STATE OF LAKE ONTARIO IN 2003

SPECIAL PUBLICATION 07-01

The Great Lakes Fishery Commission was established by the Convention on Great Lakes Fisheries between Canada and the United States, which was ratified on October 11, 1955. It was organized in April 1956 and assumed its duties as set forth in the Convention on July 1, 1956. The Commission has two major responsibilities: first, develop coordinated programs of research in the Great Lakes, and, on the basis of the findings, recommend measures which will permit the maximum sustained productivity of stocks of fish of common concern; second, formulate and implement a program to eradicate or minimize sea lamprey populations in the Great Lakes.

The Commission is also required to publish or authorize the publication of scientific or other information obtained in the performance of its duties. In fulfillment of this requirement the Commission publishes the Technical Report Series, intended for peer-reviewed scientific literature; Special Publications, designed primarily for dissemination of reports produced by working committees of the Commission; and other (non-serial) publications. Technical Reports are most suitable for either interdisciplinary review and synthesis papers of general interest to Great Lakes fisheries researchers, managers, and administrators, or more narrowly focused material with special relevance to a single but important aspect of the Commission's program. Special Publications, being working documents, may evolve with the findings of and charges to a particular committee. Both publications follow the style of the *Canadian Journal of Fisheries and Aquatic Sciences.* Sponsorship of Technical Reports or Special Publications does not necessarily imply that the findings or conclusions contained therein are endorsed by the Commission.

COMMISSIONERS

April 2007

THE STATE OF LAKE ONTARIO IN 2003

Edited by

Bruce J. Morrison
Ontario Ministry of Natural Resources
Glenora Fisheries Station
RR#4, 41 Hatchery Lane
Picton, Ontario, Canada K0K 2T0

Steven R. LaPan
New York State Department of Environmental Conservation
Cape Vincent Fisheries Station
P.O. Box 292
Cape Vincent, New York 13618

Citation (general): Morrison, B.J., and LaPan, S.R. [EDS.]. The state of Lake Ontario in 2003. Great Lakes Fish. Comm. Spec. Pub. 07-01.

Citation (individual paper): Hoyle, J.A., Bowlby, J.N., Mathers, A., Schaner, T., Eckert, T.H., Casselman, J.M. 2007. The nearshore fish community. *In* The state of Lake Ontario in 2003. *Edited by* B.J. Morrison and S.R. LaPan. Great Lakes Fish. Comm. Spec. Pub. 07-01, pp. 45-58.

Special Publication 07-01

Great Lakes Fishery Commission
2100 Commonwealth Blvd., Suite 100
Ann Arbor, Michigan 48105-1563

April 2007

ISSN 1090-1051

Printed on recycled paper.
SP07-01/04-2007/1100

TABLE OF CONTENTS

EXECUTIVE SUMMARY

Progress has been made toward achieving objectives for some elements of Lake Ontario's environment and fish communities, but major deficiencies remain. Specific objectives should be drafted covering productivity, lower trophic-level organisms, physical habitat, and invasive species, and more-recognizable (measurable) end points should be provided for existing objectives. Productivity has decreased in recent years due to actions by jurisdictions within the basin to reduce phosphorus loading and to the filter-feeding of expanding populations of dreissenid mussels (zebra and quagga). Dreissenid mussels have changed some of the lake's physical habitat by increasing water clarity, which has resulted in expanded macrophyte beds. Dreissenid colonies encrust hard and soft natural habitat perhaps causing the decline in abundance of the amphipod *Diporeia* spp. Some embayments and coastal wetlands are degraded due to development, water-level control, and pollution by an expanding human population. Contaminant levels have declined in Lake Ontario fishes, but levels of PCBs, mirex, dioxin, and mercury are still high enough to trigger fish-consumption advisories. However, Lake Ontario remains an oligotrophic lake capable of providing an environment in which the fish-community objectives can be achieved. Fish-community objectives for the three fish communities (nearshore, offshore benthic, and offshore pelagic) and progress and deficiencies relating to achievement of those objectives as of 2003 are as follows:

The nearshore fish community will be composed of a diversity of self-sustaining native-fish species characterized by maintenance of existing walleye populations and expansion of walleye populations into favorable habitats, maintenance of existing yellow perch populations and expansion of yellow perch populations into favorable habitats, a population recovery of the lake sturgeon sufficient for its removal from New York's list of threatened species, population levels of smallmouth bass, largemouth bass, and sunfishes attractive to anglers, and increasing numbers of American eels consistent with global efforts for their rehabilitation.

The nearshore fish community remains diverse, self-sustaining, and mostly native, but abundance of some important fishes has declined. The Bay of Quinte walleye population, the largest in the lake, has declined in recent years likely due to increased water clarity caused by the filtering action of dreissenid mussels. The outlook for this population is stability but at 40% of the abundance in the early 1990s. There is no evidence of expansion of any walleye population. Yellow perch abundance has been variable—increasing along the south shore but decreasing in the eastern basin. Strong year classes have been produced in both areas in recent years, but predation by double-crested cormorants has caused declining abundance in eastern waters. Smallmouth bass abundance has also increased along the south shore but deceased in eastern waters. Predation by cormorants has been blamed for the decrease. American eel abundance has declined drastically since the early 1980s—implicated factors are loss of habitat, mortality in hydro turbines, commercial harvest, and changes in oceanic habitat. Lake sturgeon abundance remains low, and they are a threatened species in New York waters, but there have been increased catches of sub-adults in eastern waters and in the lower Niagara River. Largemouth bass and other centrarchids have increased in abundance in areas such as the Bay of Quinte. Warm summer water temperatures have resulted in better reproduction, and filtering by dreissenid mussels has increased water clarity resulting in the expansion of macrophyte beds, which provides more-favorable habitat for these fishes. Expansion of macrophyte beds has provided more-physical habitat for northern pike, but their numbers have declined likely due to loss of spawning habitat (water-level control impacts on coastal wetlands) and recreational-fishing mortality. The abundance of non-native white perch decreased in the late 1970s, has not recovered, and likely will not recover due to predation by walleye and loss of *Diporeia*, a major food item. Round gobies were discovered in the lake in the late 1990s and their population is spreading. This non-native species could have a major impact on the nearshore food web.

The offshore benthic fish community will be composed of self-sustaining native fishes characterized by lake trout as the top predator, a population expansion of lake whitefish from northeastern waters to other areas of the lake, and rehabilitation of native prey fishes.

Most fishes in this community are native and self-sustaining except that lake trout populations are maintained by stocking and rainbow smelt and sea lamprey are non-native. The lake trout is the top predator, but the number of adults is decreasing because of decreased first-year survival of stocked yearlings, which is likely due to predation by adult lake trout. The surviving adults have failed to establish self-sustaining populations due to egg and fry mortality—major factors causing this mortality are believed to be early mortality syndrome (EMS) and/or predation by round gobies and alewives. The abundance of burbot, the other native predator, has decreased since the mid-1990s. Reasons for this decrease are unknown, and the outlook for burbot is uncertain. Abundance of lake whitefish has likewise declined since the mid-1990s. This decline in abundance, a shift in distribution to deeper water, and decreased condition and growth have been linked to the decline in abundance of *Diporeia* following the expansion of dreissenid mussel populations. The major native prey species in the benthic food web is the slimy sculpin, and its abundance has declined during the 1990s because of predation by lake trout and reduced abundance of *Diporeia*. Of the remaining native benthic prey fishes, deepwater sculpin are extremely rare, and bloater, formally the dominant native prey fish, declined in abundance following expansion of alewife and rainbow smelt populations and now may be extirpated. The rainbow smelt is the major non-native prey fish in the benthic food web and second to alewife in the diet of stocked salmonines (trout and salmon), but its abundance declined to record low levels during 2000-2002 because of increased predation by salmonines. Sea lamprey control has successfully reduced the abundance of this pest such that the wounding rate on lake trout has been at or near the fish-community objective target (2 wounds per 100 fish) since the mid-1980s.

The offshore pelagic fish community will be characterized by a diversity of salmon and trout, Chinook salmon as the top predator, abundant populations of rainbow trout (steelhead), fishable populations of coho salmon and brown trout, populations of stocked Atlantic salmon at levels consistent with investigating the feasibility of restoring self-sustaining populations, amounts of naturally produced (wild) salmon and trout, especially rainbow trout, that are consistent with fishery and watershed plans, and a diverse prey-fish community with the alewife as an important species.

The predator-fish component of the offshore pelagic food web consists of the above-mentioned trout and salmon. Although their abundance is maintained largely by stocking, natural reproduction is evident for Chinook salmon, coho salmon, and rainbow trout. Trout and salmon stocking totaled near 9 million annually during the 1980s, but it has been reduced to near 5 million since 1992. The reduction was an attempt to restore predator-prey balance following a decrease in alewife abundance. Since the mid-1990s, harvest rates have increased slightly for Chinook salmon; have been stable for coho salmon, rainbow trout, and brown trout; and have decreased for Atlantic salmon. Increased natural reproduction may be the reason for increased harvest of Chinook salmon, whereas the decreased harvest of Atlantic salmon may be due to the shift from stocking yearlings to stocking mainly fry. The major problem with the offshore pelagic food web is the lack of prey-fish diversity. Although abundance of the native threespine stickleback appears to have increased in recent years, the non-native alewife is the most-abundant pelagic prey fish. Alewife abundance is now lower than in the 1980s, but alewives remain the primary prey for most predators in the lake. Consequently, reduced alewife abundance has resulted in reduced trout and salmon stocking, which has led to reduced fishing effort and lower harvest in the recreational fishery. Lake herring (cisco) historically were the dominant prey species, but they now exist only in remnant populations, and their restoration is unlikely.

INTRODUCTION

Goals and objectives for the Lake Ontario fish community (Stewart et al. 1999) were established as a result of the Joint Strategic Plan for the Management of Great Lakes Fisheries (Joint Plan) (Great Lakes Fishery Commission 1997). The Joint Plan charged the Lake Ontario Committee (LOC) to define objectives for the fish community and to develop means for measuring progress toward their accomplishment. The LOC is composed of fishery managers from the state of New York and the Province of Ontario. This state-of-the-lake report describes progress or lack of progress toward achievement of fish-community objectives in Lake Ontario as of 2003.

Achievement of water-quality goals is a prerequisite for achievement of fishery-related goals and objectives. The Great Lakes Water Quality Agreement (GLWQA) of 1972 (amended in 1987) obligates the governments of Canada and the United States to develop and implement Lakewide Management Plans (LAMPs), which address open-water critical pollutants, and Remedial Action Plans, which address specific Areas of Concern (AOCs).

Description of Lake Ontario and Fish Community

Lake Ontario is the seventeenth largest lake in the world with a surface area of 18,960 km^2 (Beeton et al. 1999) and a maximum depth of 244 m (Fig. 1). The lake receives 86% of its inflow from the upper Great Lakes via the Niagara River. Major habitat zones in the lake are offshore (>30-m deep), nearshore (<30 m-deep), and embayments (morphologically semi-isolated from the main lake). The relatively shallow Kingston Basin in northeast waters has numerous embayments and islands and includes >50% of the lake's shoreline. The major nutrient dynamic has been phosphorus loading, which peaked in the late 1960s and then decreased to a target level by the early 1980s in response to phosphorus management mandated by the GLWQA.

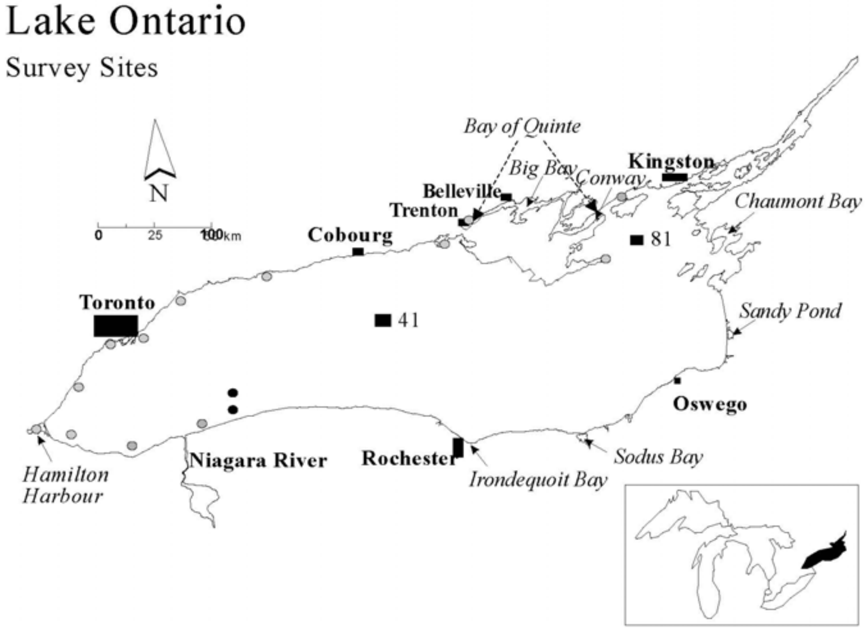

Fig. 1. Map of Lake Ontario referencing sampling locations. Light gray circles denote the Ontario Ministry of Natural Resources nearshore program; black circles denote the SUNY Brockport sampling program; and black rectangles (stations 41, 81, and 93) denote Department of Fisheries and Oceans Canada bioindex sampling sites.

An alphabetical list of the common names of fishes included in this report and their corresponding scientific names is presented in Table 1.

Table 1. An alphabetical list of common names of fishes in the text of this report and their corresponding scientific names.

Common Name	Scientific Name
Alewife	*Alosa pseudoharengus*
American eel	*Anguilla rostrata*
Atlantic salmon	*Salmo salar*
black crappie	*Pomoxis nigromaculatus*
Bloater	*Coregonus hoyi*
Bluegill	*Lepomis macrochirus*
Bowfin	*Amia calva*
brown bullhead	*Ictalurus nebulosus*
brown trout	*Salmo trutta*
Burbot	*Lota lota*
channel catfish	*Ictalurus punctatus*
Chinook salmon	*Oncorhynchus tshawytscha*
cisco(es)	*Coregonus* spp.
coho salmon	*Oncorhynchus kisutch*
common carp	*Cyprinus carpio*
Coregonines	*Coregonus* spp.
deepwater sculpin	*Myoxocephalus thompsonii*
freshwater drum	*Aplodinotus grunniens*
gizzard shad	*Dorosoma cepedianum*
Kiyi	*Coregonus kiyi*
lake herring (cisco)	*Coregonus artedi*
lake sturgeon	*Acipenser fulvescens*
lake trout	*Salvelinus namaycush*
lake whitefish	*Coregonus clupeaformis*
largemouth bass	*Micropterus salmoides*
longnose gar	*Lepisosteus osseus*
minnows	Cyprinidae
muskellunge	*Esox masquinongy*
northern pike	*Esox lucius*

Table 1, continued

Common Name	Scientific Name
Pacific salmon	*Oncorhynchus* spp.
pumpkinseed	*Lepomis gibbosus*
rainbow smelt	*Osmerus mordax*
rainbow trout (steelhead)	*Oncorhynchus mykiss*
rock bass	*Ambloplites rupestris*
round goby	*Neogobius melanostomus*
sculpins	*Cottus* spp.
sea lamprey	*Petromyzon marinus*
shortnose cisco	*Coregonus reighardi*
slimy sculpin	*Cottus cognatus*
smallmouth bass	*Micropterus dolomieu*
splake (male brook trout x female lake trout hybrid	*Salvelinus fontinalis* x *S. namaycush*
suckers	*Catostomus* spp.
sunfishes	Centrarchidae (esp. *Lepomis* spp.)
threespine stickleback	*Gasterosteus aculeatus*
trout-perch	*Percopsis omiscomaycus*
walleye	*Sander vitreus*
white perch	*Morone americana*
white sucker	*Catostomus commersoni*
yellow perch	*Perca flavescens*

Historically, the major native predators were lake trout, Atlantic salmon, and burbot, and the major native forage fishes were coregonines (ciscoes and whitefishes). The major factors leading to destabilization of the historical fish community were habitat loss or degradation (dams, eutrophication, and contaminants), over-fishing, and colonization by non-native species. Habitat loss, overfishing, and predation by sea lamprey were major factors prior to the 1970s. By the 1970s, Atlantic salmon, lake trout, burbot, deepwater sculpin, and some coregonines had disappeared or their populations seriously decreased in abundance, whereas abundance of non-native alewife, white perch, and rainbow smelt had increased (Christie 1972). Pacific salmon (mainly Chinook and coho salmon) were stocked beginning in 1968 to utilize the abundant alewife, provide a recreational fishery, and restore a predator-prey balance to the fish community (Owens et al. 2003). Efforts to stabilize the fish community continued in the early 1970s with the initiation of sea lamprey control (treating tributaries with lampricide) and of lake trout restoration (stocking hatchery-reared fish) (Pearce et al. 1980; Elrod et al. 1995). Although some predator-prey balance has been restored, the major predators and prey (Pacific salmon and alewife) are non-native species. Other exotic species have become established during the 1980s and 1990s and threaten the status quo. Some non-native trout and salmon have established naturalized populations but not in an abundance sufficient to satisfy demand in the recreational fishery. Lake trout have also reproduced in the lake but not sufficiently to establish self-sustaining populations. The alewife is the primary forage fish, and predator stocking has been reduced in recent years to maintain adequate populations. However, the alewife has played a role in the destruction of the original fish community and presents a threat, through competition and predation, to restoration of some native fishes (O'Gorman and Stewart 1999; Ketola et al. 2000). The rainbow smelt, although not as abundant as alewife, presents a similar threat to native fishes (Christie 1973). Proliferation of zebra and quagga mussels (*Dreissena polymorpha* and *D. bugensis*) has been linked to the collapse of native amphipod (*Diporeia* spp., hereafter, diporeia as a common name) populations during the 1990s. The round goby was found in the lake in 1998, and it likely will proliferate in Lake Ontario as it has in the other Great Lakes (Jude et al. 1992). Understanding food-web relationships associated with the current native and non-native fish community and lower trophic levels and predicting changes to these relationships brought about by management actions will be the challenge for agencies responsible for stabilizing the Lake Ontario fish community and satisfying the demands of stakeholders.

