives that identify specific vulnerable habitats (e.g., wetlands inside Black Bay) and the amount of that habitat that should be protected or restored. Structured in this way, habitat objectives would be quantifiable. A second approach involves quantifying the amount of habitat necessary to achieve an objective for a single species or group of species. Unfortunately, objectives established either way depend on quantifying the relationship between available suitable habitat and fish production—something fishery agencies have not achieved.

Achievement of the FCOs will require balancing sea lamprey-control interests with rehabilitation of indigenous fishes. Barriers constructed to block spawning migrations of sea lampreys also block spawning migrations of walleye and brook trout and reduce species diversity above barriers (Dodd et al. 2003). Present and future sea lamprey barriers must be able to effectively pass all species of fish except sea lampreys. The lampricide-treatment protocol to protect lake sturgeon has reduced the risk of nontarget mortality on juvenile sturgeon, but the protocol may also have resulted in more sea lampreys surviving treatment. In Canada, the designation of the northern brook lamprey as a “species at risk” could negatively affect sea lamprey control in those Ontario streams that it inhabits.

Fishery and environmental managers must focus their efforts on rehabilitation and protection of tributary habitat to achieve the FCOs for habitat, lake sturgeon, brook trout, walleye, desirable non-native salmonines, and species diversity. Efforts to rehabilitate watersheds will be impeded by mining, lumber, agricultural, shipping, hydroelectric, and industrial interests; thus, objectives should explicitly recognize that an extended period will be necessary for their achievement. Achievement of objectives will also require action by agencies other than fishery-management agencies; therefore, the LSC should collaborate with agencies such as USEPA, Environment Canada, non-governmental organizations, township and municipal planning boards, and Sea Grant.

Reducing contaminants in fish to levels called for in the Lake Superior Lakewide Management Plan (LAMP) or to levels sufficient to eliminate fish-consumption advisories may not be achievable. The LAMP calls for contaminant-level reductions of 60% in mercury, 33% in PCBs, and 80% in dioxin by 2000 or 2005, and 100% reductions by 2020 (Lake Superior Work Group 2000). It is difficult to imagine achieving these reductions given that much of the contamination originates from outside the Great Lakes basin, and even outside North America.

Effective ballast-water management must be developed and implemented to stop the entry of invasive species into Lake Superior. Although most of the recent invaders appear to be benign, the next invader could be one that causes a major ecological disruption, such as the disruptions caused by sea lamprey and rainbow smelt. Effective ballast-water management should include identifying vectors for potential invaders as well as preventing their introduction.

Lack of information on abundance and biomass of important species and definition of appropriate harvest levels may prevent achievement of the FCOs. SCAA models are being used to estimate abundance and mortality of lake trout, whitefish, and lake herring in selected areas of Lake Superior but not lakewide. Biomass and abundance estimates are necessary for evaluating fishery yields as well as predator-prey interactions. Target maximum mortality rates of 45% for lake trout and 65% for whitefish are being used in some areas to set harvest limits, but these rates may not be appropriate (Bence and Ebener 2002).

Lastly, evaluation and achievement of the FCOs depends on agency commitment. Information gathering and analysis of data are essential parts of environmental and fisheries management. Recent downsizing of fishery agencies by state and provincial governments is hindering the process of collecting and analyzing the data needed to evaluate and measure progress toward achieving objectives. By signing the Joint Plan, each state and province bordering the Great Lakes committed itself to cooperative management of aquatic resources. Before beginning to downsize, each agency needs to evaluate how that will effect its commitment to the Joint Plan.

REFERENCES

Adams, C.E., and Kregear, R.D. 1969. Sedimentary and faunal environments of eastern Lake Superior. *In* Proc. 12th Conf. Great Lakes Res., Internat. Assoc. Great Lakes Res. pp. 1-20.

Anderson, E.D., and Smith, L.L., Jr. 1971a. Factors affecting abundance of lake herring (*Coregonus artedii* Lesueur) in western Lake Superior. Trans. Am. Fish. Soc. **100**: 691-707.

Anderson, E.D., and Smith, L.L. 1971b. A synoptic study of food habits of 30 fish species from western Lake Superior. Minnesota Agricultural Experimental Station Rep. No. 279. University of Minnesota, St. Paul, MN, 55101.

Assel, R.A. 1985. Lake Superior cooling season temperature climatology. NOAA Tech. Mem. ERL GLERL-58, Ann Arbor, MI.

Auer, N.A. 1996. Response of spawning lake sturgeons to change in hydroelectric facility operation. Trans. Am. Fish. Soc. **125**: 66-77.

Auer, N.A. 1999. Population characteristics and movements of lake sturgeon in the Sturgeon River and Lake Superior. J. Great Lakes Res. **25**: 282-293.

Auer, N.A. [ED.]. 2003. A lake sturgeon rehabilitation plan for Lake Superior. Great Lakes Fish. Comm. Misc. Pub. 2003-02.

Bailey, M.M. 1972. Age, growth, reproduction and food of the burbot, *Lota lota* (Linnaeus), in southwestern Lake Superior. Trans. Am. Fish. Soc. **101**: 667-674.

Baldwin, N.S., Saalfeld, R.W., Ross, M.A., and Buettner, H.J. 1979. Commercial fish production in the Great Lakes 1867-1977. Great Lakes Fish. Comm. Tech. Rep. 3.

Baldwin, N.A., Saalfeld, R.W., Dochoda, M.R., Buettner, H.J., and Eshenroder, R.L. 2002. Commercial fish production in the Great Lakes 1867-2000. http://www.glfc.org/databases/commercial/commerc.asp. (August 2002).

Barbiero, R.P., Little, R.E., and Tuchman, M.L. 2001. Results from the USEPA's biological open water surveillance program of the Laurentian Great Lakes: III. Zooplankton. J. Great Lakes Res. **27**: 167-184.