Goals and Guiding Principles

The goal for Lake Ontario conforms to the common goal presented in the Joint Plan for all of the Great Lakes—*to secure fish communities based on foundations of stable self-sustaining stocks, supplemented by judicious plantings of hatchery-reared fish, and provide from these communities an optimum contribution of fish, fishing opportunities, and associated benefits to meet needs identified by society for wholesome food, recreation, cultural heritage, employment and income, and a healthy aquatic ecosystem* (Great Lakes Fishery Commission 1997; Stewart et al. 1999). To achieve that goal, the LOC established 13 guiding principles and objectives for fish communities in the lake's fish-habitat zones (Stewart et al. 1999). The fish-habitat zones are based on food-web considerations and include the nearshore, offshore pelagic, and offshore benthic zones. A special conference focusing on progress toward achieving the Lake Ontario fish-community objectives was held at Niagara Falls, Ontario, in March 2003. This report is a compilation of some of the papers presented at this special conference. Because lower trophic-level productivity is integral to achieving the objectives for fish communities (Stewart et al. 1999), a section entitled productivity and limnology presents the status of nutrients, zooplankton, and invertebrate benthos, and it is followed by a section updating the status of contaminant levels in fishes. Double-crested cormorants, *Phalacrocorax auritus,* have been linked to declines in fish populations throughout the Great Lakes, and the effects of this predation and efforts to control their numbers in Lake Ontario are discussed. The final three sections of this report address the objectives for the nearshore, offshore benthic, and offshore pelagic fish communities. This report serves to focus attention on critical fisheries issues and to enhance communication and understanding among fishery agencies, environmental agencies, political bodies, and the public. Subsequent reports will be prepared every five years.

PRODUCTIVITY AND LIMNOLOGY

Ora E. Johannsson[1], Murray Charlton[2], Pat Chow-Fraser[3], Ron M. Dermott[1], E. Todd Howell[4], Joseph C. Makarewicz[5], E. Scott Millard[1], E. L. Mills[6], and Violeta Richardson[7]

Productivity regimes in Lake Ontario have responded to alterations in phosphorus loading, to habitat alteration, and, more recently, to the invasion of the lake by exotic invertebrates. Phosphorus was brought under control, and the lake had responded by the mid-1980s (Johannsson et al. 1998; Millard et al. 2003; Mills et al. 2003). Exotic invertebrates, especially dreissenids (zebra and quagga mussels) and a predatory cladoceran, are further altering nutrient flow, food-web structure, and productivity. This report summarizes our knowledge of present-day (late 1990s-2002) nutrient levels and the biomass/abundance, distribution, and productivity of the phytoplankton, zooplankton, and macrobenthos in the major habitat zones within the lake: embayments, nearshore (<30 m), an intermediate zone (30-100 m), and offshore (>100 m). This zonation follows temperature and structural-habitat gradients that strongly influence lower-trophic-level structure and productivity. Although there are as yet no specific fish-community objectives relating to nutrients and lower-trophic-level organisms in Lake Ontario, lower trophic levels are recognized as a major influence on the fish community (Stewart et al. 1999).

---

[1] Great Lakes Laboratory for Fisheries and Aquatic Sciences, Fisheries and Oceans Canada, 867 Lakeshore Rd., Burlington, Ontario, Canada, L7R 4A6

[2] National Water Research Institute, Environment Canada, 867 Lakeshore Rd., Burlington, Ontario, Canada, L7R 4A6

[3] Department of Biology, McMaster University, 1280 Main St. W., Hamilton, Ontario, Canada, L8S 4K1

[4] Environmental Monitoring and Reporting Branch, Ontario Ministry of the Environment, 125 Resources Rd., Toronto, Ontario, Canada, M9P 3V6

[5] Department of Environmental Science and Biology, SUNY College at Brockport, Brockport, New York, U.S.A., 14420

[6] Cornell Biological Field Station, University of Cornell, 900 Shackleton Point Rd., Bridgeport, New York, U.S.A., 13030

[7] Environmental Conservation Branch-Ontario Region, Environment Canada, 867 Lakeshore Rd., Burlington, Ontario, Canada, L7R 4A6

Nutrients, Chlorophyll *a*, and Phytoplankton

Embayments are the most-productive regions of the lake with higher levels of total phosphorus (TP), chlorophyll *a*, and primary production (PP). In the Bay of Quinte, Hamilton Harbour, Chaumont Bay, Sandy Pond, Sodus Bay, and Irondequoit Bay (Fig. 1), after the establishment of dreissenids in the early 1990s, TP levels ranged from 20 to 36 $\mu g \cdot L^{-1}$ and chlorophyll *a* concentrations generally averaged from 10 to 16 $\mu g \cdot L^{-1}$ during May-October (Klumb et al. 2003; MC, unpubl. data; ESM, unpubl. data). Irondequoit Bay had the lowest TP levels, and summer chlorophyll *a* was 5 $\mu g \cdot L^{-1}$ (Klumb et al. 2003). Chlorophyll *a*/TP ratios ranged from 0.25 to 0.43. May-October PP in the upper Bay of Quinte averaged 204 g $C \cdot m^{-2}$ over the post-establishment period (1995-2001), whereas, at the mouth of the bay (Conway), where the bay and lake waters mix, TP levels were lower (11.6 $\mu g \cdot L^{-1}$) and the PP averaged 138 g $C \cdot m^{-2}$ over the same period (Millard and Burley 2003; ESM, unpubl. data).

Embayments are some of the most-degraded regions in the lake due to shoreline alteration, high turbidity, and low oxygen levels in deeper reaches. This degradation altered production, shifting the community from a diverse, benthic-dominated system towards a more-pelagic system. Wetlands, both in embayments and along the shore, in the western end of the lake tended to be more degraded than wetlands in the eastern end of the lake (PCF, unpubl. data). In some embayments, e.g., the Bay of Quinte, dreissenids have increased water clarity, which has promoted the return of macrophyte beds and their associated communities (Seifried, 2000).

The nearshore zone (open waters <30-m bottom depth) is a physically and chemically dynamic environment with diverse biological communities. Dreissenids have colonized this region and exert a strong biological influence. Along the Canadian shoreline, water clarity, measured by Secchi disc, increased from 3-5 m in 1994 to 6-10 m in 1997, consistent with an increase in dreissenid abundance during this period (ETH, unpubl. data). Phosphorus and chlorophyll *a* exhibited no trend in concentration between 1994 and 2000 but variability increased. Concentrations in the nearshore zone were similar around the lake (Hall et al. 2003; JCM, unpubl. data; VR, unpubl. data). Phosphorus ranged from 6-17 $\mu g \cdot L^{-1}$, and chlorophyll *a* ranged from 1-5 $\mu g \cdot L^{-1}$. The total nitrogen/TP ratios were >30 and chlorophyll *a*/TP ratios were <0.5 (JCM, unpubl. data; ETH, unpubl. data). Chlorophyll *a* levels increased toward the offshore such that Secchi-disc readings at bottom depths <5 m were usually <3 m and ranged from 4.1 m to 17.5 m at bottom depths of 5-30 m. This pattern is likely due to dreissenid filtration. Record-low chlorophyll *a* levels occurred in the nearshore zone in spring as algal replacement could not match dreissenid filtration (ETH, unpubl. data). Replacement was depressed by the slow rate of algal growth at cold temperatures and by thermal bars, which restricted an influx of offshore water. This phenomenon undoubtedly has a major impact on the nearshore food web.

Examination of long-term trends suggests that spring TP concentrations in 1998 were lower in all zones in the main lake, while summer epilimnetic TP concentrations were lower in the intermediate and offshore zones than in the nearshore zone (Fig. 2; Hall et al. 2003; JCM, unpubl. data; VR, unpubl. data). When dreissenids invade a system, TP levels decline initially as the expanding mussel populations incorporate phosphorus and/or route it to the sediments. In the eastern and central basins of Lake Erie, TP rose again about six years after the dreissenid invasion (Charlton et al. 1999; MC, personal communication). The lack of an increase in TP levels in Lake Ontario could be associated with a continued increase and expansion of mussel populations (they now extend beyond a 100-m depth), a longer water-retention time in Lake Ontario (which would slow re-equilibrium of phosphorus with inputs), or decreased inputs. Formal loading estimates for phosphorus have not been calculated for Lake Ontario in recent years, and this hampers interpretation of water-quality findings.

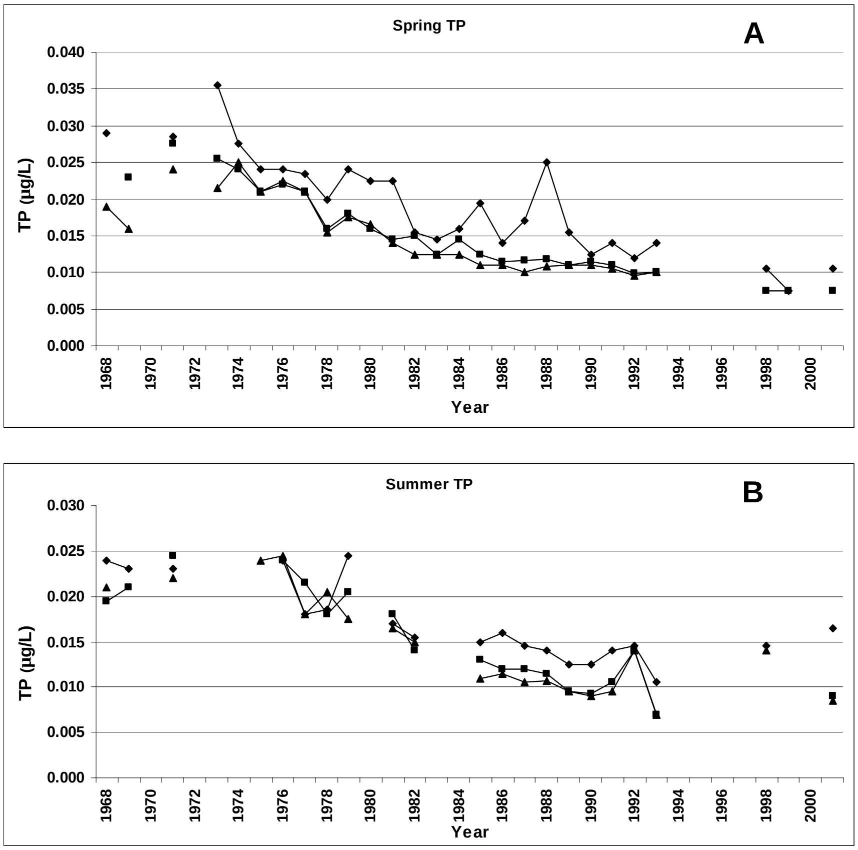

Fig. 2. Long-term trends (1968-2002) in spring (A) and summer (B) total phosphorus (TP) levels in Lake Ontario. Data are from the Environment Canada Surveillance Program. Samples collected at a 1-m depth in August and April from the three open-water habitat zones: nearshore (♦; <30 m), intermediate (■; 30-100 m), and offshore (▲; >100 m).

Offshore chlorophyll *a* was higher in the summer surveys of 1998 and 2001 than in surveys of the mid-1980s to early 1990s (Fig. 3; VR, unpubl. data). Phosphorus levels have not increased (Fig. 2), suggesting that grazing pressure on phytoplankton had decreased through this time period.

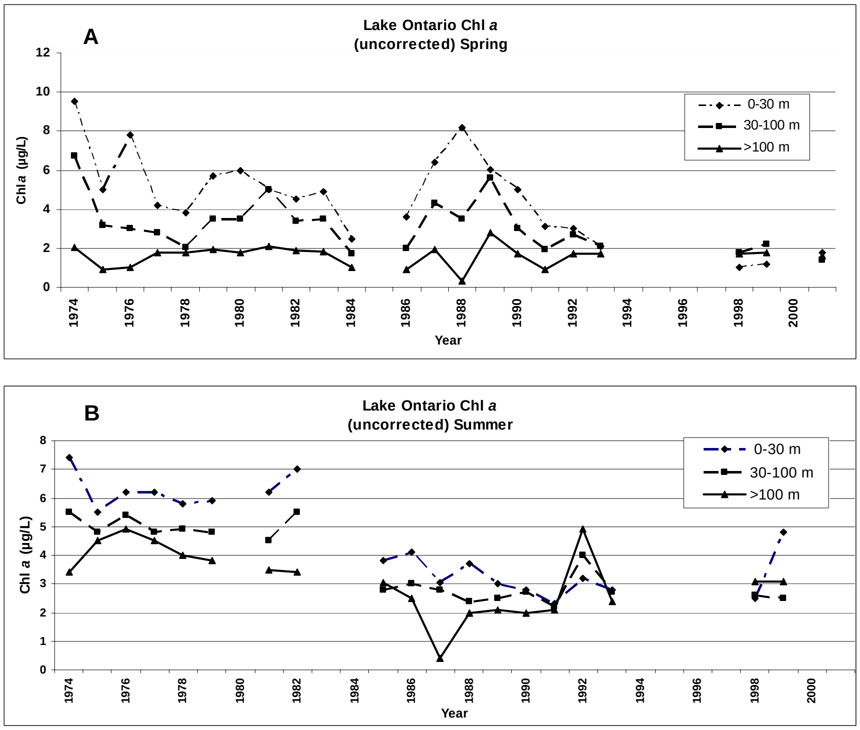

Fig. 3. Long-term trends (1974-2002) in spring (A) and summer (B) chlorophyll *a* (Chl *a*) levels in Lake Ontario. Data are from the Environment Canada Surveillance Program. Samples collected from a 0-20-m depth range in August and April from three open-water habitat zones: nearshore (♦; <30 m), intermediate (■; 30-100 m), and offshore (▲; >100 m).

Seasonal April-October primary productivity averaged 143 g C·m^{-2} in the Kingston Basin during 1987-1995 (Fig. 1, station 81), at the top of the intermediate zone, and 138 g C·m^{-2} in mid-lake south of Cobourg (Fig. 1, station 41) (Millard et al. 1996; Johannsson et al. 1998). No recent data are available, but PP is likely depressed as it depends on both algal biomass and TP.

Zooplankton

The predatory cladoceran, *Cercopagis pengoi,* invaded Lake Ontario in 1998, and its impact on the zooplankton community and effects on the food web are not fully understood. *Cercopagis* is small (the size of *Daphnia galeata mendotae*), inhabits all lake zones, and, unlike *Bythotrephes longimanus,* which invaded the lake in 1982, has not been controlled by fish predation. In offshore vertical profiles, *Cercopagis* occurred predominantly at the bottom of the epilimnion and top of the metalimnion, which suggests it may have trouble sustaining itself in clear, shallow waters (Benoit et al. 2002). Peak abundance occurred from mid-July to mid-August. Benoit et al. (2002) and Laxson et al. (2003) suggest that *Cercopagis* predation may reduce the abundance of smaller zooplankters, such as *Bosmina, Ceriodaphnia*, copepodids, and nauplii. Slightly higher chlorophyll *a* levels in the summers of 1998, 1999, and 2001 may indicate reduced grazing pressure and support the suggestion that *Cercopagis* can depress the abundance of these smaller herbivorous species.

Summer zooplankton populations develop first in embayments, which warm more rapidly and reach higher temperatures than the lake proper and are generally more productive. These zooplankters provide an early source of food for small fish, including young-of-the-year (YOY) and adult planktivores such as alewife and rainbow smelt. May-October mean zooplankton biomass was similar (0.22-0.26 g dry wt·m^{-3}) in the upper Bay of Quinte and in three bays on the south shore during 1995-2001 (Johannsson et al. 2003; Hall et al. 2003). Seasonal mean zooplankton biomass during 1995-1997 in the lower Bay of Quinte (0.07 g dry wt·m^{-3}) and in the southern nearshore zone (0.10 g dry wt·m^{-3}) were similar but slightly lower than that at offshore station 41 (0.17 g dry wt·m^{-3}) (Hall et al. 2003; OEJ, unpubl. data). Zooplankton abundance and biomass normally increase from nearshore to offshore as water depth increases (Johannsson et al. 1991). Individual zooplankters in the offshore zone were larger than in the nearshore or embayment zones, which is indicative of lower levels of fish predation offshore (Hall et al. 2003).

Following the establishment of dreissenid populations, zooplankton production in the intermediate zone of the Kingston Basin decreased in 1993-1995 to 12.9 g dry wt ·m^{-2}, which was comparable to levels observed in the offshore zone (12.5 to 24.3 g dry wt·m^{-2}) (Kuns and Sprules 2000; Johannsson 2003). Epilimnetic zooplankton production accounted for approximately 80% of total water-column production (Johannsson 2003). Production of the veliger larvae of dreissenids in the Kingston Basin is normally only a small percentage of epilimnetic zooplankton production in the main lake (<3%), but veliger production is variable and could, at times, be as high as 39% of epilimnetic zooplankton production (Johannsson 2003). No production data are available for 1996-2002.

The distribution, abundance, biomass, size-structure, and production of *Mysis relicta* were well-documented for Lake Ontario in 1990 (Johannsson 1995) and additional population data were obtained in 1995-1996 (OEJ, unpubl. data). Abundance and biomass increased rapidly with depth through the intermediate zone, and maximum densities were observed offshore. The whole-lake October survey in 1995 and a restricted survey in October 1999 off Oswego, New York (Fig. 1), detected no decrease in mysid abundance with the onset of the dreissenid population expansion. However, reports of the absence or low abundance of mysids along the south shore of the lake in the spring of 2002 (Robert O'Gorman, U.S. Geological Survey, 17 Lake St., Oswego, New York, 13126, personal communication) prompted a whole-lake survey in early November 2002. Mysid abundance at bottom depths >100 m was 49% lower in 2002 than in the early 1990s. Mysid whole-lake production in 1990 and 1995-1996 was 507 x 10^8 and 540 x 10^8 g dry wt, respectively (Johannsson 1995; OEJ, unpubl. data). If the decline in abundance in 2002 is real and not transitory, whole-lake mysid production also will have decreased.

Macrobenthos

Since its discovery in the lake in 1989, the zebra mussel has had profound effects in embayments, especially in the Bay of Quinte. The most-notable effect has been increased water quality and increased rate of expansion of macrophyte beds, particularly in the furthest reach between Trenton and Belleville, Ontario (Fig. 1; Seifried 2000). In high-macrophyte regions of the upper Bay of Quinte (Big Bay) in 2001, biomass of non-dreissenid benthic invertebrates was twice as high as in low-macrophyte areas and six times as high as in exposed mud areas (Bowen et al. 2003). Zebra mussels were most abundant (highest biomass) on the cobbled shores of the low-macrophyte areas and negligible in exposed mud areas. Each habitat supported a different benthic community; communities on the macrophytes, themselves, were not sampled (Dermott 2003). Oligochaete worms and chironomids comprised most of the exposed mud assemblage. Amphipods, caddisflies (Trichoptera), and snails (Gastropoda) were important in both high- and low-macrophyte communities, while flatworms (Turbellaria) and isopods (Isopoda) also contributed a significant amount to biomass in the high-macrophyte areas. Increased water clarity, expansion of macrophyte beds, and increased biomass of macroinvertebrates have likely occurred in other embayments, but, with the exception of the Bay of Quinte, Sodus Bay, and Toronto Harbour, information is lacking or limited. Further changes to the habitats and macroinvertebrate communities in embayments can be expected with the expansion of round goby populations and colonization of the exposed mud areas by quagga mussels.

As of 1995, information has been obtained for the nearshore benthic community on soft sediments and reefs near Olcott, New York (Hayes et al. 1999), and on soft sediments at the mouth of the Bay of Quinte (Dermott and Legner 2002) (Fig. 1). The composition of the benthic community at a 10-m depth off Olcott was primarily zebra mussels and secondarily the amphipod *Gammarus fasciatus* and oligochaete worms. The diversity of this community was similar to that at the same locations in 1983, although abundance of individual taxa was lower. How much variation is normal in these sparse, long-term comparisons and how much is related to changes in nutrient levels and the presence of zebra mussels is unknown.

Dense mats of the bacterium *Thioploca ingrica* developed at the mouth of the Bay of Quinte and in the Kingston Basin after the disappearance of the burrowing amphipod diporeia (Dermott and Legner 2002). As a consequence, benthic invertebrate biomass and species richness decreased. The invertebrate community in these areas is now predominantly oligochaete worms. The biomass of oligochaetes at a Kingston Basin site was 1.6 g dry $wt{\cdot}m^{-2}$ in 2000 compared to 3.6 g dry $wt{\cdot}m^{-2}$ for diporeia at this site in 1991. *Thioploca* has been observed at 28-146-m depths in Lake Ontario. However, since *Thioploca* appears to be successful only on very-fine soft sediments, it is unlikely to spread over large areas of the lake or come into competition with dreissenids.

By 2002, dreissenids had colonized most hard substrates out to a 100-m depth off of the north and south shores and had started to increase in abundance on softer sediments in the nearshore zone (Mills et al. 2003; M. Keir, Fisheries and Oceans Canada, 867 Lakeshore Rd., Burlington, Ontario, Canada, L7R 4A6, personal communication; ETH, unpubl. data). U.S. Environmental Protection Agency (USEPA) surveys in 1997 and 1999 indicated that the loss of diporeia, which began in eastern Lake Ontario during 1995, was expanding further offshore and westward. By 1997, diporeia had disappeared from much of the bottom at depths <80 m (Lozano et al. 2001). Abundance of diporeia at a 70-m-deep site off the Niagara River decreased from about $16{,}000{\cdot}m^{-2}$ in 1990 to 0 in 1997 and remained 0 in December 2002 (Dermott 2003; RMD, unpubl. data). The population of diporeia south of Cobourg at a depth of 125 m declined from $>5{,}000{\cdot}m^{-2}$ in

1996 to 100·m^{-2} in October 1999, but increased back to 1,800·m^{-2} by November 2002 (Fig. 4). This increase represents a return of this population to levels more typical of the deep profundal zone.

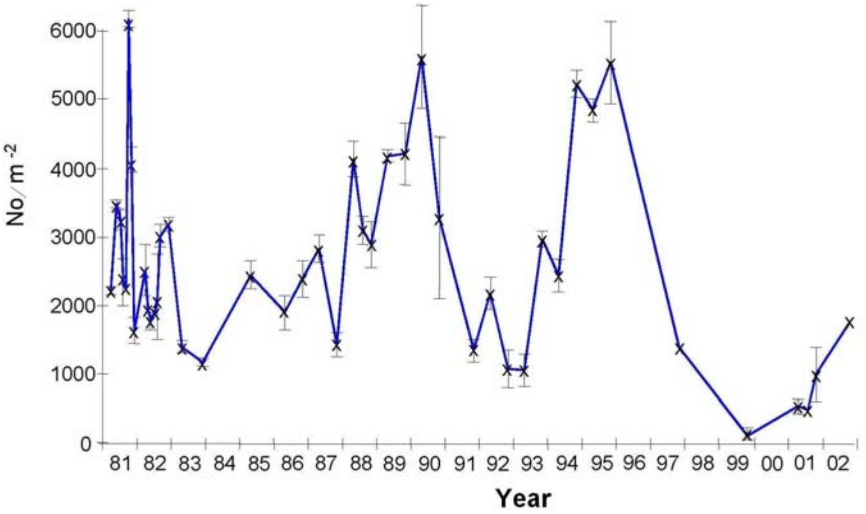

Fig. 4. Density of *Diporeia* spp. in Lake Ontario, 1981-2002, at a depth of 125 m at mid-lake station 41 (Fig. 1).

Wet-weight biomass of non-dreissenid macroinvertebrates in the lower Bay of Quinte and eastern Lake Ontario at a 30-m depth has decreased progressively since 1986 (Fig. 5; RMD, unpubl. data). This gradual decrease reflects the loss of diporeia, which disappeared in the Kingston Basin in 1995 and the loss of the fingernail clams *Sphaerium* and *Pisidium*. Decreased biomass of native macroinvertebrates was evident prior to the arrival of zebra mussels, indicating a response to decreased nutrients (phosphorus abatement).

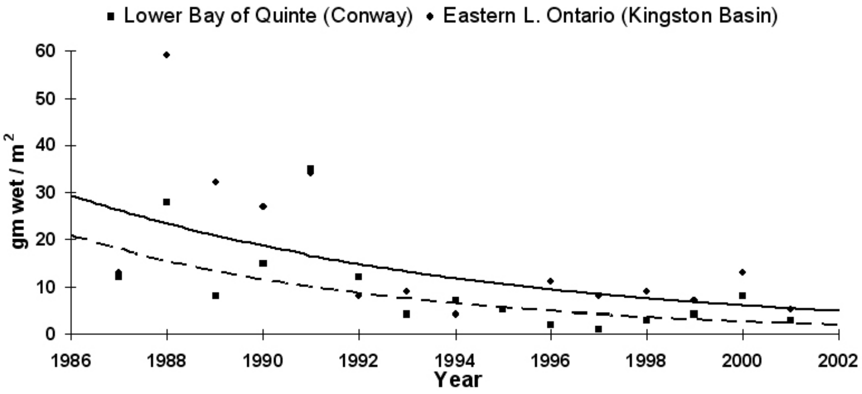

Fig. 5. Biomass of non-dreissenid macroinvertebrates (g·m^{-2} + shells, wet weight) in the lower Bay of Quinte (Conway; dashed line) and eastern Lake Ontario (Kingston Basin; solid line), 1986-2001 (RMD, unpubl. data).

Questions for the Future

Given that increased loadings, especially of phosphorus, can be expected in the future due to increases in the human population in the basin, to what degree will filtering activity of dreissenids buffer this loading? What are the consequences of the continuous increases in organic matter and shells on the bottom associated with dreissenid activity? Will other native invertebrates, besides diporeia, be lost from the ecosystem? With the continuing expansion of dreissenids into deeper water and the consequent changes in the macroinvertebrate community, including loss of diporeia and the decrease in mysids, what is the prognosis for the deepwater fish community? Climate change could have serious repercussions on the productivity of the food web. How will disturbances to the hypolimnetic temperature/oxygen regime (as predicted by some climate-change models), or alterations in the production of diatoms due to development of thermal stratification before the spring diatom bloom has developed, alter the production of zooplankton, *Mysis,* and diporeia?

Recommendations

Energy flow/production in ecosystems is envisaged generally in much too static a fashion. Processes such as production occur along gradients, and these phenomena are rarely incorporated into thinking or models: monitoring and modeling of these systems need to account for gradients. Overlaid on these gradients are distributional mechanisms—organisms or forces that move energy from one part of the gradient to another. The current biomonitoring programs on Lake Ontario do not adequately address questions of gradients. Managers and researchers need a coordinated monitoring program that is sensitive to existing pressures (nutrient loading and exotic species) and expected new pressures (human population expansion and climate change) and that can detect and assess change expeditiously. Surveys should be designed to account for seasonal and inter-annual variability. Benthos and mysids should be sampled once or twice a year on a broad spatial pattern, whereas zooplankton, phytoplankton, and water chemistry should be sampled every 1-2 weeks at a few representative sites.

ANTHROPOGENIC CHEMICAL CONTAMINANTS IN FISH

Frederick J. Luckey[1], Lawrence C. Skinner[2], Mike Whittle[3], Donald J. Williams[4], and Alan Hayton[5]

Environmental Impacts

The contribution of persistent toxic substances to the loss of certain Lake Ontario fisheries is unclear because fish populations were already severely degraded at the onset of significant contaminant releases in the 1950s. Changes in Lake Ontario nutrient levels, changes in fish populations, and the introduction of exotic species have obscured any impacts that contaminants may have had (Lakewide Management Plan 1998). Although a clear link between fish reproduction and contaminants was not identified at the time, some retrospective risk assessments suggest that contaminant levels during the 1970s may have been sufficient to prevent natural reproduction of some salmonids (Cook et al. 1997). However, current levels do not appear to have a measurable impact on fish reproduction, as successful hatchery culture of eggs taken from Lake Ontario trout and salmon suggest that they are capable of natural reproduction in the wild. Despite successful culture in hatcheries, the reestablishment of naturally reproducing populations of trout and salmon has been difficult. One factor inhibiting natural reproduction is the reliance on exotic alewife as the principal prey. Thiaminase, an enzyme that occurs naturally in alewife, induces a vitamin-B deficiency in trout and salmon that inhibits their ability to reproduce (Lakewide Management Plan 1998).

---

[1] U.S. Environmental Protection Agency, Communities & Ecosystem Protection Branch, 290 Broadway, New York, New York, U.S.A., 10007

[2] New York State Department of Environmental Conservation; Division of Fish, Wildlife and Marine Resources; 625 Broadway, Albany, New York, U.S.A., 12233

[3] Department of Fisheries and Oceans Canada, Canada Centre for Inland Waters, 867 Lakeshore Road, Burlington, Ontario, Canada, L7R 4A6

[4] Environment Canada, Ecosystem Health Division, 867 Lakeshore Rd., Burlington, Ontario, Canada, L7R 4A6

[5] Ontario Ministry of the Environment, Sport Fish and Biomonitoring Unit, 125 Resources Rd., Etobicoke, Ontario, Canada, M9P 3V6

Contaminant impacts on open-lake benthos are no longer a concern. Analysis of open-lake sediment cores indicates that concentrations of most of the regulated contaminants in sediments have declined significantly (e.g., PCBs, Fig. 6; Lakewide Management Plan 2002a). A 1997 USEPA-New York State Department of Environmental Conservation (NYSDEC) assessment of Lake Ontario surficial-sediment quality determined that, although contaminants in some samples exceeded the Ontario Ministry of the Environment's (OMOE) lowest effect levels (LEL) (the lowest concentration of a toxic contaminant that may affect sensitive benthic organisms), toxicity tests using pollution-sensitive benthic organisms showed no sign of toxicity. It was concluded that the percentage of samples that exceeded the LEL were sufficiently low, and that contaminant levels pose little or no acute toxic threat to pollution-sensitive benthic organisms (U.S. Environmental Protection Agency 2003).

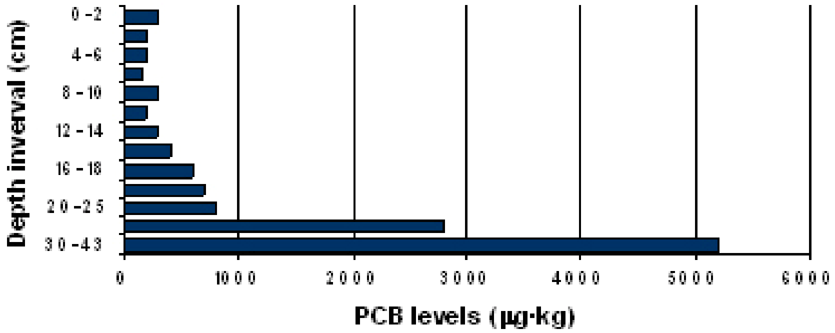

Fig. 6. Total PCB levels (µg·kg) in Lake Ontario sediment cores collected near the mouth of the Niagara River, 1995 (Lakewide Management Plan 2002a).

Although contaminants do not appear to be impacting the food web of the open lake, localized contaminant problems remain in some Lake Ontario harbors and embayments, which have been designated as AOCs and are a focus of ongoing remedial efforts. Furthermore, the current bioaccumulation rates in Lake Ontario's aquatic food web will need to be reexamined now that exotic species such as dreissenid mussels and exotic zooplankton have established and altered the flow of nutrients and contaminants.

Fish Contaminant Trends

Lake Ontario long-term (>25 years) fish contaminant monitoring programs have provided key information to assess potential fish, wildlife, and human-health contaminant issues, and these programs help track progress in reducing levels of these contaminants in the environment. Long-term, basinwide monitoring programs that measure whole-body concentrations of contaminants in top-predator fish (lake trout) and in forage fish (rainbow smelt) are conducted by the USEPA Great Lakes National Program Office (GLNPO) through the Great Lakes Fish Monitoring Program and by the Canadian Department of Fisheries and Oceans (DFO) through the Fish Contaminants Surveillance Program. DFO reports annually on contaminant burdens in 4- to 6-year-old lake trout and in rainbow smelt. GLNPO annually monitors contaminant burdens in 600- to 700-mm total length (TL) lake trout (Great Lakes National Program Office 2003). The NYSDEC, OMOE, and GLNPO monitor contaminant levels in edible fish fillets from a variety of sport fish for the purpose of publishing fish consumption advisories. Fish fillets are more representative of a typical meal serving and tend to have lower contaminant levels than whole-fish samples.

PCB levels in whole-body lake trout have declined since the late 1970s (Fig. 7; Lakewide Management Plan 2002b) and total DDT levels have declined since 1994 (Fig. 8; Lakewide Management Plan 2002b). Levels of both PCBs and total DDT in whole-fish samples of rainbow smelt have declined significantly since the most-recent peak in 1997 (Figs. 9, 10; Lakewide Management Plan 2002b), but there has been very little change in mean mercury levels since the mid-1980s (Fig. 11; Lakewide Management Plan 2002b). Declines in contaminant concentrations have also been observed in fish-fillet samples. Concentrations of total PCB, mirex, mercury, and total DDT in fillets from 50-cm coho salmon collected by the OMOE in Credit River spawning runs have decreased steadily since monitoring commenced in the late 1970s. Total PCB concentrations have declined from >1.5 ppm in the late 1970s to approximately 0.5 ppm in 2000. Concentrations of mirex declined from >0.1 ppm in the early 1980s to <0.05 ppm by 2000. Similar declines have been observed for mercury and DDT.

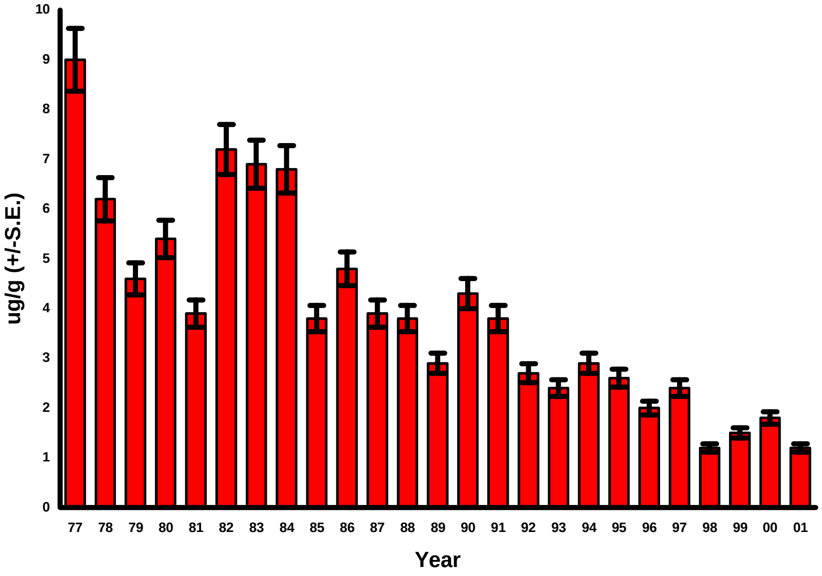

Fig. 7. Total PCB levels (µg·g wet weight, whole fish) in Lake Ontario lake trout, 1977-2001 (Lakewide Management Plan 2002b).

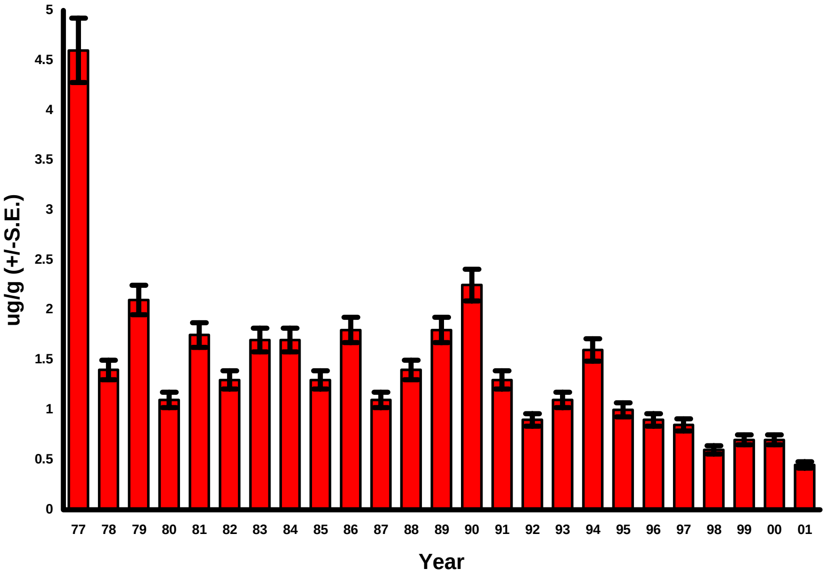

Fig. 8. Total DDT levels (µg·g wet weight, whole fish) in Lake Ontario lake trout, 1977-2001 (Lakewide Management Plan 2002b).