Barbiero, R.P., and Tuchman, M.L. 2001. Results from the USEPA's biological open water surveillance program of the Laurentian Great Lakes: I. Introduction and phytoplankton results. J. Great Lakes Res. **27**: 134-154.

Barbiero, R.P., Tuchman, M.L., Warren, G.J., and Rockwell, D.C. 2002. Evidence of recovery from phosphorus enrichment in Lake Michigan. Can. J. Fish. Aquat. Sci. **59**: 1639-1647.

Becker, G.C. 1983. Fishes of Wisconsin. The University of Wisconsin Press, Madison, WI.

Bence, J.R. 2002. Stock assessment models. *In* Summary status of lake trout and lake whitefish populations in the 1836 treaty-ceded waters of Lakes Superior, Huron and Michigan in 2000, with recommended yield and effort levels for 2001. *Edited by* J.R. Bence and M.P. Ebener. Technical Fisheries Committee, 1836 Treaty-Ceded Waters of Lakes Superior, Huron and Michigan. Michigan Department of Natural Resources, Charlevoix, MI. pp 8-17.

Bence, J.R., and Smith, K.M. 1999. An overview of recreational fisheries of the Great Lakes. *In* Great Lakes fisheries policy and management. *Edited by* W.W. Taylor and C.P. Ferreri. Michigan State University Press, East Lansing, MI. pp. 259-306.

Bence, J.R., and Ebener, M.P. [ED.]. 2002. Summary status of lake trout and lake whitefish populations in the 1836 treaty-ceded waters of Lakes Superior, Huron and Michigan in 2000, with recommended yield and effort levels for 2001. Technical Fisheries Committee, 1836 Treat-Ceded Waters of Lakes Superior, Huron and Michigan. Michigan Department of Natural Resources, Charlevoix, MI.

Bergstedt, R.A., McDonald, R.B., Twohey, M.B., Mullet, K.M., Young, R.J., and Heinrich, J.W. 2003. Reduction in sea lamprey hatching success due to release of sterilized males. J. Great Lakes Res. **29**: 435-444.

Boyer, L.F., Cooper, R.A., Long, D.T., and Askew, T.M. 1989. Burbot (*Lota lota*) biogenic sedimentary structures in Lake Superior. J. Great Lakes Res. **15**: 174-185.

Brege, D.A., and Kevern, N.A. 1978. Michigan commercial fishing regulations: a summary of Public acts and conservation commission orders, 1865 through 1975. Michigan Sea Grant Program, Ref. Rep. MICHU-SG-78-605.

Bronte, C.R., Selgeby, J.H., and Swedberg, D.V. 1993. Dynamics of yellow perch in western Lake Superior. N. Am. J. Fish. Manage. **13**: 511-523.

Bronte, C.R., Hesselberg, R.J., Shoesmith, J.A., and Hoff, M.H. 1996. Discrimination among spawning concentrations of Lake Superior lake herring based on trace element profiles in sagittae. Trans. Am. Fish. Soc. **125**: 852-859.

Bronte, C.R., Evrard, L.M., Brown, W.P., Mayo, K.R., and Edwards, A.J. 1998. Fish community changes in the St. Louis River estuary, Lake Superior, 1989-1996: is it ruffe or population dynamics? J. Great Lakes Res. **24**: 309-318.

Bronte, C.R., Ebener, M.P., Schreiner, D.R., DeVault, D.S., Petzold, M.M., Jensen, D.A., Richards, C., and Lozano, S.J. 2003. Fish community change in Lake Superior, 1970-2000. Can. J. Fish. Aquat. Sci. **60**: 1552-1574.

Brown, R.W., Ebener, M.P., and Gorenflo, T. 1999. Great Lakes commercial fisheries: historical overview and prognosis for the future. *In* Great Lakes fisheries policy and management. *Edited by* W.W. Taylor and C.P. Ferreri. Michigan State University Press, East Lansing, MI. pp. 307-354.

Bruch, R.M. 1999. Management of lake sturgeon on the Winnebago system—long-term impacts of harvest and regulations on population structure. J. Appl. Ichthyol. **15**: 142-152.

Burnham-Curtis, M.K. 1993. Intralacustrine speciation of *Salvelinus namaycush* in Lake Superior: an investigation of genetic and morphological variation and evolution of lake trout in the Laurentian Great Lakes. Ph.D. dissertation, University of Michigan, Ann Arbor, MI.

Busiahn, T.R. [ED.]. 1990. Fish community objectives for Lake Superior. Great Lakes Fish. Comm. Spec. Pub. 90-1.

Busiahn, T.R. 1997. Ruffe control: a case study of an aquatic nuisance species control program. *In* Zebra mussels and aquatic nuisance species. *Edited by* F.M. D'Itri. Ann Arbor Press, Ann Arbor, MI. pp. 69-86.

Christie, W.J. 1978. A study of freshwater fishery regulations based on North American experience. FAO Fish. Rep. 180.

Christie, C.G., Adams, J.V., Steeves, T.B., Slade, J.W., Cuddy, D.W., Fodale, M.F., Young, R.J., Kuc, M., and Jones, M.L. 2003. Selecting Great Lakes streams for lampricide treatment based on larval sea lamprey surveys. J. Great Lakes Res. **29**(Supple. 1): 152-160.

Clemens, H.P. 1951. The growth of the burbot, *Lota lota maculosa*, (Le Sueur), in Lake Erie. Trans. Am. Fish. Soc. **80**: 163-173.

Colby, P.J., and Nepszy, S.J. 1981. Variation among stocks of walleye (*Stizostedion vitreum vitreum*): management implications. Can. J. Fish. Aquat. Sci. **38**: 1814-1831.