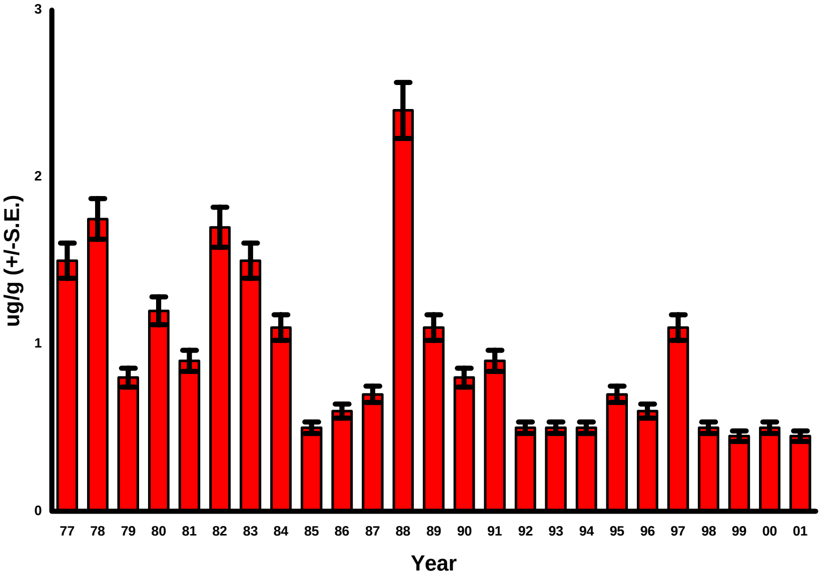

Fig. 9. Total PCB levels (µg·g wet weight, whole fish) in Lake Ontario rainbow smelt, 1977-2001 (Lakewide Management Plan 2002b).

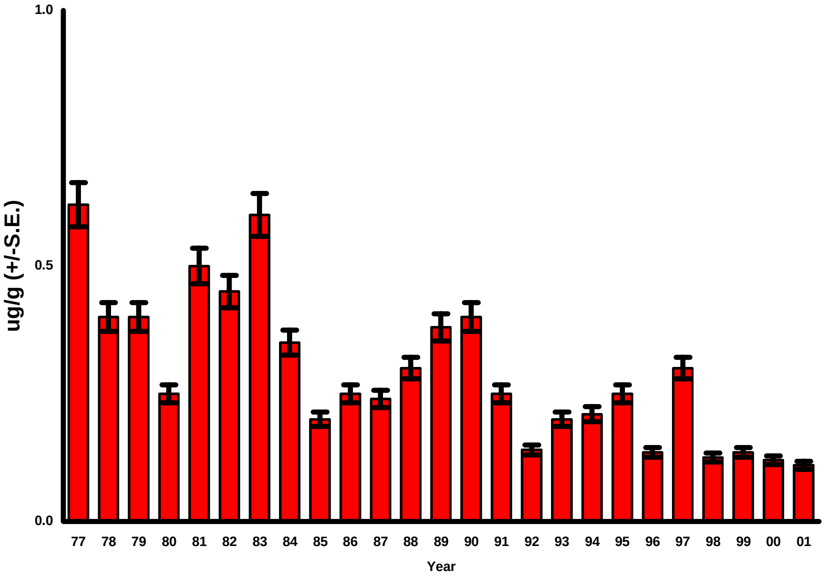

Fig. 10. Total DDT levels (µg·g wet weight, whole fish) in Lake Ontario rainbow smelt, 1977-2001 (Lakewide Management Plan 2002b).

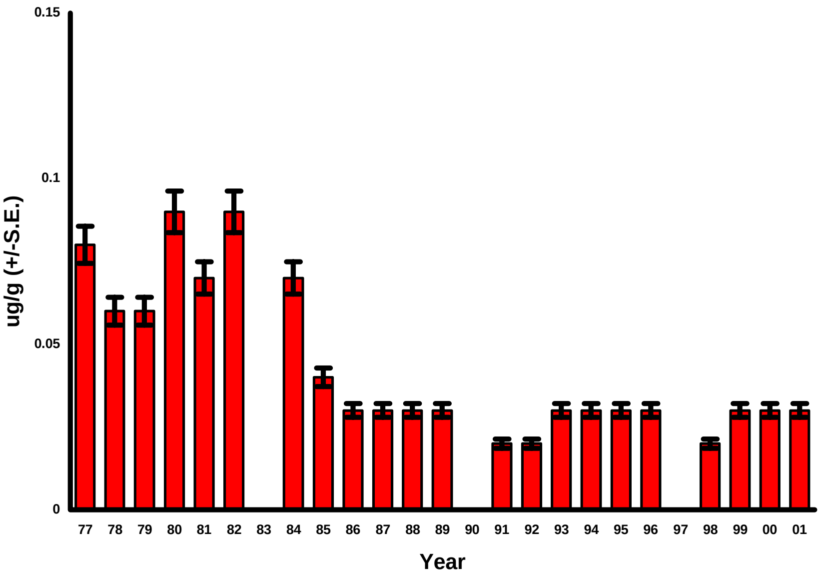

Fig. 11. Total mercury levels (µg·g wet weight, whole fish) in Lake Ontario rainbow smelt, 1977-2001 (Lakewide Management Plan 2002b).

Although the initial sharp declines seen in fish-contaminant body burdens are likely attributable to regulatory controls and remedial actions taken to address bioaccumulative contaminant sources (such as waste sites and sewage treatment plants), recent PCB mass-balance modeling suggests that current inputs of PCBs from anthropogenic sources have decreased to the point that they now contribute a negligible percentage of the total fish contaminant burdens. Models predict that fish-flesh PCB burdens are now largely controlled by the continued recycling of PCBs from sediments back into the water column (Limno-Tech, Inc. 2004).

Although there is no specific Lake Ontario fish-community objective relating to contaminants, the GLWQA has established criteria (whole-fish, wet-weight basis) for total PCBs, total DDT, and mercury for the protection of birds and animals that consume fish. The concentration of PCBs in Lake Ontario lake trout and rainbow smelt exceed the GLWQA PCB criterion (0.1 µg·g^{-1}) but have not exceeded the total DDT criterion (0.1 µg·g^{-1}) since 1995. Concentrations of total DDT or mercury in rainbow smelt have never exceeded the GLWQA criterion (0.5 µg·g^{-1}).

Fish Consumption Advisories

The Province of Ontario and New York State issue consumption advisories for fish caught in their waters. In general, the consumption advisories are based on contaminant levels in different species and ages of fish taking into account that contaminant levels are generally higher in older, larger fish. Although differences in the fish-tissue monitoring methodologies used by the two governments exist, both jurisdictions agree that environmental levels of PCBs, dioxin, DDT, and mirex require lakewide fish consumption advisories. Ontario's more-restrictive mercury advisory criterion at times has been exceeded. Ontario anglers should refer to the Guide to Eating Ontario Sport Fish published every two years for size and species-specific consumption advice (www.ene.gov.on.ca). U.S. anglers should refer to New York State Department of Health's Chemicals in Sportfish and Game (www.health.state.ny.us/nysdoh/environ/fish.htm).

PCBs, mirex, and dioxin/furans are responsible for lakewide consumption advisories for trout, salmon, channel catfish, common carp, and white sucker. Concentrations of PCBs and mirex in lake trout fillets tend to be higher in the western basin than the eastern basin, reflecting greater contaminant inputs from the upper Great Lakes, the Niagara River basin, and the industrialized nature of the western end of the lake. Mercury concentrations in smallmouth bass and walleye exceed Ontario's advisory criteria.

Lake Ontario Critical Pollutants

The four contaminants associated with lakewide fish-consumption advisories plus dieldrin, whose level exceeds criteria designed to limit bioaccumulation in fish, have been designated as critical pollutants requiring coordinated binational control actions by the U.S.-Canada Lake Ontario LAMP. PCBs and mirex were both widely used in the Great Lakes basin as flame retardants in a variety of electrical, mechanical, and manufacturing applications. Dieldrin and DDT were used as agricultural pesticides. Dioxins and furans are unwanted byproducts of chemical and combustion processes including the operation of internal-combustion engines, incinerators, and steel mills and the burning of wood and waste. Historically, manufacturing operations in the Niagara River basin were significant sources of these contaminants, but loadings from these sources have been effectively controlled since the 1970s. Mercury is a naturally occurring metal found in low concentrations in most soils and rocks and is used in a variety of medical and dental products, batteries, and the production of various synthetic materials, such as urethane foam. Coal-burning power plants continue to be a major source of atmospheric mercury emissions (Lakewide Management Plan 2002a).

Critical pollutant levels in Lake Ontario surface waters have generally declined over the last 20 to 25 years in response to regulatory and voluntary actions (Williams et al. 2001). Nevertheless, some of these pollutants are still detected in surface waters at levels that exceed the most-stringent criteria designed to protect wildlife and humans who consume fish (Table 2; Litten and Donlon 1998). Accurate measurement of contaminant loadings from tributary, wastewater, and atmospheric sources is very difficult because contaminant concentrations found in air and water are extremely low, and analytical results can be affected significantly by relatively minor factors in sample collection and analysis. Loading estimates are, therefore, best considered as rough approximations subject to considerable change as monitoring and analytical techniques improve. Nevertheless, some broad generalizations can be made regarding the relative importance of various contaminant inputs (Lakewide Management Plan 2002a). Surface water entering Lake Ontario via the Niagara River is the single largest overall source of critical pollutants. In addition to past releases from chemical facilities and hazardous-waste sites located along the Niagara River, the lake receives critical pollutants originating from the upstream Great Lakes. Atmospheric loads of critical pollutants, with the exception of PCBs, are roughly equal to combined tributary and wastewater loads. The estimated tributary and wastewater PCB load is approximately three times the atmospheric load. When loss of contaminants from the basin via atmospheric volatilization from the lake and surface water transport out of the basin via the St. Lawrence River are considered, the total mass of both PCBs and dieldrin in the lake is decreasing at a rate of >200 kg·yr (Lakewide Management Plan 2002a).

Table 2. New York water-quality standards and measured concentrations, in parts per quadrillion (pg·L^{-1}), for critical pollutants in Lake Ontario open waters, 1997 (Litten and Donlon 1998).

New York Water-Quality Standards		**Measured**
Critical pollutant	Standard (pg·L^{-1})	Concentration (pg·L^{-1})
PCBs	1	110
dioxins + furans	0.0006	0.0046
p,p' – DDE	7	10
p,p' – DDD	80	13
p,p' – DDT	10	2.6
dieldrin	0.6	51
mirex	1	—
dissolved mercury	700	—

In addition to critical pollutants, a number of other poorly regulated bioaccumulative contaminants have been recently recognized and detected in Great Lakes fish including polybrominated diphenyl ethers (PBDEs) and perfluoroctanesulfonate (PFOS). Concentrations of PBDEs have been increasing in Lake Ontario lake trout in recent years (Fig. 12; Luross et al. 2002). PBDEs have been used during the last two decades in the manufacture of brominated flame retardants for textiles, polyurethane foam, acrylonitrile butadiene styrene plastic (ABS), and electrical components. They are also used in many types of electrical equipment such as computers and televisions. PFOS is used in surfactants such as water-repellent coatings (i.e., Scotchguard®) and fire-suppressing foams). The potential health impacts of PBDEs, PFOS, and other newly recognized bioaccumulative contaminants are not well understood.

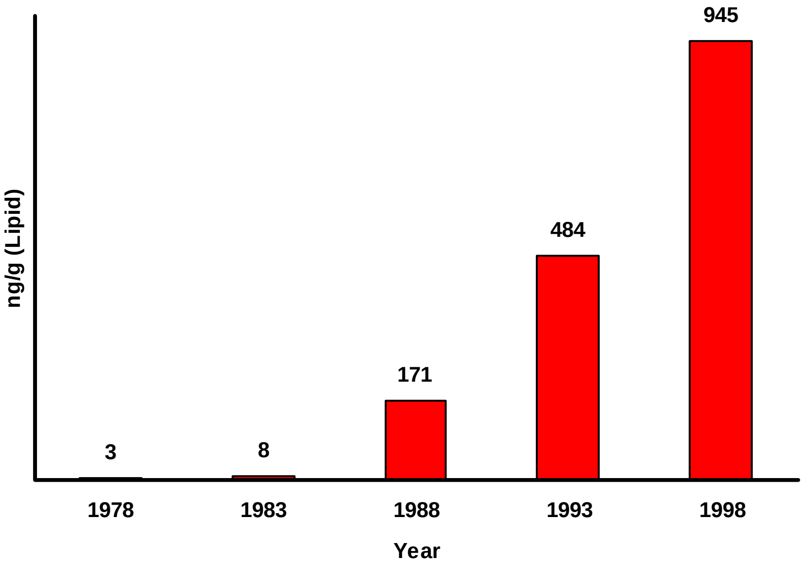

Fig. 12. Total PBDE (ng·g^{-1} lipid, whole fish) trends in Lake Ontario lake trout, 1978-1998 (Luross et al. 2002).

DOUBLE-CRESTED CORMORANTS—IMPACTS AND MANAGEMENT

Russell D. McCullough[1] and D.V. “Chip” Weseloh[2]

The double-crested cormorant *Phalacrocorax auritus* (hereafter, cormorant as a common name) nests across the continent in numbers estimated between 1 and 2 million birds (Hatch 1995) and is the only cormorant species that breeds in the Great Lakes. Cormorant numbers have increased tremendously on the Great Lakes over the past 30 years (Hatch 1995). They were first reported on Lake Superior in 1913, expanded eastward throughout the Great Lakes, and were first observed in Canadian waters of Lake Ontario in 1938 (Baillie 1947). The first reported breeding in New York waters occurred in 1945 at Gull Island (Miller 1998). The cormorant breeding population declined on all the Great Lakes from a peak of about 900 nests in 1950 to 114 nests in 1973 (Weseloh and Collier 1995; Weseloh et al. 1995). Cormorants disappeared from Lake Ontario in the 1960s (Price and Weseloh 1986). This decline, along with that of other fish-eating birds, was associated with high levels of toxic contaminants, particularly DDE and PCBs (Miller 1998). From 1970 to 1991, the breeding population in the Great Lakes increased from 89 nests to >38,000 nests, an average annual increase of 29% (Weseloh et al. 1995). By 1997, the number of nests had increased to approximately 93,000, an average annual increase of 22%. And, by 2000, the estimated number of nests was 115,000 (DVW, unpubl. data).

In Lake Ontario, cormorant numbers recovered through the 1980s and 1990s. Nest numbers increased rapidly from two at one colony in 1970, 9,190 at 11 colonies in 1991, and 28,180 nests at 23 colonies in 2002 (Price and Weseloh 1986; Weseloh et al. 1995; Weseloh and Pekarik 1999; DVW, unpubl. data). By 2000, there were more than 8,000 pairs of cormorants in the eastern basin alone nesting on five Canadian sites and one U.S. site (Weseloh et al. 2002). In addition, there were 1,166 nests at nine sites on the

---

[1] New York State Department of Environmental Conservation, 317 Washington Street, Watertown, New York, U.S.A., 13601

[2] Canadian Wildlife Service, 4905 Dufferin Street, Downsview, Ontario, Canada, M3H 5T4

St. Lawrence River in 2002. Factors contributing to the resurgence of cormorant populations include reduced levels of environmental contaminants, particularly DDT; increased food availability in breeding and wintering areas; and reduced human persecution (U.S. Fish and Wildlife Service 2001).

Little Galloo Island in New York waters of the eastern basin is one of the largest and most-intensively monitored cormorant nesting sites on the Great lakes. A colony of 22 nests on the island in 1974 grew quickly to 8,410 nests by 1996. Cormorant population reduction efforts (i.e., egg oiling) reduced the nest count to 5,440 by 2001 (Farquhar et al. 2002).

Little Galloo Island, Pigeon Island, and Snake Island in Ontario waters are home to the largest cormorant colonies in the eastern basin. The proliferation of cormorants in this area has created concern about the effects that these piscivorous birds may have on local fish populations (Schneider et al. 1999). Initially, concern regarded predation on stocked salmonines, mainly brown trout and lake trout. Cormorants were observed congregating at salmonine stocking sites, and diet studies confirmed that they were consuming recently released trout and salmon (Ross and Johnson 1995). Later, concern regarding cormorant predation expanded to natural fish populations, initially smallmouth bass and then yellow perch. Declines in smallmouth bass and yellow perch populations in the vicinity of Little Galloo Island have been associated with increases in cormorant abundance (Burnett et al. 2002; Lantry et. al. 2002).

The diet of Little Galloo Island cormorants was dominated by alewife and yellow perch during 1992-2001 (Johnson et al. 2002a). Temporal trends in diet composition in 2002 were generally consistent with 1992-2001. Within-year consumption patterns include the importance of pumpkinseed and threespine stickleback during the pre-chick period, the dominance of alewife during the chick period, and the increased importance of smallmouth bass in the post-chick period (Table 3). Reproductive output and survival of cormorants appears to be strongly associated with alewife abundance. Years of abundant alewife have also been years of high post-fledging survival (Weseloh and Ewins 1994).

Table 3. Seasonal (pre-chick, chick, and post-chick) and total percent diet composition of double-crested cormorants from Little Galloo Island, New York waters of Lake Ontario, 2002.