Close, T., and Hassinger, R. 1981. Evaluation of Madison, Donaldson, and Kamloops strains of rainbow trout *Salmo gairdneri* in Lake Superior. Minnesota Department of Natural Resources, Division of Fish and Wildlife, Section of Fisheries Investigational Report 372, St. Paul, MN.

Collins, J.J. 1979. Relative efficiency of multifilament and monofilament nylon gillnet towards lake whitefish (*Coregonus clupeaformis*) in Lake Huron. J. Fish. Res. Board Can. **36**: 1180-1185.

Collins, J.J. 1987. Increased catchability of the deep monofilament nylon gillnet and its expression in a simulated fishery. Can. J. Fish. Aquat. Sci. **44**(Supple. 2): 129-135.

Conner, D.J., Bronte, C.R., Selgeby, J.H., and Collins, H.L. 1993. Food of salmonine predators in Lake Superior, 1981-1987. Great Lakes Fish. Comm. Tech. Rep. 59.

Conway, J.B., Ruschmeyer, O.R., Olson, T.A. and Odlaug, T.O. 1973. The distribution, composition, and biomass of the crustacean zooplankton population in western Lake Superior. University of Minnesota, Water Resources Center Bulletin 63.

Cook, D.G. 1975. A preliminary report on the benthic macroinvertebrates of Lake Superior. Canadian Fish. Marine Serv. Tech. Rep. No. 572.

Cox, S.P., and Kitchell, J.F. 2004. The Lake Superior ecosystem, 1929-1998: simulating alternative hypotheses for lake herring recruitment failure. Bull. Mar. Sci. **74**: 671-684.

D'Amelio, S. 2002. Conservation genetics and metapopulation structure of brook trout (*Salvelinus fontinalis*) in Nipigon Bay (Lake Superior, Ontario). M.Sc. thesis, Trent University, Peterborough, Ontario, Canada.

Davis, C.C. 1966. Plankton studies in the largest great lakes of the world, with special reference to the St. Lawrence Great Lakes of North America. University of Michigan, Great Lakes Research Division, Pub. No. 14. pp. 1-36.

DeHaan, P.W. 2003. Demographic and life history characteristics of remnant lake sturgeon populations in the upper Great Lakes basin: Inferences based on genetic analyses. M.Sc. thesis, Department Fisheries and Wildlife, Michigan State University, East Lansing, MI.

Dermott, R. 1978. Benthic diversity and substrate-fauna associations in Lake Superior. J. Great Lakes Res. **4**: 505-512.

Dodd, H.R., Hayes, D.B., Baylis, J.R., Carl, L.M., Goldstein, J.D., McLaughlin, R.L., Noakes, D.L.G., Porto, L.M., and Jones, M.L. 2003. Low-head sea lamprey barrier effects on stream habitat and fish communities in the Great Lakes basin. J. Great Lakes Res. **29**(Supple. 1): 386-402.

Doherty, R. 1990. Disputed waters: Native Americans and the Great Lakes Fishery. The University Press of Kentucky, Lexington, KY.

Dryer, W.R., Erkkila, L.F., and Tetzloff, C.L. 1965. Food of lake trout in Lake Superior. Trans. Am. Fish. Soc. **94**: 169-176.

Ebener, M.P. 1997. Recovery of lake whitefish populations in the Great Lakes. Fisheries **22**(7): 18-20.

Edsall, T.A., and Gannon, J.E. 1993. A profile of the St. Marys River. Michigan Sea Grant College Program, MICHU-SG-93-700.

Fahnenstiel, G.L., and Glime, J.M. 1983. Subsurface chlorophyll maximum and associated *Cyclotella* pulse in Lake Superior. Int. Rev. Gesamten. Hydrobiol. **68**: 605-618.

Fields, R.D., and Philip, D.P. 1995. A comparison of protein electrophoresis, mitochondrial DNA RFLP analysis, and genomic DNA RAPDs for their ability to detect genetic variation among populations of lake herring (*Coregonus artedii*). Illinois Natural History Survey, Aquatic Ecology Technical Rep. 95/10, Champaign, IL.

Great Lakes Fishery Commission. 1992. Strategic vision of the Great Lakes Fishery Commission for the decade of the 1990s. Great Lakes Fish. Comm., Ann Arbor, MI.

Great Lakes Fishery Commission. 1997. A joint strategic plan for management of Great Lakes fisheries. Ann Arbor, MI. http://www.glfc.org/pubs_out/docs.htm. (Supersedes 1980 version).

Glassmeyer, S.T., De Vault, D.S., Myers, T.R., and Hites, R.A. 1997. Toxaphene in Great Lakes fish: A temporal, spatial, and trophic study. Environ. Sci. Technol. **31**: 84-88.

Hansen, M.J. [ED.]. 1994. The state of Lake Superior in 1992. Great Lakes Fish. Comm. Spec. Pub. 94-1.

Hansen, M.J. [ED.]. 1996. A lake trout restoration plan for Lake Superior. Great Lakes Fish. Comm., Ann Arbor, MI. 34 p.

Hansen, M.J., and Schorfhaar, R.G. 1994. History. *In* The state of Lake Superior in 1992. *Edited by* M.J. Hansen. Great Lakes Fish. Comm. Spec. Pub. 94-1. pp. 7-12.

Hansen, M.J., Schorfhaar, R.G., and Selgeby, J.H. 1998. Gill-net saturation by lake trout in Michigan waters of Lake Superior. North Am. J. Fish. Manage. **18**: 847-853.

Hansen, M.J., Madenjian, C.P., Selgeby, J.H., and Helser, T.E. 1997. Gillnet selectivity for lake trout (*Salvelinus namaycush*) in Lake Superior. Can. J. Fish. Aquat. Sci. **54**: 2483-2490.

Hansen, M.J., Ebener, M.P., Schorfhaar, R.G., Schram, S.T., Schreiner, D.R., and Selgeby, J.H. 1994. Declining survival of lake trout stocked during 1963-1986 in U.S. waters of Lake Superior. North Am. J. Fish. Manage. **14**: 395-402.

Hansen, M.J., Peck, J.W., Schorfhaar, R.G., Selgeby, J.H., Schreiner, D.R., Schram, S.T., Swanson, B.L., MacCallum, W.R., Burnham-Curtis, M.K., Curtis, G.L., Heinrich, J.W., and Young, R.J. 1995. Lake Trout (*Salvelinus namaycush*) populations in Lake Superior and their restoration in 1959-1993. J. Great Lakes Res. **21**(Supple. 1): 152-175.

Harkness, W.J.K., and Dymond, J.R. 1961. The lake sturgeon: the history of its fishery and problems of conservation. Ontario Department of Lands and Forests, Fish and Wildlife Branch, Toronto, Ontario, Canada.

Harvey, C.J., and Kitchell, J.F. 2000. A stable isotope evaluation of the structure and spatial heterogeneity of a Lake Superior food web. Can. J. Fish. Aquat. Sci. **57**: 1395-1403.

Harvey, C.J., Schram, S.T., and Kitchell, J.F. 2003. Trophic relationships among lean and siscowet lake trout in Lake Superior. Trans. Am. Fish. Soc. **132**: 219-228.

Hassinger, R., and Kuechenmeister, A. 1972. Status of the herring stocks in Minnesota waters of Lake Superior. Lake Superior Committee, Great Lakes Fish. Comm., Ann Arbor, MI.

Hay-Chmielewski, E.M., and Whelan, G.E. [EDS.]. 1997. Lake sturgeon rehabilitation strategy: Lake sturgeon committee report. Michigan Department of Natural Resources, Fisheries Special Rep. No. 18.

Healey, M.C. 1975. Dynamics of exploited whitefish populations and their management with special reference to the Northwest Territories. J. Fish. Res. Board Can. **32**: 427-448.

Healey, M.C. 1978. Fecundity changes in exploited populations of lake whitefish (*Coregonus clupeaformis*) and lake trout (*Salvelinus namaycush*). J. Fish. Res. Board Can. **35**: 945-950.

Healey, M.C. 1980. Growth and recruitment in experimentally exploited lake whitefish (*Coregonus clupeaformis*) populations. Can. J. Fish. Aquat. Sci. **37**: 255-267.

Heinrich, J.W., Mullett, K.M., Hansen, M.J., Adams, J.V., Klar, G.T., Johnson, D.A., Christie, G.C., and Young, R.J. 2003. Sea lamprey abundance and control in Lake Superior 1957-1999. J. Great Lakes Res. **29**(Supple. 1): 566-583.

Henderson, B.A., and Anderson, D.M. 2002. Phenotypic differences in buoyancy and energetics of lean and siscowet lake charr in Lake Superior. Environ. Bio. Fish. **64**: 203-209.

Henson, E.B., and Herrington, H.B. 1965. Sphaeriidae (Mollusca: Pelecypoda) of Lakes Huron and Michigan in the vicinity of the Straits of Mackinac. Great Lakes Research Division, University of Michigan, Pub. No. 13, Ann Arbor, MI.

Hiltunen, J.K. 1969. Invertebrate macrobenthos of western Lake Superior. Mich. Academician **1**: 123-133.

Hoff, M.H. [ED.]. 1996. Status of walleye in Lake Superior and its tributaries. Minutes of the 1996 (annual) meeting of the Lake Superior Committee, Attachment 4.g. (pp.89-146). Great Lakes Fish. Comm., Ann Arbor, MI.

Hoff, M.H. [ED.]. 2003. A rehabilitation plan for walleye populations and habitats in Lake Superior. Great Lakes Fish. Comm. Misc. Pub. 2003-01.

Hoff, M.H. 2004. Discrimination among spawning aggregations of lake herring from Lake Superior using whole-body morphometric characteristics. J. Great Lakes Res. **30**(Supple. 1): 385-394.

Hoff, M.H., Bronte, C.R., and Selgeby, J.H. 1995. Population status and trends for Lake Superior forage fishes, 1978-94. Minutes of the 1995 (annual) meeting of the Lake Superior Committee, Appendix VII (pp. 75-84). Great Lakes Fish. Comm., Ann Arbor, MI.

Hoff, M.H., Link, J., and Haskell, C. 1997a. Piscivory by Lake Superior lake herring (*Coregonus artedi*) on rainbow smelt (*Osmerus mordax*) in winter, 1993-1995. J. Great Lakes Res. **23**: 210-211.

Hoff, M.H., Pronin, N., and Baldanova, D.R. 1997b. Parasites of lake herring (*Coregonus artedi*) from Lake Superior, with special reference to use of parasites as markers of stock structure. J. Great Lakes Res. **23**: 458-467.

Holland, R.E. 1965. The distribution and abundance of planktonic diatoms in Lake Superior. In Proc. 8th Conf. Great Lakes Res., Internat. Assoc. Great Lakes Res. pp. 96-105.

Horns, W.H., Bronte, C.R., Busiahn, T.R., Ebener, M.P., Eshenroder, R.L., Gorenflo, T., Kmiecik, N., Mattes, W., Peck, J.W., Petzold, M., and Schreiner, D.R. 2003. Fish-community objectives for Lake Superior. Great Lakes Fish. Comm. Spec. Pub. 03-01.

James, R.R., McDonald, J.G., Symonik, D.M., Swackhamer, D.L., and Hites, R.A. 2001. Volatilization of toxaphene from Lakes Michigan and Superior. Environ. Sci. Technol. **35**: 3653-3660.