	Pre-Chick	**Chick**	**Post-Chick**	**Total**
Season	6/6-6/19	6/20-7/24	8/7-10/8	
No. of pellets	587	591	750	1,928
Fish/pellet (adjusted)	14.1	9.5	6.4	9.7
alewife	21.0	85.5	30.6	39.9
yellow perch	25.6	4.6	22.0	19.3
pumpkinseed	20.6	<0.1	4.4	11.7
threespine stickleback	9.3	0.7	0.5	5.2
cyprinids	8.5	1.6	10.6	7.2
rock bass	5.1	2.8	17.7	7.2
slimy sculpin	2.7	2.3	3.2	2.7
smallmouth bass	1.0	2.0	8.0	2.8
trout-perch	3.4	0.2	0.6	2.0
ictalurids	1.0	<1.0	1.1	0.8
catostomids	0.2	0.1	0.2	0.2
esocids	0.1	<0.1	0.2	0.1
white perch	0.1	<0.1	0.1	0.1
rainbow smelt	0.2	<0.1	—	0.1
burbot	<0.1	—	<0.1	<0.1
darters	1.2	0.1	—	0.7
gizzard shad	—	—	<0.1	<0.1
lake trout	<0.1	—	00	<0.1
American eel	<0.1	—	—	<0.1
banded killifish	<0.1	—	0.1	<0.1
coregonids	—	—	<0.1	<0.1
	100.0	100.0	100.0	100.0

The estimated consumption of 17 million fish by the Little Galloo Island colony in 2002 was the lowest since cormorant control measures were implemented on the island in 1999 (estimates for 1999-2001 ranged between 18 and 22 million). In 2002, Little Galloo Island cormorants consumed:

- 9.7 million forage fish: 6.8 million alewife, 1.2 million cyprinids (common carp and minnows), 0.9 million threespine stickleback, 0.5 million slimy sculpin, and 0.3 million trout-perch
- 6.5 million panfish: 3.3 million yellow perch, 2.0 million pumpkinseed, and 1.2 million rock bass
- 0.5 million game fish: almost all smallmouth bass with a few esocids and walleye

The percentage composition of their diet was 58% forage species, 39% panfish, and 3% game fish (Johnson et al. 2002a).

The main prey fish eaten in 2002 by cormorants from colonies at Snake and Pigeon Islands were threespine stickleback, yellow perch, alewife, cyprinids, rock bass, pumpkinseed, and slimy sculpin (Tables 4, 5). Round goby appeared in the diet of Snake and Pigeon Island cormorants for the first time in 2002 (Ross et al. 2003). Radiotelemetric tracking of cormorants from Pigeon and Snake Islands indicated that at least 44% of the population fed in New York's waters of the eastern basin and in the St. Lawrence River (Mazzocchi 2002).

Table 4. Seasonal (pre-chick and chick) and total percent diet composition of double-crested cormorants from Pigeon Island, New York waters of Lake Ontario, 2002. Cormorant pellets were sampled only during the pre-chick and chick feeding periods because cormorants abandoned the colony early due to predators.

	Pre-Chick	**Chick**	**Total**
Season	5/21	6/13, 7/16	
No. of pellets	150	187	337
Fish/pellet (adjusted)	27.7	7.4	16.4
threespine stickleback	65.9	10.7	54.8
alewife	5.0	77.1	19.5
slimy sculpin	12.1	1.4	9.9
yellow perch	6.9	3.6	6.2
cyprinids	3.5	3.6	3.5
rock bass	2.4	2.0	2.3
pumpkinseed	2.6	--	2.0
ictalurids	0.9	0.1	0.7
trout-perch	0.4	0.2	0.4
smallmouth bass	0.1	1.0	0.2
esocids	0.1	0.1	0.1
white perch	0.1	0.1	0.1
catostomids	0.1	—	0.1
rainbow smelt	<0.1	—	<0.1
round goby	<0.1	—	<0.1
	100.0	100.0	100.0

Table 5. Seasonal (pre-chick, chick, and post-chick) and total percent diet composition of double-crested cormorants from Snake Island, New York waters of Lake Ontario, 2002.

	Pre-chick	**Chick**	**Post-chick**	**Total**
Season	5/21	6/13, 7/16	8/15, 9/12	
No. of pellets	150	300	300	750
Fish/pellet (adjusted)	26.7	17.2	12.4	17.2
threespine stickleback	65.3	28.7	0.1	30.6
yellow perch	10.5	17.5	53.0	26.4
alewife	2.6	27.5	3.6	12.9
cyprinids	6.7	8.3	21.2	11.8
rock bass	2.2	9.2	8.0	6.8
pumpkinseed	6.3	1.2	3.5	3.4
slimy sculpin	3.4	4.7	1.6	3.4
round goby	0.1	0.4	4.5	1.6
ictalurids	2.2	0.7	1.7	1.4
smallmouth bass	0.1	1.4	2.2	1.3
esocids	0.2	0.1	0.3	0.2
trout-perch	0.2	0.2	<0.1	0.1
catostomids	0.1	0.1	<0.1	0.1
rainbow smelt	—	<0.1	0.1	<0.1
freshwater drum	—	<0.1	0.1	<0.1
lake trout	—	—	<0.1	<0.1
burbot	—	<0.1	<0.1	<0.1
bluegill	—	—	0.1	<0.1
	100.0	100.0	100.0	100.0

Cormorant diet studies were initiated in 1999 on Griswold, McNair, and Strachan Islands in the upper St. Lawrence River. Although smaller in size than colonies studied on Lake Ontario, populations at these three islands consumed an estimated 17.8 million fish from 1999 to 2001, of which 75% were panfish (Johnson et al. 2000a, 2001a, 2002b). This consumption has increased each year since 1999 (4.8 million fish in 1999, 6.3 million fish in 2000, and 6.6 million fish in 2001). Total consumption in 2001 included 3.3 million yellow perch, 1.2 million rock bass, 1.0 million cyprinids, 0.4 million pumpkinseeds, and 0.1 million smallmouth bass.

Impact on Fisheries

Consumption of sport fish by cormorants has affected sport fisheries and sport-fish populations, especially of smallmouth bass and yellow perch. Smallmouth bass harvest rates in New York waters of the eastern basin in the 1990s were 34-45% of those reported in the 1960s through the 1980s (McCullough and Einhouse 1999; Eckert 2002a). This decrease in harvest rate coincided with an increase in mortality of smallmouth bass and a rapid increase in cormorant numbers at nearby Little Galloo Island (Lantry et al. 2002). Despite the presence of four strong year classes, abundance of smallmouth bass has declined since 1990 (Eckert 2002b). Mortality of smallmouth bass (between ages 3 and 6) in the eastern basin increased substantially after 1988 and was associated strongly with three-year-mean cormorant nest counts that exceeded 3,500 (Weseloh and Ewins 1994). Smallmouth bass population projections demonstrated that observed age-specific cormorant predation could severely limit recruitment to the adult population. Predation on yellow perch by cormorants was also determined to be substantial and could affect the fishery. Based on a yellow perch standing stock estimate of 35 $kg \cdot ha^{-1}$ for eastern Lake Ontario, predation by Little Galloo Island cormorants could remove 54% of up to age-4 yellow perch (Burnett et al. 2002). Results from a standardized sampling program indicated that abundance of both smallmouth bass and yellow perch in New York waters of the eastern basin in 2000-2001 was 86% less than in 1976-1979, whereas smallmouth bass abundance increased 438% and yellow perch abundance increased 478% during the same period at a site (Pultneyville) farther west along the south shore (Eckert and Pearsall 2002). Fishery and environmental factors appeared to be similar in both areas, except that there was no concentration of cormorants at or near Pultneyville.

Cormorant colonies in the upper St Lawrence River also may be affecting nearby fish populations, and the Ontario Ministry of Natural Resources (OMNR) is investigating a large cormorant nesting colony at Presquile

Provincial Park for possible impacts on local fish populations in the Brighton and Bay of Quinte areas. Cormorants can also negatively impact other bird species through competition and habitat destruction at nesting and roosting sites. Such competition and habitat destruction can be particularly detrimental for species of special concern such as the black-crowned night heron *Nyctocorax nyctocorax* (U.S. Fish and Wildlife Service 2001).

Management of Cormorant Populations

In 1999, the NYSDEC began a five-year cormorant management plan on U.S. waters of the eastern basin of Lake Ontario. This plan involved egg oiling, nest removal, and habitat modification but no lethal control of adults or chicks. The overall goal of this plan was to reduce the number of nesting pairs to 1,500. The only cormorant control effort prior to this plan was a nest-removal program begun in 1994 on Gull, Bass, and Calf Islands. Large trees on Gull Island were removed to discourage cormorant nesting and to make nest removal efforts more effective. Two to five nest-removal efforts were required annually to successfully prevent reproduction. Cormorants have not reproduced successfully on these islands since 1994. Nesting attempts (including re-nesting) have varied from year to year with a peak of 1,368 in 2000.

Oiling of cormorant eggs with food-grade vegetable oil was done at Little Galloo Island from 1999 through 2003. All accessible nests (on or near the ground) were oiled three or four times annually. Hatching success for oiled eggs was about 1% each year, which was well below normal (Hatch and Weseloh, 1999). Overall production on Little Galloo Island, primarily from untreated nests, was less than 200 fledged young each year. Egg oiling reduced cormorant chick production on Little Galloo Island by approximately 94% and reduced the number of cormorant feeding days by 0.6 million. The resulting reduction in fish consumption was estimated at 0.3 smallmouth bass and 1.3 million yellow perch (Johnson et al. 2001b). Since 1999, the number of fish consumed and the biomass of fish consumed have been reduced by 48% and 46%, respectively, from the previous seven-year period. The Little Galloo Island colony consumed about 1.8 million pounds of fish in 2002. To reach the goal of 1,500 nesting pairs, oiling of all nests on Little Galloo Island would need to occur through 2008 and a less-intensive maintenance program beginning in 2009 would also be needed. Residual effects would carry into the year 2010 at which time the target population of 1,500 pairs would be achieved.

As demonstrated in New York waters, site-specific management actions can be effectively implemented to address cormorant problems at a local scale. The nest-removal and egg-oiling efforts in New York have exceeded expectations for limiting production of cormorants. These control efforts are meeting objectives for protecting nesting populations of black-crowned night heron on Bass and Gull Islands and have substantially reduced the consumption of smallmouth bass by cormorants that otherwise would have nested on Little Galloo Island (Johnson et al. 2000b).

Many of the cormorant colony sites and most of the Lake Ontario cormorant population are currently located in Ontario waters. Predicted changes in the eastern basin cormorant population after 2010 will depend on the level and type of management at these sites. In 2003, Ontario authorities began an experimental program of cormorant control at Presquile involving oiling of cormorant eggs and disruption of nests. Public consultation has been undertaken regarding possible expansion of the type and extent of the program. No cormorant management is currently planned for the St. Lawrence River by either Ontario or New York, so this cormorant population may continue to increase. Some private U.S. landowners along the river have taken action to control cormorants under federal permit. Private landowners in Ontario can also seek authorization under the Fish and Wildlife Conservation Act to take action for the protection of property.

THE NEARSHORE FISH COMMUNITY

James A. Hoyle[1], Thomas H. Eckert[2], James N. Bowlby[1], John M. Casselman[3], A. Mathers[1], and Ted Schaner[1]

For a description of fish communities, the Lake Ontario nearshore habitat zone is defined as shallow coastal waters <15 m deep plus all associated embayments (Stewart et al. 1999). Fish production is greatest and species composition is most diverse in embayment areas and in the more-protected eastern basin. Although most fishes make some use of the nearshore zone during some portion of their life cycle, the nearshore fish community is considered to be those warm-water and cool-water species that live primarily within the nearshore zone during the warmest months of the year. Six top predators are common in the nearshore zone: walleye, smallmouth bass, largemouth bass, northern pike, longnose gar, and bowfin. Other common species include gizzard shad, common carp, various species of minnows and suckers, brown bullhead, channel catfish, American eel, trout-perch, white perch, several sunfishes (pumpkinseed, bluegill, black crappie, and rock bass), yellow perch, and freshwater drum. The lake sturgeon, a formerly common but now rare species, has shown a modest resurgence in recent years.

The fish-community objectives for nearshore fishes as described in Stewart et al. (1999) are as follows: “The nearshore fish community will be composed of a diversity of self-sustaining native-fish species characterized by maintenance of existing walleye and yellow perch populations and expansion of walleye and yellow perch populations into favorable habitats; a population recovery of the lake sturgeon sufficient for its removal from New York’s list of threatened species; population levels of smallmouth bass, largemouth bass, and sunfishes attractive to anglers; and increasing numbers of American eels consistent with global efforts for their rehabilitation.”

---

[1] Lake Ontario Management Unit, Ontario Ministry of Natural Resources, Glenora Fisheries Station, 41 Fish Hatchery Lane, R.R. 4, Picton, Ontario, Canada, K0K 2T0

[2] New York State Department of Environmental Conservation, Cape Vincent, New York, U.S.A., 13618

[3] Queen's University, Department of Biology, 2406 Biosciences Complex, 116 Barrie Street, Kingston, Ontario, Canada, K7L 3NG

Weather, nutrient loading and cycling, and predation and competition by invasive species have had a major influence on the distribution and abundance of Lake Ontario's nearshore fishes. Climate warming (i.e., global warming) has caused increased water temperatures in nearshore waters (Casselman 2002) resulting in increased recruitment of centrarchids, and perhaps of gizzard shad and white perch. Recruitment of smallmouth bass and midsummer water temperatures are significantly and positively related (Casselman et al. 2002). Extremely cold winters can result in mortality, especially for gizzard shad and white perch. Nutrient loading has decreased in recent years, and the filtering activity of expanded dreissenid mussel populations has increased water clarity resulting in expansion of rooted aquatic-plant beds (macrophytes) (Johannsson et al., this report). Increased water clarity may be detrimental for walleye, but expansion of macrophyte beds has favored other fishes such as centrarchids.

The alewife, an offshore pelagic fish, is the most-abundant species in Lake Ontario and exerts a major influence on the nearshore fish community. It is a predator on and competitor with larval/juvenile stages of fishes such as yellow perch and is important in the diet of the nearshore piscivores, especially walleye and smallmouth bass. Predation by an avian predator, the cormorant, has been a major factor affecting abundance, especially of yellow perch and smallmouth bass (Burnett et al. 2002; Lantry et al. 2002).

Walleye

The walleye is the dominant piscivore in the Bay of Quinte and throughout eastern Lake Ontario's nearshore waters. Walleye exert a top-down influence on fish community structure because their diet is primarily planktivores, especially alewife (Hurley and Christie 1977; Hurley 1986; Ridgeway et al. 1990; Bowlby et al. 1991). The Bay of Quinte and eastern Lake Ontario (Fig. 1) walleye populations support important recreational, commercial, and aboriginal fisheries whose harvest ranged from 75,000 to 200,000 fish from 1984 to 2000 (Stewart et al. 2002). Other, much smaller, populations of walleye are associated with the Niagara, Oswego, and Black Rivers; Sodus Bay; Irondequoit Bay; and the Pickering Generating Station (Fig. 1; Bowlby et al. 1991). Marking/tagging studies indicate that juvenile walleye inhabit the Bay of Quinte (Fig. 1) year-round, and mature walleye migrate after spawning to the open waters of eastern Lake Ontario in spring and return to the bay in late fall (Payne 1963; Bowlby et al. 1991; JNB, unpubl. data).

Bay of Quinte walleye abundance has fluctuated considerably over the last several decades (Fig. 13) and appears to be governed by a combination of changes in water quality, physical habitat characteristics, and fish-community interactions—especially with exotic species. Walleye abundance decreased substantially in the 1960s due to excessive phosphorus loading and invasion of the white perch, a larval-fish predator (Hurley and Christie 1977). The walleye population recovered in the late 1970s and 1980s following control of phosphorus inputs (Minns and Hurley 1986) and major overwinter mortality of white perch (Hurley 1986; Bowlby et al. 1991; Eckert 2002a). Walleye abundance peaked in the early 1990s and declined notably after the invasion of dreissenid mussels in 1994.

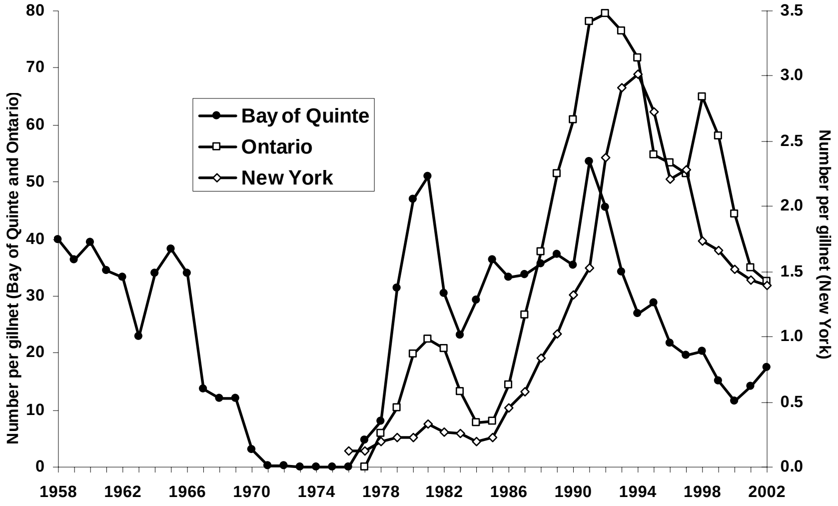

Fig. 13. Walleye abundance (indexed as a 3-year running average of number per standard gillnet) in the Bay of Quinte (1958-2002), in Ontario waters outside the Bay of Quinte (1977-2002), and in New York (1976-2002) waters of eastern Lake Ontario.

The recent decline in walleye abundance is consistent with increased water clarity due to the filtering action of dreissenid mussels and the resultant proliferation of macrophyte beds. This changed habitat is more suited to other species including the walleye's competitors and predators. Walleye year-class strength decreased after 1996 and now appears to have stabilized at a lower level consistent with current trophic and fish-community characteristics of the bay (Bowlby and Hoyle 2002; Schaner et al. 2002). Adult mortality has not changed significantly (Bowlby and Hoyle 2002), although the potential for an increase in mortality due to exploitation is high given the current low abundance and high demand by the various fisheries (Schaner et al. 2002; Stewart et al. 2002). The abundance of adult walleye is currently declining as weak year classes recruit. The short-term outlook is for a relatively stable population of about 400,000 fish three-years old and older as compared to a population of about 1 million a decade ago (Schaner et al. 2002).