Jensen, A.L. 1976. Assessment of the United States lake whitefish (*Coregonus clupeaformis*) fisheries of Lake Superior, Lake Michigan, and Lake Huron. J. Fish. Res. Board Can. **33**: 747-759.

Jensen, A.L. 1985. Relations among net reproductive rate and life history parameters for lake whitefish (*Coregonus clupeaformis*). Can. J. Fish. Aquat. Sci. **42**: 164-168.

Kannan, K., Yamashita, N., Imagawa, T., Decoen, W., Khim, J.S., Day, R.M, Summer, C.L., and Giesy, J.P. 2000. Polychlorinated naphthalenes and polychlorinated biphenyls in fishes from Michigan waters including the Great Lakes. Environ. Sci. Technol. **34**: 566-572.

King, E.L., Jr. 1980. Classification of sea lamprey (*Petromyzon marinus*) attack marks on Great Lakes lake trout (*Salvelinus namaycush*). Can. J. Fish. Aquat. Sci. **37**: 1989-2006.

Kitchell, J.F., Cox, S.P., Harvey, C.J., Johnson, T.B., Mason, D.M., Schoen, K.K., Aydin, K., Bronte, C., Ebener, M., Hansen, M., Hoff, M., Schram, S., Schreiner, D., and Walters, C.J. 2000. Sustainability of the Lake Superior fish community: interactions in a food web context. Ecosystems **3**: 545-560.

Kocik, J.F., and Jones, M.L. 1999. Pacific salmonines in the Great Lakes basin. *In* Great Lakes Fisheries Policy and Management, a Binational Perspective. *Edited by* W.W. Taylor, and C.P. Ferreri. Michigan State University Press, East Lansing, MI. pp. 455-488.

Krueger, C.C., and May, B. 1987a. Genetic comparison of naturalized rainbow trout populations among Lake Superior tributaries: differentiation based on allozyme data. Trans. Am. Fish. Soc. **116**: 795-806.

Krueger, C.C., and May, B. 1987b. Stock identification of naturalized brown trout in Lake Superior tributaries: differentiation based on allozyme data. Trans. Am. Fish. Soc. **116**: 785-794.

Krueger, C.C., Perkins, D.L., Everett, R.J., Schreiner, D.R., and May, B. 1994. Genetic variation in naturalized rainbow trout (*Oncorhynchus mykiss*) from Minnesota tributaries to Lake Superior. J. Great Lakes Res. **20**: 299-316.

Lake Superior Work Group. 2000. Lake Superior lakewide management plan (LaMP) 2000 [online]. Available from http://www.epa.gov/glnpo/lakesuperiorlamp2000 [accessed 7 Jun 2007].

Lake Superior Work Group. 2002. Lake Superior lakewide management plan LaMP: 2002 progress report (Parts A,B,C). Available from http://www.epa.gov/glnpo/lakesuperior/index.html [accessed 7 Jun 2007].

Lawrie, A.H., and Rahrer, J.F. 1972. Lake Superior: effects of exploitation and introduction on the salmonid community. J. Fish. Res. Board Can. 29: 765-776.

Lawrie, A.H., and Rahrer, J.F. 1973. Lake Superior: A case history of the lake and its fisheries. Great Lakes Fish. Comm. Tech. Rep. No. 19. 69 pp.

Leach, J.H., Mills, E.L. and Dochoda, M.R. 1999. Non-indigenous species in the Great Lakes: ecosystem impacts, binational politics, and management. *In* Great Lakes Fisheries Policy and Management. *Edited by* W.W. Taylor and C.P. Ferreri. Michigan University Press, East Lansing, MI. pp. 185-208.

Legault, J., Kuchenberg, T., and Sisulak, M.E. 1978. Reflections in a tarnished mirror: the use and abuse of the Great Lakes. Golden Glow Publishing, Sturgeon Bay, WI.

Link, J., Selgeby, J.H., Hoff, M.H., and Haskell, C. 1995. Winter diet of lake herring (*Coregonus artedi*) in western Lake Superior. J. Great Lakes Res. **21**: 395-399.

Liu, K., and Jensen, A.L. 1993. Simulation of population regulation in lake whitefish (*Coregonus clupeaformis*). J. Fish. Soc. Taiwan **20**: 153-166.

Lodge, D.M., Taylor, C.A., Holdich, D.M., and Skurdal, J. 2000. Nonindigenous crayfishes threaten North American freshwater biodiversity: lessons from Europe. Fisheries **25**(8): 7-22.

Luross, J.M., Alaee, M., Sergeant, D.B., Cannon, C.M., Solomon, K.R., and Muir, D.C.G. 2002. Spatial distribution of polybrominated diphenyl ethers and polybrominated biphenyls in lake trout from the Laurentian Great Lakes. Chemosphere **46**: 665-672.

MacCallum, W.R., and Selgeby, J.H. 1984. Lake Superior revisited 1984. Can. J. Fish. Aquat. Sci. **44**(Supple. 2): 23-36.

MacEachen, D.C., Russel, R.W., and Whittle, D.M. 2000. Spatial distribution of mercury and organochlorine contaminants in Great Lakes sea lamprey (*Petromyzon marinus*). J. Great Lakes Res. **26**: 112-119.

MacCrimmon, H.R. 1971. World distribution of rainbow trout (*Salmo gairdneri*). J. Fish. Res. Board Can. **28**: 663-704.

Mayo, K.R., Selgeby, J.H., and McDonald, M.E. 1998. A bioenergetics modeling evaluation of top-down control of ruffe in the St. Louis River, western Lake Superior. J. Great Lakes Res. **24**: 329-342.