Smallmouth Bass

Abundance of smallmouth bass has fluctuated in response principally to variations in year-class strength and predation. An analysis by Casselman et al. (2002) showed that a significant positive relationship existed between July-August water temperatures and smallmouth bass year-class strength. The strongest year classes were produced in years with the warmest temperatures, which are often associated with El Niño events. Significant increases in mortality of juvenile smallmouth bass in New York waters of the eastern basin were correlated with increases in predation by cormorants (Lantry et al. 2002). This predation reduced the abundance of even strong year classes, such as the one produced in 1995, before they recruited to the spawning population or the recreational fishery.

Smallmouth bass catch data from Ontario and New York waters of the eastern basin indicate high abundance from the 1970s through 1990 (Fig. 14). Variations in abundance during this period are attributable to variations in year-class strength, with the highest abundance due to strong 1973 and 1983 year classes. Abundance has declined since 1991, with the lowest catches occurring in 2000-2002 (Casselman et al. 2002; Eckert 2003a). This decline correlates with increased mortality of juvenile smallmouth bass and has occurred despite moderate to strong year classes produced in 1987, 1988, 1995, and 1997. Declines in relative abundance are corroborated by catch data from the New York recreational fishery (McCullough and Einhouse 1999).

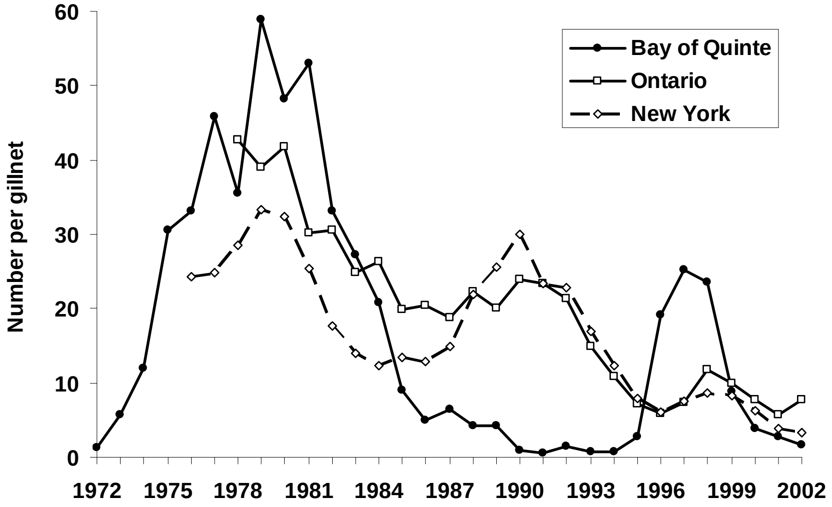

Fig. 14. Smallmouth bass abundance (indexed as a 3-year running average of number per standard gillnet) during July-August in eastern Lake Ontario (Ontario waters, 1978-2002; New York waters, 1976-2002) and in the Bay of Quinte, 1981-2002.

In contrast to declines in the eastern basin, smallmouth bass abundance along New York's southeastern shoreline has increased dramatically (Eckert and Pearsall 2002) as evidenced by record-high catch rates in the recreational fishery during 1998-2002. Specific reasons for the differences in smallmouth bass abundance between the two areas are speculative, but the most-apparent reason is likely the lower number of cormorants along the southeastern shoreline.

Smallmouth bass abundance in the Bay of Quinte was high in the late 1970s and early 1980s (Fig. 14). Abundance declined during the mid-1980s and early 1990s, increased during the mid- to late 1990s (due to a strong 1995 year class), but declined thereafter. No fish from the 1995 year class were caught after 2000 (Hoyle and Schaner 2002). Low abundance during the late 1980s and early 1990s may have been due to predation by the abundant walleye population. The increase in smallmouth bass abundance in the mid-1990s was associated with lower walleye abundance and favorable weather conditions (warm summers), but the factors responsible for the most-recent decline have not been identified.

Yellow Perch

The yellow perch is one of the most numerous and widespread nearshore species, and it feeds on a wide variety of prey items including benthic macroinvertebrates, zooplankton, and small fishes. The yellow perch is an important forage species for nearshore piscivorous fishes and cormorants. Yellow perch abundance in eastern Lake Ontario was very high in the late 1970s and early 1980s, declined to moderate levels during the mid-1980s, and has remained relatively stable thereafter (Fig. 15). These changes in abundance are mirrored by commercial harvests (Fig. 16). High abundance in the 1970s may have been related to elevated productivity levels associated with increased phosphorus loading and lower piscivore levels (Hurley and Christie 1977). Alewife predation on larval yellow perch is also considered important in structuring yellow perch populations (Brandt et al. 1987; O'Gorman and Burnett 2001). Following a massive alewife die-off in the winter of 1976-1977, yellow perch abundance increased considerably. As alewife abundance recovered, yellow perch abundance declined—recreational harvest in open-lake areas in New York waters was low during 1991-1995 (Eckert 2002b), and commercial harvest was relatively low in Ontario waters throughout the 1990s (Fig. 16). Yellow perch recreational harvest along the south shore has increased significantly since 1996 (Eckert 2002b) and coincides with changes in the distribution of alewife during winter and spring (O’Gorman et al. 2000) and with observed increases in age-0 yellow perch abundance in northeastern Lake Ontario (O’Gorman and Burnett 2001). Most recently, predation by cormorants appears to be a major cause of decreased yellow perch abundance in eastern Lake Ontario, negating the increases in age-0 abundance observed in the 1990s (Burnett et al. 2002). In the short term, yellow perch abundance will likely not increase in eastern Lake Ontario unless predation by cormorants is reduced.

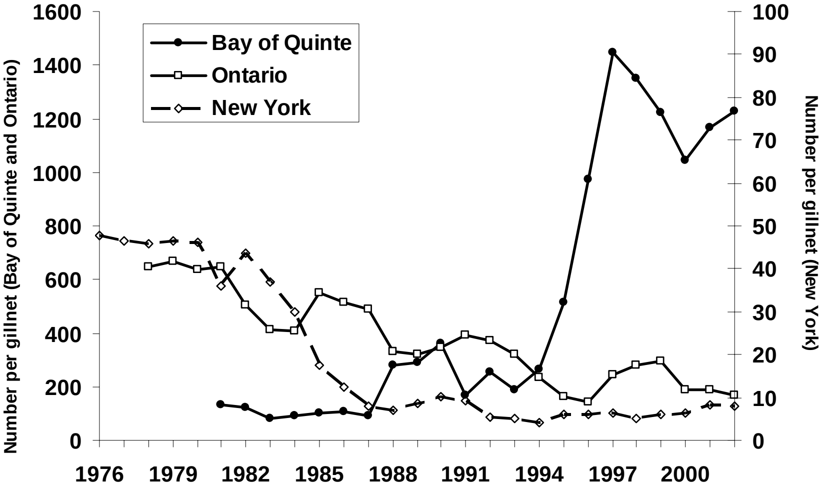

Fig. 15. Yellow perch abundance (indexed as a 3-year running average of number per standard gillnet) during July-August in the Bay of Quinte (1981-2002) and eastern Lake Ontario (Ontario waters, 1978-2002; New York waters, 1976-2002).

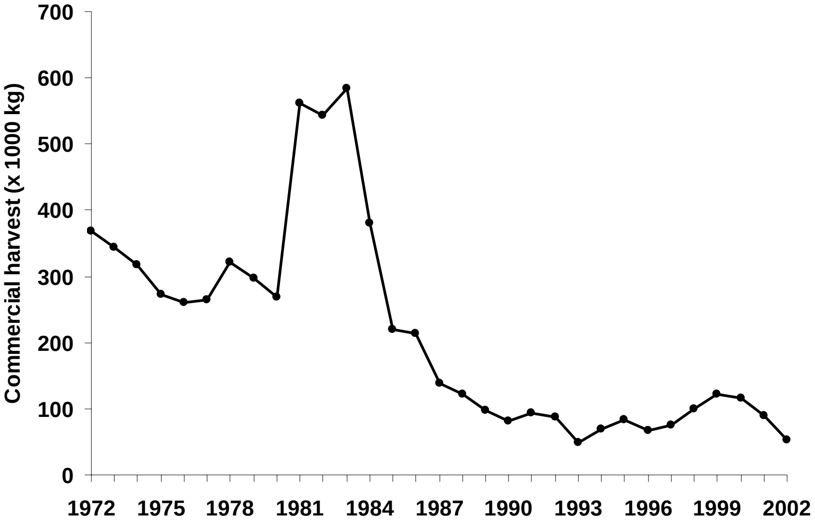

Fig. 16. Yellow perch commercial harvest (kg) in Ontario waters of Lake Ontario, 1972-2002.

Trends in yellow perch abundance in the Bay of Quinte contrast with those in the lake proper. Yellow perch abundance in the lake proper was low in the 1980s and increased during the mid-1990s following the invasion of dreissenid mussels and the related increase in aquatic macrophytes. Yellow perch use macrophyte beds for protection from predators. Although yellow perch abundance remains very high in the Bay of Quinte, the size structure is skewed toward small fish.

White Perch

The white perch is an exotic species that was first found in the lake in 1950 (Scott and Christie 1963). It proliferated throughout eastern Lake Ontario and the Bay of Quinte in the 1960s and 1970s when predators were scarce and productivity was high. In the Bay of Quinte, white perch suffered a severe die-off during the winter of 1977-1978 (Minns and Hurley 1986), and the population never recovered to its former level of abundance (Fig. 17). Coincident with the white perch die-off, walleyes produced an exceptional year class that led to a population recovery. These observations support the hypothesis that white perch negatively impacted walleye populations, apparently through predation on walleye fry. Walleye, once released from this predation pressure, rebounded, became the dominant predator in the Bay, and thereafter exerted more predation pressure on white perch. Disruption of the benthic food web may also be a factor keeping white perch abundance low in the Bay of Quinte. The white perch diet there once consisted primarily of the deepwater amphipod, diporeia. White perch abundance declined further after the disappearance of diporeia in 1993-1995 (Dermott 2001).

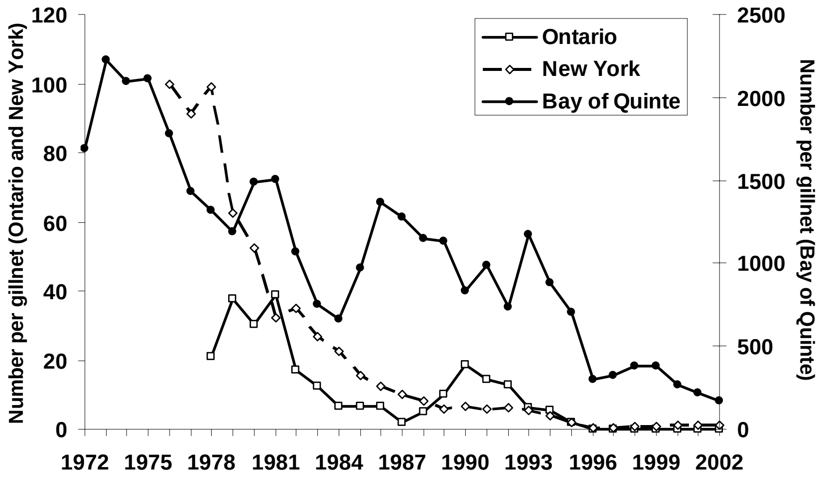

Fig. 17. White perch abundance (indexed as a 3-year running average of number per standard gillnet) during July-August in eastern Lake Ontario (Ontario waters, 1978-2002; New York waters, 1976-2002) and in the Bay of Quinte (1972-2002).

White perch abundance also declined considerably in other areas of Lake Ontario (Fig. 17), although the timing was different than for the Bay of Quinte, and impacts on other fishes were less apparent. In New York waters of the eastern basin, white perch catches in assessment nets and in the recreational fishery were high from 1976-1979, declined gradually from 1980-1994, and then suddenly dropped to near 0 by 1995 (Eckert 2002a; Eckert 2002b).

Largemouth Bass and Other Centrarchids

Largemouth bass, moderately abundant in many Lake Ontario embayments, have significantly increased in abundance in the Bay of Quinte recently—an event presumably related to the increase in suitable habitat following zebra mussel invasion (i.e., clearer water and more aquatic macrophytes) and warm summers. Other centrarchids that associate with aquatic plants have also flourished: pumpkinseed, bluegill, and black crappie. Rock bass and smallmouth bass abundance has declined in the bay.

American Eel

The American eel, a catadromous fish (spawns in salt water but matures in freshwater), has long been an important member of the fish community of the upper St. Lawrence River and eastern Lake Ontario. Subadult females live and mature in this freshwater environment for 5 to 32 years. Their average generation time is approximately 21 years. The American eel population is one genetic stock throughout its range (Labrador to the Caribbean); however, eels from the upper St. Lawrence River-eastern Lake Ontario watershed comprise a significant portion of the spawning stock (Castonguay et al. 1994; Casselman 2001).

An eel ladder at the Moses-Saunders Generating Station provides long-term partial counts for juvenile American eels migrating to the upper St. Lawrence River-eastern Lake Ontario population (Fig. 18). The numbers of juveniles passed peaked in 1982-1983, but numbers declined by three orders of magnitude from this peak to the 1990s, especially from 1986 to 1990. Passage of young juveniles virtually ceased in the late 1990s, and those few that now ascend the ladder are much older and larger (11 years, 493-mm TL) than in the 1970s and 1980s (6 years, 363-mm TL). Declines in all other indices of abundance (with appropriate lags) were synchronous with declines observed at the eel ladder (Casselman 2001).

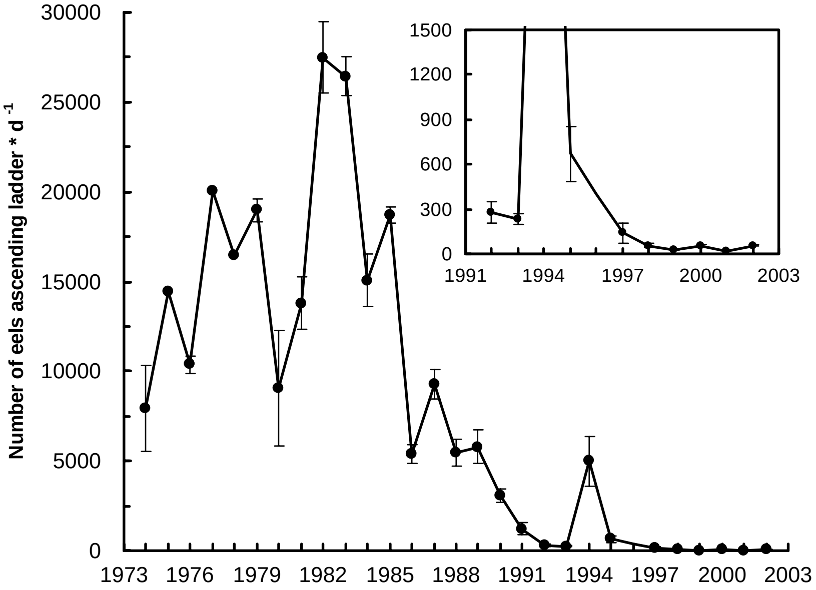

Fig. 18. Number of American eels per day ascending the eel ladder at the Moses-Saunders Dam in the upper St. Lawrence River during the 31-day peak migration period, 1974-2001. Inset shows the expanded scale for recent years. No counts were done in 1996.

The abundances of all life stages of American eel are declining in varying degrees throughout most of its range. In the commercial fishery for adult American eels, harvest exceeded 90,000 kg in most years during the 1970s, 1980s, and early 1990s (Fig. 19), but an unprecedented, precipitous decline in the commercial harvest began after 1993—harvest in 2002 was <5,000 kg.

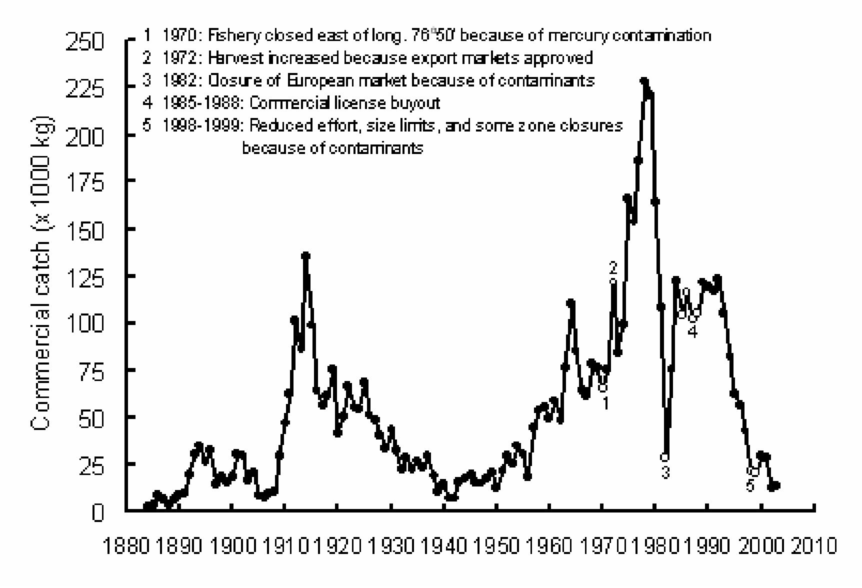

Fig. 19. Annual commercial catch of American eels in the upper St. Lawrence River-Lake Ontario during 1884-2000. Major events that could have affected the catch are indicated.

Factors implicated in the decline in abundance of American eels in the Lake Ontario-upper St. Lawrence River include loss of habitat, mortality in hydro turbines, commercial harvest, and changes in oceanic habitat. The correlation between the North Atlantic Oscillation Index (a measure of the northern extent of the Gulf Stream) and the recruitment index (eel ladder passage) for this population (lag of four years) suggests that changes in oceanic currents that carry larval eels from the spawning grounds may also be important (Castonguay et al. 1994; EIFAC-ICES Working Group on Eels 2001; Casselman 2001).

The outlook for American eels in Lake Ontario is bleak. The fish-community objective for Lake Ontario includes "increasing numbers of American eel consistent with global efforts for their rehabilitation" (Stewart et al. 1999). For management of this panmictic species to be effective, a coordinated, multijurisdictional approach encompassing the entire range of the species is required.