McMahon, R.F. 1983. Ecology of an invasion pest bivalve, Corbicula. *In* The Mollusca, vol. 6, Ecology. *Edited by* W.D. Russell-Hunter, Academic Press, San Diego, CA. pp. 505-561

Mills, E.L., Leach, J.H., Carlton, J.T., and Secor, C.L. 1993. Exotic species in the Great Lakes: a history of biotic crises and anthropogenic introductions. J. Great Lakes. Res. **19**: 1-54.

Momot, W.T. 1996. History of the range extension of *Orconectes rusticus* into northwestern Ontario and Lake Superior. Freshwater Crayfish **XI**: 61-72.

Moore, H.H., and Braem, R.A. 1965. Distribution of fishes in U.S. streams tributary to Lake Superior. U.S. Fish and Wildlife Service, Special Scientific Report–Fisheries No. 516.

Moore, S.A., and Bronte, C.R. 2001. Delineation of sympatric morphotypes of lake trout in Lake Superior. Trans. Am. Fish. Soc. **130**: 1233-1240.

Muth, K., and Smith, L.L., Jr. 1974. The burbot fishery in Lake of the Woods. University of Minnesota Agricultural Experimental Station Tech. Bull. No. 296. 68 pp.

Nalepa, T.F., and Schloesser, D.W. [EDS.]. 1993. Zebra mussels: biology, impacts, and control. Lewis Publishers, Boca Raton, FL.

Negus, M.T. 1995. Bioenergetics modeling as a salmonine management tool applied to Minnesota waters of Lake Superior. North Am. J. Fish. Manage. **15**: 60-78.

Newman, L.E., and Dubois, D.B. 1997. Status of brook trout in Lake Superior. Lake Superior Technical Committee, Great Lakes Fish. Comm., Ann Arbor, MI. 33 pp.

Newman, L.E., DuBois, R.B., and Halpern, T.N. 2003. A brook trout rehabilitation plan for Lake Superior. Great Lakes Fish. Comm. Misc. Pub. 2003-03. 39 pp.

Nunan, P.J. 1967. Pink salmon in Lake Superior. Ontario Fish Wildl. Rev. **6**: 8-13.

Parsons, J.W. 1973. History of salmon in the Great Lakes, 1850-1970. U.S. Fish and Wildlife Service, Bureau of Sport Fisheries and Wildlife, Tech. Paper 68.

Patalas, K. 1972. Crustacean plankton and the eutrophication of St. Lawrence Great Lakes. J. Fish. Res. Board Can. **29**: 1451-1462.

Peck, J.W. 1970. Straying and reproduction of coho salmon, *Oncorhynchus kisutch*, planted in a Lake Superior tributary. Trans. Am. Fish. Soc. **99**: 591-595.

Peck, J.W. 1992. The sport fishery and contribution of hatchery trout and salmon in Lake Superior and tributaries at Marquette, Michigan, 1984-87. Michigan Department of Natural Resources, Fisheries Research Rep. 1975.

Peck, J.W., and Wright, A.T. 1972. Preliminary status report on lake herring in the Michigan waters of Lake Superior. Lake Superior Committee, Great Lakes Fish. Comm., Ann Arbor, MI.

Peck, J.W., Jones, T.S., MacCallum, W.R., and Schram, S.T. 1999. Contribution of hatchery-reared fish to Chinook salmon populations and sport fisheries in Lake Superior. North Am. J. Fish. Manage. **19**: 155-164.

Peck, J.W., MacCallum, W.R., Schram, S.T., Schreiner, D.R., and Shively, J.D. 1994. Other salmonines. *In* The state of Lake Superior in 1992. *Edited by* M.J. Hansen. Great Lakes Fish. Comm. Spec. Pub. 94-1. pp. 35-52.

Peck, J.W., Schorfhaar, R., and Wright, A.T. 1974. Status of selected fish stocks in Michigan's Great Lakes waters and recommendations for commercial harvest. Michigan Department of Natural Resources, Fisheries Tech. Rep. 73-33.

Pratt, D.M., Blust, W.H., and Selgeby, J.H. 1992. Ruffe, *Gymnocephalus cernuus*: newly introduced in North America. Can. J. Fish Aquat. Sci. **49**: 1616-1618.

Pycha, R.L., and King, G.R. 1975. Changes in the lake trout population of southern Lake Superior in relation to the fishery, sea lamprey, and stocking, 1950-70. Great Lakes Fish. Comm. Tech. Rep. 28. 34 p.

Ray, B.A., Hrabik, T.R., Ebener, M.P., Gorman, O.T., Schreiner, D.R., Schram, S.T., Sitar, S.P., Mattes, W.P., and Bronte, C.R. 2007. Diet and prey selection by Lake Superior lake trout during spring, 1986-2001. J. Great Lakes Res. **33**: 104-113.

Reynoldson, T.B., and Day, K.E. 1998. Biological guidelines for the assessment of sediment quality in the Laurentian Great Lakes. National Water Research Institute Rep. No. 98-232, Environment Canada, Burlington, Ontario, Canada.

Richards, J.M., Hansen, M.J., Bronte, C.R., and Sitar S.P. 2004. Recruitment dynamics of the 1971-1991 year-classes of lake trout in Michigan waters of Lake Superior. North Am. J. Fish. Manage. **24**: 475-489.

Ricker, W.E. 1975. Computation and interpretation of biological statistics of fish populations. Bull. Fish. Res. Board Can. No. 191.

Roosevelt, R.B. 1865. Superior fishing. Minnesota Historical Society Press, St. Paul, MN. 310 p.

Schelske, C.L., and Roth, J.C. 1973. Limnological survey of lakes Michigan, Superior, Huron and Erie. Great Lakes Research Division, University of Michigan, Pub. No. 17, Ann Arbor, MI.

Schelske, C.L., Feldt, L.E., Santiago, M.A., and Stoermer, E.F. 1972. Nutrient enrichment and its effect on phytoplankton production and species composition in Lake Superior. *In* Proc. 15th Conf. Great Lakes Res., Internat. Assoc. Great Lakes Res. pp. 149-165

Schneider, J.C., and Leach, J.H. 1979. Walleye stocks in the Great Lakes, 1800-1975: Fluctuations and possible causes. Great Lakes Fish. Comm. Tech. Rep. 31. 51 p.

Schneider, K., Assel, R.A., and Croley, II, T.E. 1993. Normal water temperature and ice cover of the Laurentian Great Lakes a computer animation, data base, and analysis tool. NOAA Technical Mem., ERL GLERL-81, Ann Arbor, MI. 47 p.

Schram, S.T. 2000. Seasonal movement and mortality estimates of burbot in Wisconsin waters of western Lake Superior. *In* Burbot: biology, ecology, and management. *Edited by* V.L. Paragamian and D.W. Willis. Am. Fish. Soc. Fish. Manage. Sec. Pub. No. 1. pp. 90-95.

Schram, S.T., and Fabrizio, M.C. 1998. Longevity of Lake Superior lake trout. N. Am. J. Fish. Manage. **18**: 700-703.

Schram, S.T., Atkinson, J.R., and Pereira, D.L. 1991. Lake Superior walleye stocks: status and management. *In* Status of walleye in the Great Lakes: case studies prepared for the 1989 workshop. *Edited by* P.J. Colby, C.A. Lewis, and R.L. Eshenroder. Great Lakes Fish. Comm. Spec. Pub. 91-1. pp. 1-22.

Schram, S.T., Margenau, T.L., and Blust, W.H. 1992. Population biology and management of the walleye in western Lake Superior. Wisconsin Department of Natural Resources, Tech. Bull. 177. 28 p.

Schram, S.T., J. Lindgren, L.M. Evrard. 1999. Reintroduction of lake sturgeon in the St. Louis River, western Lake Superior. North Am. J. Fish. Manage. **19**: 815-823.

Schreiner, D.R. [ED.]. 1995. Fisheries management plan for the Minnesota waters of Lake Superior. Minnesota Department of Natural Resources, Spec. Fish. Pub. No. 149, St. Paul, MN. 86 p.

Schreiner, D.R., and Schram, S.T. 1997. Lake trout rehabilitation in Lake Superior: maintaining our progress. Fisheries **22**(7): 12-14.

Scott, W.B., and Crossman, E.J. 1973. Freshwater fishes of Canada. Fish. Res. Board Can., Bull. 184. 966 pp.

Selgeby, J.H. 1975. Life histories and abundance of crustacean zooplankton in the outlet of Lake Superior, 1971-72. J. Fish. Res. Board Can. **32**: 461-470.

Selgeby, J.H. 1982. Decline of lake herring (*Coregonus artedii*) in Lake Superior: an analysis of the Wisconsin herring fishery, 1936-78. Can. J. Fish. Aquat. Sci. **39**: 554-563.

Selgeby, J.H., MacCallum, W.R., and Swedberg, D.V. 1978. Predation by rainbow smelt (*Osmerus mordax*) on lake herring (*Coregonus artedii*) in western Lake Superior. J. Fish. Res. Board Can. **35**: 1457-1463

Selgeby, J.H., MacCallum, W.R., and Hoff, M.H. 1994. Rainbow smelt—larval lake herring interactions: competitors or casual acquaintances. U.S. Department of Interior, National Biological Survey, Bio. Rep. 25, Washington, DC. 9 pp.

Smith, S.H. 1968. Species succession and fishery exploitation in the Great Lakes. J. Fish. Res. Board Can. **25**: 667-693.

Smith, B.R., and Tibbles, J.J. 1980. Sea lamprey *Petromyzon marinus* in Lakes Huron, Michigan, and Superior: History of invasion and control, 1936-78. Can. J. Fish. Aquat. Sci. **37**: 1780-1801.

Stephenson, S.A. 1998. Status of lake sturgeon (*Acipenser fulvescens*) in three Canadian Lake Superior tributaries. Ontario Ministry of Natural Resources, Lake Superior Manage. Unit, Tech. Rep. 99-1.

Stephenson, S.A., and Momot, W.T. 2000. Threespine, *Gasterosteus aculeatus*, and fourspine, *Apeltes quadracus*, sticklebacks in the Lake Superior basin. Can. Field-Naturalist **114**(2): 211-216.

Swackhamer, D.L., Pearson, R.F., and Schottler, S.P. 1998. Toxaphene in the Great Lakes. Chemosphere **37**: 2545-2561.

Todd, T.N. 1981. Allelic variability in species and stocks of Lake Superior ciscoes (Coregoninae). Can. J. Fish. Aquat. Sci. **38**: 1808-1813.

Thurston, C.E. 1962. Physical characteristics and chemical composition of two subspecies of lake trout. J. Fish. Res. Board Can. **19**: 39-44.

Twohey, M.B., Heinrich, J.W., Seelye, J.G., Bergstedt, R.A., Fredicks, K.T., Kaye, C.A., Scholefield, R.J., McDonald, R.B, and Christie, G.C. 2003. The sterile-male release technique in Great Lakes sea lamprey management. J. Great Lakes Res. **29**(Supple. 1): 410-423.

Vander Zanden, M.J., Hulshof, M., Ridgway, M.S., and Rasmussen, J.B. 1998. Application of stable isotope techniques to trophic studies of age-0 smallmouth bass. Trans. Am. Fish. Soc. **127**: 729-739.

Watson, N.H.F., and Wilson, J.B. 1978. Crustacean zooplankton of Lake Superior. J. Great Lakes Res. **4**: 481-496.