Northern Pike

Although less abundant than walleye, the northern pike is an important cool-water piscivore in the nearshore fish community of eastern Lake Ontario, particularly in the Bay of Quinte. Its larger esocid relative, the muskellunge, is present but relatively rare. Increased water transparency associated with the invasion of dreissenid mussels and the associated expansion of macrophyte beds has provided more-favorable physical habitat (Casselman and Lewis 1996). However, increasing water temperature in these nearshore waters is negatively affecting abundance and size of northern pike (Casselman 2002). In the Bay of Quinte, northern pike abundance and biomass peaked in the late 1980s, then declined in the 1990s. Northern pike is particularly vulnerable to angling pressure, and incidental catch in the walleye recreational fishery could be reducing its abundance and size potential. In addition, regulation of water levels has compressed natural fluctuations and altered water-level cycles, which may have reduced the amount and availability of wetland spawning and nursery habitat (Levels Reference Study Board 1993). The outlook for northern pike is not favorable given the negative factors described above.

Lake Sturgeon

The lake sturgeon was abundant historically in Lake Ontario, supporting a peak commercial harvest of over 225,000 kg in 1890, but abundance declined by the early 1900s to levels too low to support a commercial fishery. Although the lake sturgeon is often cited as the exemplar victim of overfishing (Christie 1973), the damming and degradation of tributary spawning areas likely were significant in its population decline. Commercial harvest of lake sturgeon was banned in New York waters in 1976 and banned in Ontario waters in 1978. In 1983, the species was listed as threatened in New York waters (Carlson 1995).

The lake sturgeon population may be recovering in parts of the lake. Prior to 1996, only two lake sturgeon were observed in the long-term assessment surveys in eastern Lake Ontario and the Bay of Quinte, but 24 subadult lake sturgeon have been caught since 1996. Commercial fishermen have also reported the incidental capture of up to 50 juveniles annually in eastern waters since 1996. Adult lake sturgeon have been observed during the spawning period in the Black (New York) and Trent (Ontario) Rivers during the mid-1990s. Abundance of lake sturgeon in the lower Niagara River also may be increasing. The total length of lake sturgeon from the river in 1998-1999 ranged from 337 mm to 1314 mm, indicating continued recruitment.

Increased incidental catches of lake sturgeon by recreational anglers and sightings by recreational SCUBA divers also suggest an increase in abundance.

Round Goby

Round gobies were first observed in western Lake Ontario in 1998 near the mouth of the Welland Canal and in eastern waters in 1999 in the Bay of Quinte. Round gobies are now spreading from these two areas. Round gobies likely will have a major effect on the nearshore ecosystem of Lake Ontario. They achieve high population densities, feed extensively on dreissenid mussels (French and Jude 2001; Taraborelli and Schaner 2002), are eaten by predators such as smallmouth bass, and may be an important new link in the nearshore benthic food web. Round gobies have been described as voracious foragers with a diverse diet (Taraborelli and Schaner 2002) and undoubtedly will compete with other fishes. Round gobies have also been linked to outbreaks of Type E botulism poisoning in piscivorous fishes and birds in Lake Erie since 1999 (Stone and Okoniewski 2002).

THE OFFSHORE BENTHIC FISH COMMUNITY

Brian Lantry[1], Ted Schaner[2], John Fitzsimons[3], James A. Hoyle[2],
Robert O'Gorman[1], Randall Owens[1], and Paul Sullivan[4]

The offshore benthic fish community in Lake Ontario is composed primarily of the slimy sculpin, lake whitefish, rainbow smelt, lake trout, burbot, and sea lamprey. Lake trout are the subject of a multiagency restoration program, and populations are currently maintained by stocking, whereas populations of the others are maintained through natural reproduction. Several historically important species are the now rare (deepwater sculpin), imperiled or possibly extirpated (bloater), or extirpated (kiyi and shortnose cisco) (Todd and Smith 1992).

Critically important changes in the benthic zone since the beginning of the 1990s include the introduction and proliferation of dreissenid mussels, the virtual disappearance of the amphipod diporeia, and colonization by round goby. Diporeia is an energy-rich invertebrate that previously accounted for 86% of total benthic production in Lake Ontario (Dermott 2001) and was important prey for most benthic fishes at some point in their life history. Round gobies are small benthic fish that compete with other small fishes, and they are aggressive egg and fry predators in rocky habitats used for spawning by lake trout and lake whitefish (Chotkowski and Marsden 1999; Charlebois et al. 2001).

---

[1] U.S. Geological Survey, Great Lakes Science Center, Lake Ontario Biological Station, 17 Lake Street, Oswego, New York, U.S.A., 13126

[2] Lake Ontario Management Unit, Ontario Ministry of Natural Resources, Glenora Fisheries Station, 41 Fish Hatchery Lane, R.R. 4, Picton, Ontario, Canada, K0K 2T0

[3] Fisheries and Oceans Canada, Great Lakes Laboratory for Fisheries and Aquatic Sciences, Burlington, Ontario, Canada, L7R 4A6

[4] Fisheries and Oceans Canada, Sea Lamprey Control Centre, Sault Ste. Marie, Ontario, Canada, P6A 6W4

The fish-community objectives for the offshore benthic fish community as described in Stewart et al. (1999) are: “The offshore benthic fish community will be composed of self-sustaining native fishes characterized by lake trout as the top predator, a population expansion of lake whitefish from northeastern waters to other areas of the lake, and rehabilitation of native prey fishes.” The status and recent population trends of the major offshore benthic fishes are presented below.

Slimy Sculpin

Slimy sculpins were still abundant in U.S. waters by the early 1980s (Wells 1969; Owens et al. 2003), but their abundance declined at depths <70 m after the early 1980s (Fig. 20). Trawling was discontinued after 1995 in western waters at depths <70 m, but data for shallow eastern U. S. waters indicated low but stable abundance during the 1990s and increased abundance during 2000-2002. The decline in shallower waters coincided with intensified annual stocking and greater abundance of juvenile lake trout (Owens and Bergstedt 1994). Slimy sculpins were the principal prey of juvenile lake trout (Elrod and O’Gorman 1991). In U.S. waters deeper than 70 m where juvenile lake trout were scarce, slimy sculpin abundance remained high through 1991 (Fig. 20). Prior to the early 1990s, slimy sculpin densities in southeastern Lake Ontario, east of Sodus Bay, New York (Fig. 1), were extremely high and probably at or near carrying capacity (Owens et al. 2003). After 1991, abundance west of Sodus Bay remained stable, but abundance east of Sodus declined drastically. This decline coincided with establishment of dreissenid mussels and the disappearance of diporeia in this part of the lake (Owens and Dittman 2003). Diporeia were a staple in the slimy sculpin diet (Owens and Weber 1995), and, after their disappearance, dead and emaciated slimy sculpins were taken in trawls near Oswego, New York. Unusual numbers of sculpin bones were found in ponar sediment samples during autumn 1992. At locations west of Sodus Bay, New York, there has been a 20% decline in average length of slimy sculpins since 1991, suggesting that food resources became limited after diporeia disappeared.

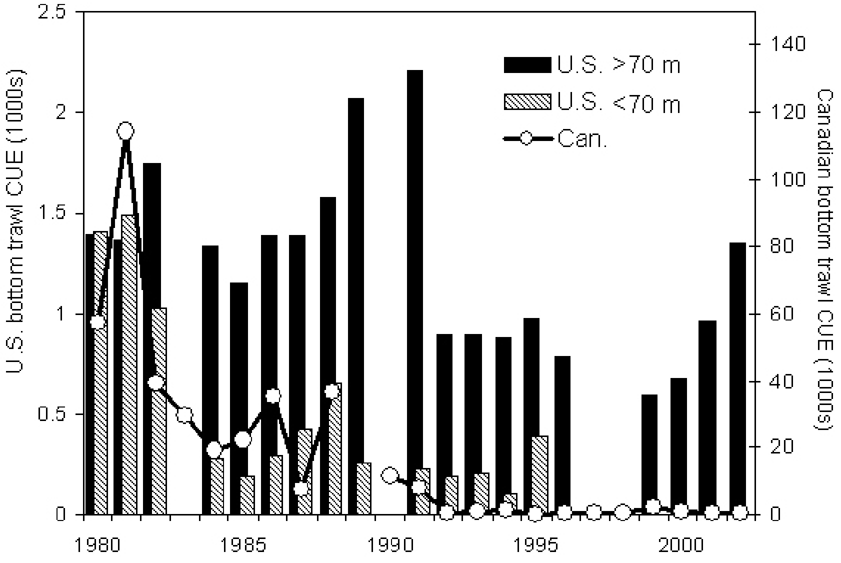

Fig. 20. Abundance of slimy sculpins in Lake Ontario indexed in bottom trawls, 1980-2002 (catch per unit effort = number per 10-min tow in U.S. waters and number per 12-min tow in Canadian waters of the eastern basin.).

In Canadian waters, trawling conducted in the eastern outlet basin at depths around 30 m corroborates the abundance trends described in shallow U.S. waters—a sharp decline in the early 1980s associated with increased predation by juvenile lake trout and a further drastic decline in 1992 coinciding with the establishment of dreissenid mussels (Fig. 20). The prospects for a full recovery of slimy sculpin in Lake Ontario are poor unless the abundance of lake trout declines substantially and the diporeia population recovers or is replaced with an energy-rich alternative prey. However, slimy sculpins may persist at low to moderate levels of abundance as a slight-to-moderate recovery in southeastern waters was evident in survey catches in 2001 and 2002.

Deepwater Sculpin

The deepwater sculpin was extremely rare by the 1950s. Only eight individuals were captured between 1961 and 1972, all in Canadian waters—five in the vicinity of Prince Edward County and three off Cobourg (Fig. 1). Several recent captures indicate that a small remnant population persists: three were captured in Ontario waters in 1996 and five were caught in U.S. waters during 1998-2000. The deepwater sculpin is one of several rare native species for which recovery may be possible. However, diporeia was a staple in the diet of deepwater sculpins (McAllister 1961), and recent declines in diporeia abundance at depths >100 m will likely limit the scope of any recovery that does take place.

Lake Whitefish

Historically, lake whitefish occurred throughout Lake Ontario but were most abundant in eastern waters. Commercial-harvest statistics suggest that lake whitefish abundance fluctuated widely throughout the 1900s (Christie 1963). Abundance declined in the 1960s (Baldwin et al. 1979), remained at low levels through the 1970s and early 1980s, increased during the mid-1980s to mid-1990s, and declined after 1993 (Fig. 21A; Hoyle et al. 2003; Owens et al. 2005).

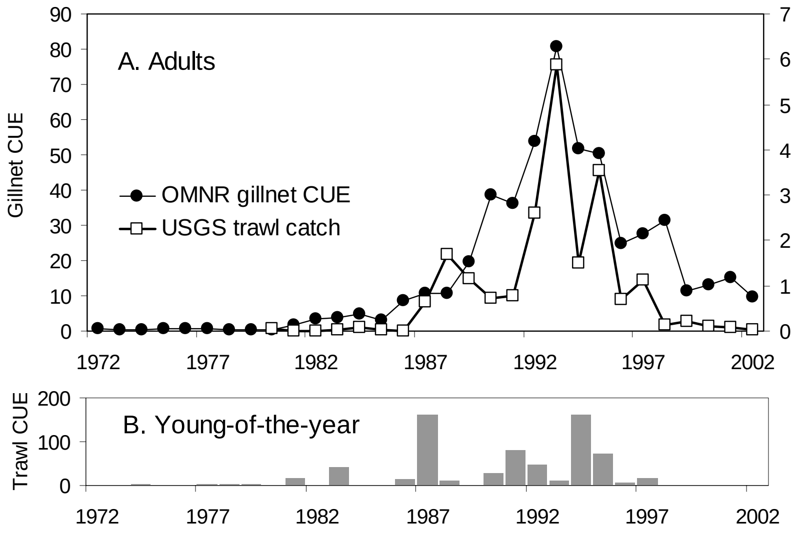

Fig. 21. Abundance of adult (A) and young-of-the-year (B) lake whitefish in Lake Ontario as indexed by catch per unit effort (CUE) in gillnets and trawls. Adults: CUE (number per 1000 m of graded-mesh gillnet) from the Canadian waters of eastern Lake Ontario (1972-2002), and total trawl catch (adjusted to a standard of 350 tows) from U.S. waters (1980-2002). Young of the year: CUE (number per 12–min tow) in Canadian waters (combined lake and Bay of Quinte indices) during 1972-2002.

A severe food-web disruption appears to be the principal cause of the most-recent lake whitefish population decline. Prior to the disruption, their diet was primarily diporeia but also included the opossum shrimp *Mysis relicta* and various gastropods and pelecypods (Hart 1931; Ihssen et al. 1981; Brandt 1986a; Christie et al. 1987; Owens and Dittman 2003; Hoyle 2005). Diporeia disappeared from eastern Lake Ontario between 1993 and 1995, and densities declined throughout the remainder of the lake at depths <100 m (Dermott 2001; Lozano et al. 2001). Diet studies in eastern waters in 1998 and 2001 confirmed that diporeia were absent and dreissenid mussels had become the primary food item in the lake whitefish diet (Hoyle 2005). Coincident with the disappearance of diporeia, the body condition and size of lake whitefish decreased (Fig. 22). Body condition has improved and size has stabilized since 1997, although both remain lower than prior to the disappearance of diporeia.

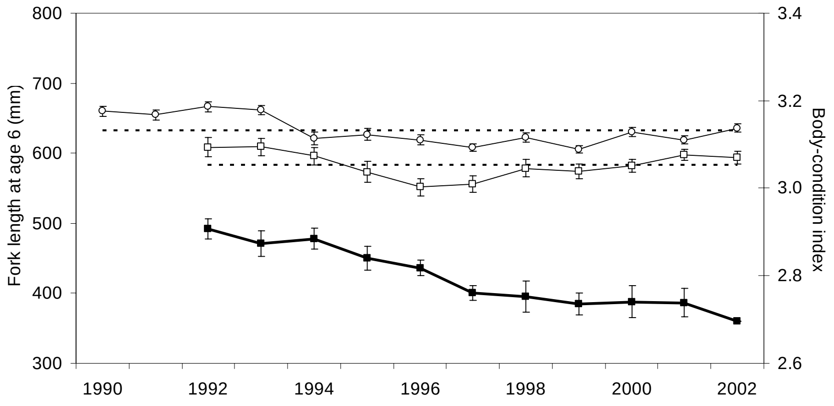

Fig. 22. Lake whitefish fork length from summer index gillnets (sexes combined, solid boxes) and body condition indexes (least squares mean log 10 round weight adjusted for differences in mean lengths among years, females only) in the Canadian waters of northeastern Lake Ontario, 1990-2002. The summer body condition (open boxes) is based on fish caught in index gillnets, and fall condition (circles) is based on fish captured in the commercial fishery. Error bars indicate +/- 2 SE; mean summer and fall condition values are also indicated (---).

Although size stabilized after 1997, the implied lower growth rate has resulted in an increase in mean age-at-maturity for female lake whitefish, from age 4 in the early to mid-1990s to age 6 or 7 by 2001. No strong year class has been produced in Canadian waters since 1995 (Fig. 21B), and there has been virtually no sustainable reproduction of lake whitefish since 1998. One hypothesis for the poor reproductive success is that poor body condition of adult fish, symptomatic of low lipid reserves, has reduced egg/fry viability. Resistance of fry to starvation may be related to egg quality (Johnston 1997) and could be particularly critical for lake whitefish, whose offspring hatch early in the spring prior to a pulse of zooplankton production.

Bloater

The last bloater observed in Lake Ontario was caught in a trawl in U.S. waters in 1983 (Owens et al. 2003). A recent bloater reintroduction effort was not successful in obtaining eggs from Lake Superior. This preliminary effort, however, provided important details about the timing of spawning of deepwater ciscoes and lake herring, suggested a disease-free status for Lake Superior bloaters (thus making this stock suitable as an egg source), and demonstrated the difficulty of separating the deepwater ciscoes by species during egg collection. The return of bloater or other deepwater ciscoes to the Lake Ontario fish community could provide a thiaminase-free food for lake trout. However, the decline in the abundance of diporeia may make reintroduction of bloaters more challenging because these invertebrates were a staple in the diet of bloaters (Pritchard 1931; Stone 1947; Wells and Beeton 1963).

Rainbow Smelt

Rainbow smelt are an important prey species and second only to alewife in the diet of stocked salmonines in Lake Ontario (Lantry 2001). The abundance of large smelt (>150 mm) declined sharply after 1983 (Fig. 23; Owens et al. 2003) likely due to predation by stocked salmonines. Based on catches from bottom trawls, relative abundance and biomass of yearling-and-older (YAO) rainbow smelt in U.S. waters fluctuated nearly tenfold during 1978-1999 and has been at record low levels during 2000-2002 (Fig. 24; O'Gorman et al. 2003). During 1978-1999, rainbow smelt abundance ranged from <300 to nearly 3,500 per trawl tow, and biomass ranged from about 1.2 kg to 11 kg per tow. Much of this variation could be attributed to the alternating pattern of strong and weak year classes likely resulting from cannibalism by yearlings on YOY (Lantry and Stewart 2000). The lakewide hydroacoustic assessment of YAO during 1997-2002 also indicated a decline in abundance (Fig. 24; Schaner and Lantry 2002). The hydroacoustic estimates have been below 500 million fish since 1998. Recent declines in alewife abundance and delayed shoreward movement of alewives in spring may be intensifying predation on rainbow smelt by salmonines. In addition, the bathymetric distribution of post-spawning rainbow smelt has shifted to greater depths, which may result in slower growth rates (O'Gorman et al. 2000). This shift in bathymetric distribution was associated with increased water clarity and the collapse of diporeia populations, both of which coincided with the establishment and rapid expansion of dreissenid mussel populations in the early 1990s. Rainbow smelt have not produced a strong year class since 1996, and low abundance of large rainbow smelt will likely persist if predation pressure remains high and their bathymetric distribution remains deeper than in the past. Based on the factors described above (increased predation, changes in distribution, and changes in the food web), the outlook for recovery of rainbow smelt populations is not promising.