Wilberg, M.J., Hansen, M.J., and Bronte, C.R. 2003. Historic and modern abundance of wild lean lake trout in Michigan waters of Lake Superior: implications for restoration goals. North Am. J. Fish. Manage. **23**: 100-108.

Whittle, D.M., Kiriluk, R.M., Carswell, A.A., Keir, M.J., and MacEachen, D.C. 2000. Toxaphene congeners in the Canadian Great Lakes basin: temporal and spatial food web dynamics. Chemosphere **40**: 1221-1226.

88-3 Age structured stock assessment of Lake Erie walleye (report of the July 22-24, 1986 Workshop). 1988. R. B. Deriso, S. J. Nepszy, and M. R. Rawson. 13 p.
88-4 The international Great Lakes sport fishery of 1980. 1988. D. R. Talhelm. 70 p.
89-1 A decision support system for the integrated management of sea lamprey. 1989. J. F. Koonce and A. B. Locci-Hernandez. 74 p.
90-1 Fish community objectives for Lake Superior. 1990. Edited by T. R. Busiahn. 24 p.
90-2 International position statement and evaluation guidelines for artificial reefs in the Great Lakes. 1990. Edited by J. E. Gannon. 24 p.
90-3 Lake Superior: the state of the lake in 1989. 1990. Edited by M. J. Hansen. 56 p.
90-4 An ecosystem approach to the integrity of the Great Lakes in turbulent times (proceedings of a 1988 workshop supported by the Great Lakes Fishery Commission and the Science Advisory Board of the International Joint Commission). 1990. Edited by C. J. Edwards and H. A. Regier. 302 p.
91-1 Status of walleye in the Great Lakes: case studies prepared for the 1989 workshop. 1991. Edited by P. J. Colby, C. A. Lewis, and R. L. Eshenroder. 222 p.
91-2 Lake Michigan: an ecosystem approach for remediation of critical pollutants and management of fish communities (report of a round table sponsored in 1990 by the Great Lakes Fishery Commission, the Science Advisory Board of the International Joint Commission, and the Lake Michigan Federation). 1991. Edited by R. L. Eshenroder, J. H. Hartig, and J. E. Gannon. 58 p.
91-3 The state of the Lake Ontario fish community in 1989. 1991. S. J. Kerr and G. C. LeTendre. 38 p.
93-1 Great Lakes fish disease control policy and model program. 1993. Edited by J. G. Hnath. 38 p. Protocol to minimize the risk of introducing emergency disease agents with importation of salmonid fishes from enzootic areas. 1993. Edited by R.W. Horner and R. L. Eshenroder. 15 p.
94-1 The state of Lake Superior in 1992. 1994. Edited by M. J. Hansen. 110 p.
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95-1 Fish-community objectives for Lake Huron. 1995. R. L. DesJardine, T. K. Gorenflo, R. N. Payne, and J. D. Schrouder. 38 p.
95-2 The state of Lake Huron in 1992. Edited by M. P. Ebener. 140 p.
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03-2 Fish-community goals and objectives for Lake Erie. P.A. Ryan, R. Knight, R. MacGregor, G. Towns, R. Hoopes, W. Culligan. 56 p.
05-01 The state of Lake Michigan in 2000. Edited by M. E. Holey and T. N. Trudeau. 114 p.
05-02 The state of Lake Huron in 1999. Edited by M. P. Ebener. 140 p.
07-01 The state of Lake Ontario in 2003. Edited by B. J. Morrison and S. R. LaPan. 108 p.

Special Publications

79-1 Illustrated field guide for the classification of sea lamprey attack marks on Great Lakes lake trout. 1979. E. L. King and T. A. Edsall. 41 p.

82-1 Recommendations for freshwater fisheries research and management from the Stock Concept Symposium (STOCS). 1982. A. H. Berst and G. R. Spangler. 24 p.

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84-1 Recommendations for standardizing the reporting of sea lamprey marking data. 1984. R. L. Eshenroder and J. F. Koonce. 22 p.

84-2 Working papers developed at the August 1983 conference on lake trout research. 1984. Edited by R. L. Eshenroder, T. P. Poe, and C. H. Olver.

84-3 Analysis of the response to the use of "Adaptive Environmental Assessment Methodology" by the Great Lakes Fishery Commission. 1984. C. K. Minns, J. M. Cooley, and J. E. Forney. 22 p.

85-1 Lake Erie fish community workshop (report of the April 4-5, 1979 meeting). 1985. Edited by J. R. Paine and R. B. Kenyon. 58 p.

85-2 A workshop concerning the application of integrated pest management (IPM) to sea lamprey control in the Great Lakes. 1985. Edited by G. R. Spangler and L. D. Jacobson. 98 p.

85-3 Presented papers from the Council of Lake Committees plenary session on Great Lakes predator-prey issues, March 20, 1985. 1985. Edited by R. L. Eshenroder. 134 p.

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85-6 TFM vs. the sea lamprey: a generation later. 1985. 18 p.

86-1 The lake trout rehabilitation model: program documentation. 1986. C. J. Walters, L. D. Jacobson, and G. R. Spangler. 34 p.

87-1 Guidelines for fish habitat management and planning in the Great Lakes (report of the Habitat Planning and Management Task Force and Habitat Advisory Board of the Great Lakes Fishery Commission). 1987. 16 p.

87-2 Workshop to evaluate sea lamprey populations "WESLP" (background papers and proceedings of the August 1985 workshop). 1987. Edited by B. G. H. Johnson.

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88-1 Committee of the Whole workshop on implementation of the Joint Strategic Plan for Management of Great Lakes Fisheries (reports and recommendations from the 18-20 February 1986 and 5-6 May 1986 meetings). 1988. Edited by M. R. Dochoda. 170 p.

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