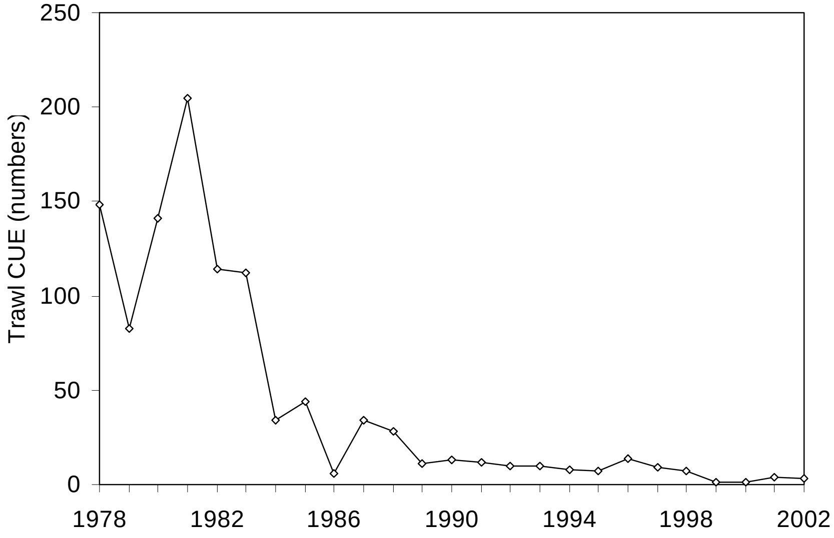

Fig. 23. Abundance (catch per unit effort = number per 10-min bottom-trawl tow) of large rainbow smelt (>150 mm, total length) in U.S. waters of Lake Ontario, 1978-2002.

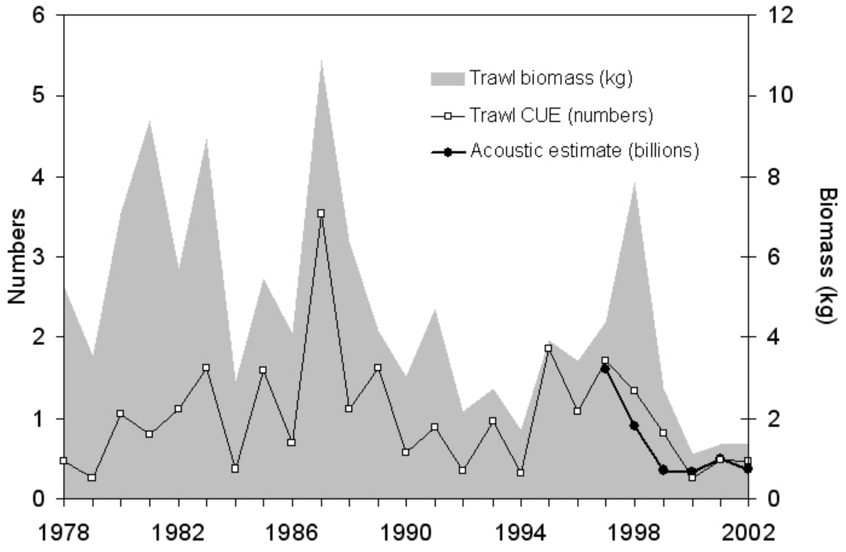

Fig. 24. Abundance of yearling and older rainbow smelt in Lake Ontario, 1978-2002. Bottom trawl data (biomass in kilograms and catch per unit effort in thousands per 10-min tow) are for U.S. waters; acoustic estimates (billions) are for the entire lake.

Lake Trout

Survival of stocked yearling lake trout during their first year in the lake, as measured by the number caught at age 2 in bottom trawls adjusted for number stocked, has declined since the 1980 year class was sampled in 1982 (Fig. 25). In U.S. waters, the annual adjusted catch of age-2 lake trout was greatest for the 1979-1982 year classes, declined by 32% with the 1983 year class, and again declined sharply starting with the 1990 year class. The latter decline is corroborated by data from Canadian waters where a large drop in survival up to age 3 occurred for the 1991-1994 year classes (Fig. 25). Predation by an increasingly abundant population of large adult lake trout may have been an important factor in the decreased survival of stocked yearlings (Elrod et al. 1993).

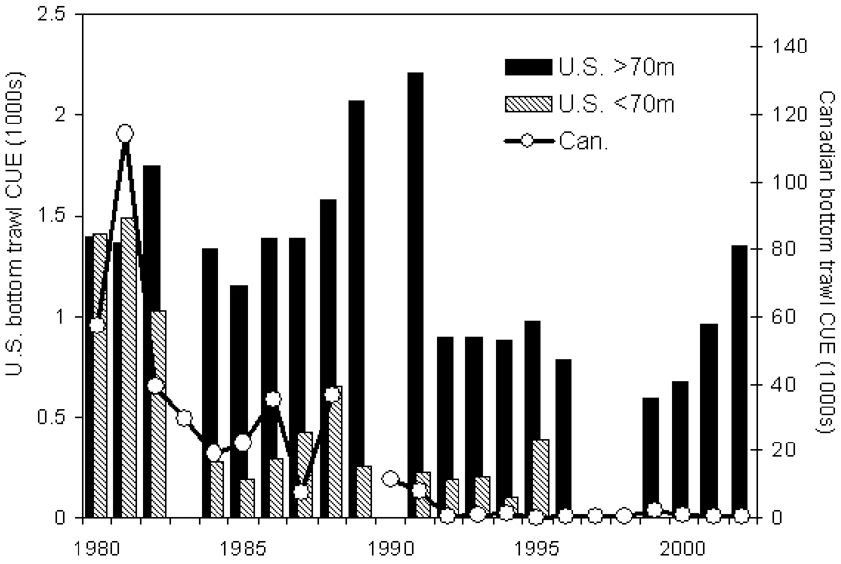

Fig. 25. First-year survival of the 1980-2000 year classes of lake trout stocked as yearlings in Lake Ontario and measured as the catch of juvenile fish corrected for numbers stocked. U.S. waters: total catch of age-2 fish in July bottom trawls per 500,000 yearlings stocked one year earlier in U.S. waters. Canadian waters: summer gillnet catch per unit effort in eastern Lake Ontario for age-3 fish per 500,000 yearlings stocked two years earlier.

Initially, abundance of adult lake trout reflected stocking history in Canadian waters and stocking and enhanced sea lamprey control in U.S. waters (Elrod et al. 1995). Abundance increased considerably in U.S. waters in the early 1980s and then declined during the late 1990s in both U.S. and Canadian waters (Fig. 26). The decline during the 1990s was the result of diminished survival of stocked fish and a 50% reduction in the number of fish stocked in U.S. waters beginning in 1992 and in Canadian waters beginning in 1993.

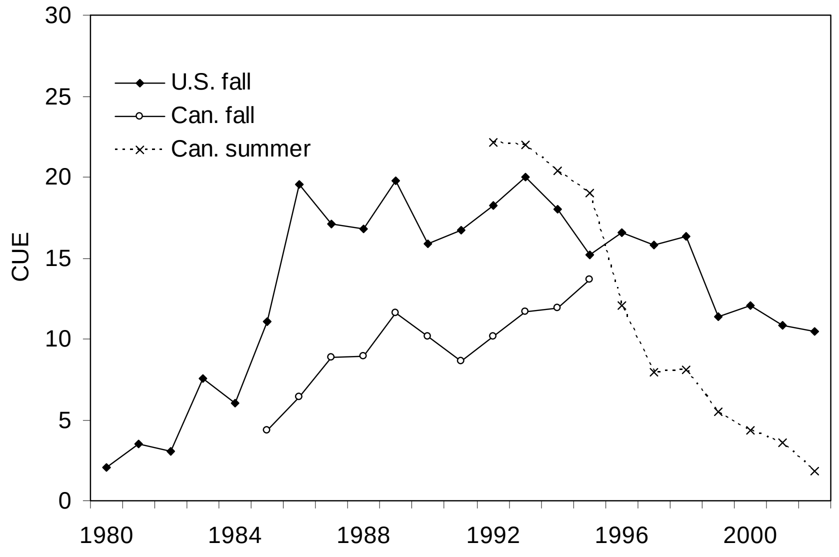

Fig. 26. Abundance of adult lake trout in Lake Ontario, 1980-2002 (catch per unit effort = number per gillnet; fall data are for all U.S. and Canadian waters; summer data are for eastern Canadian waters).

Each year, during 1994-2002, successive cohorts (1993-2001) of naturally produced yearling and older lake trout have been observed in assessment catches. The appearance of naturally produced yearling lake trout came at a time when population fecundity was peaking and coincided with an abrupt increase in the spring depth distribution of alewives (O'Gorman et al. 1998; O'Gorman et al. 2000; Lantry et al. 2003). Although the appearance of wild lake trout in survey catches is encouraging, abundance is low and not increasing. Achieving the goal of a self-sustaining population will require maintenance of a large population of hatchery-origin adults and overcoming the impediments preventing increased natural reproduction (Elrod et al. 1995; Fitzsimons et al. 2003).

Lake Trout Egg Deposition and Emergence in Southwestern Lake Ontario

Lake trout egg deposition and fry emergence have been monitored over a 10-year period at two sites in southwestern Lake Ontario (Fitzsimons 1995; Fitzsimons et al. 2003; JF, unpubl. data): annually since 1992 on a 4.5-m deep, man-made reef at Port Weller, and in 1992, 1996, and 1999 on a large natural reef (7.5-9.0-m depths) located approximately 1-km offshore of Stoney Creek, New York (egg deposition only, except in 1996 fry emergence was also monitored). Egg deposition at Port Weller increased from 1,420 eggs·m^{-2} in 1992 to a high of 5,771 eggs·m^{-2} in 1999, then declined by almost 50% to 3,141 eggs·m^{-2} in 2002. The decline corresponded to the appearance of round gobies in 2000. Even more striking than the decline in egg deposition after gobies appeared was the 95% decline in fry emergence, from 117 fry·m^{-2} in 1999 to 4 fry·m^{-2} in 2001. In 1992, egg deposition at Stoney Creek (103 eggs·m^{-2}) was less than one-tenth that at Port Weller. Egg deposition at Stoney Creek in 1996 was similar to the 1992 level even though deposition at Port Weller over this interval had increased threefold. This difference in egg deposition between natural and man-made reefs has been observed in other Great Lakes (Fitzsimons 1996). Egg deposition at Stoney Creek reef increased more than sevenfold to 851 eggs·m^{-2} in 1999 following seeding the reef with fecal material from YOY lake trout (Seneca Lake strain), and then decreased 30% to 573 eggs·m^{-2} in 2002 following the appearance of round gobies.

Despite increases in egg deposition since the early 1990s, the survival from egg to fry and from fry to age 1 may be too low to produce a self-sustaining lake trout population in Lake Ontario. In addition to predation on eggs and fry by round gobies, alewives are likely suppressing early survival through direct predation on emerging fry and indirectly by a thiamine deficiency and associated negative effects on larval stages. For lake trout, a maternal diet rich in thiaminase-containing alewives results in thiamine-deficient eggs and fry, and early mortality syndrome (EMS) is the term used to describe the erratic behavior (e.g., loss of equilibrium, hyper-excitability, and lethargy) and associated mortality of fry just preceding the swim-up stage (Fitzsimons et al. 1999). Relative to the number of eggs deposited, the number of emergent fry in southwestern Lake Ontario at Port Weller (4, 2, and 2 fry per 100 eggs in 1996, 1997, and 1999, respectively) and at Stoney Creek reef (2 fry per 100 eggs in 1996) has remained low compared to a reef in Lake Superior (9 to 16 fry per 100 eggs) (Peck 1986) where alewives were absent, and a self-sustaining population of lake trout has since been established.

The Lake Ontario fish-community objectives state that the offshore benthic fish community will be composed of self-sustaining native fishes with lake trout as the top predator (Stewart et al. 1999). The indicators of progress towards this objective for lake trout are listed in the unpublished management strategy for Lake Ontario lake trout written in 1997. Only two of the five indicators, numbers of lake trout harvested and sea lamprey wounding levels, fully meet the target values. The remaining three indicators require attention—the abundance of mature fish is decreasing lakewide (it may be below the target value in Canadian waters), the catches of wild age-2 fish continue to remain at a low level, and the target abundance of naturally produced mature fish has not yet been achieved. Major impediments to achievement of a self-sustaining population of lake trout include an adult diet composed primarily of alewife resulting in thiamine deficiency and EMS, alewife and round goby predation on eggs and fry, infestation of spawning reefs with dreissenid mussels, declines in abundance of diporeia (important prey for young lake trout), declines in the survival of stocked yearling lake trout, and declines in the abundance of adults. The short-term outlook for restoring a self-sustaining population of lake trout is poor in light of these impediments.

Burbot

Although uncommon during 1978-1984, burbot abundance increased from 1985 to the mid- to late 1990s then decreased thereafter as evidenced by catches in both gillnet and trawl surveys (Fig. 27). The increase in abundance was probably due to effective sea lamprey control (reducing predation on adult burbot), and a reduction in the number of alewife (reducing predation on pelagic burbot larvae). Lower catches in trawl surveys after 1996 may have been due partially to a conversion from a conventional bottom trawl to one with rollers on the sweep. However, burbot catches in gillnets were also declining by 1998 and the outlook for burbot remains uncertain.

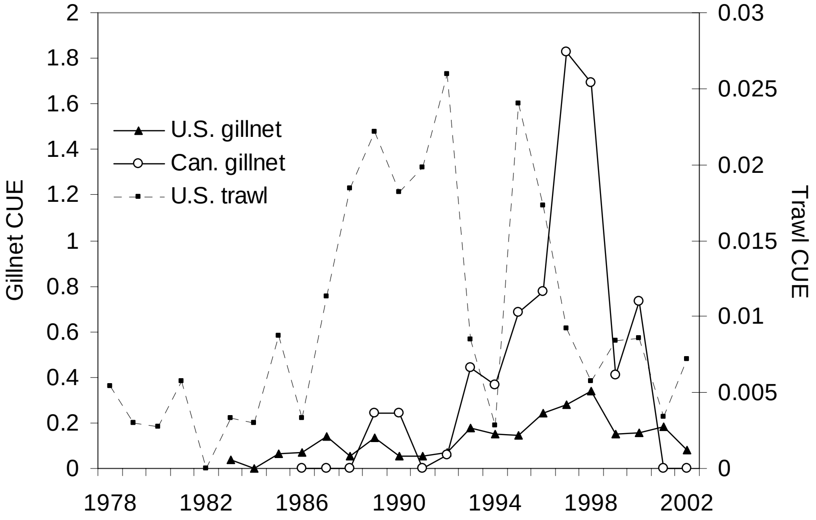

Fig. 27. Burbot abundance in Lake Ontario during 1978-2002 based on catch per unit effort (CUE) in bottom trawls and gillnets (CUE = number per 10-min bottom-trawl tow and number per 136.8 m of graded-mesh gillnet in all U.S. waters, and number per 1000 m of graded-mesh gillnet in eastern Canadian waters).

Sea Lamprey

Sea lamprey control was implemented on Lake Ontario in 1971 when 23 Canadian streams infested with sea lamprey larvae were treated with the lampricide TFM (3-trifluoromethyl-4-nitrophenol). This treatment was followed in 1972 by the treatment of 20 U.S. tributaries (Pearce et al. 1980). Dramatic reductions in sea lamprey abundance such as those observed following the first lampricide treatments in the upper Great Lakes (Smith and Tibbles 1980), were not evident during the first eight years of sea lamprey control on Lake Ontario. Untreated sources in the lake and emigration from Lake Erie (Pearce et al. 1980), where treatments did not commence until 1986, were believed responsible for the lack of initial success. However, recent evidence suggests that the influx from Lake Erie was minimal (Sullivan et al. 2003). Current control practices on Lake Ontario include lampricide applications, operation of barriers, and trapping spawning-phase adults. The abundance of sea lamprey is estimated using a model combining adult mark-recapture data, estimates of larval production potential, and standardized spawning potential in producing streams (Mullett et al. 2003). Estimates from this model show a substantial decline in the spawning-phase sea lamprey population since 1982 (Fig. 28). This decline coincides well with the initial lampricide treatments of the Black River and Fish Creek and construction of sea lamprey barriers on eight Canadian tributaries. By 1999, larval production had ceased in 20 of 57 historically known sea lamprey producing tributaries (Larson et al. 2003). The frequency of A-1 wounds (King and Edsall 1979) on lake trout has been near the target wounding rate of two A-1 wounds per 100 fish (Stewart et al. 1999) since 1985 (Lantry et al. 2003).

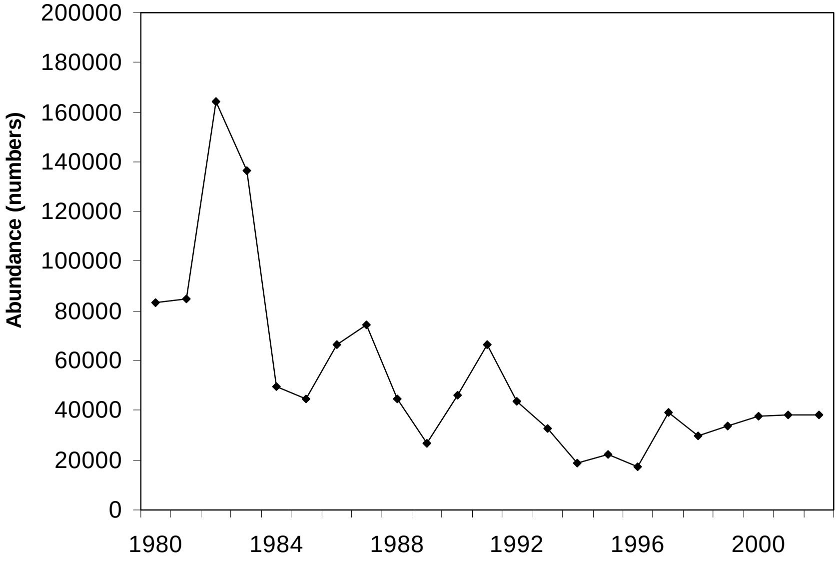

Fig. 28. Abundance of spawning-phase sea lampreys in Lake Ontario, 1980-2002. The estimates were generated by a model described in Mullet et al. (2003).

Current sources of parasitic-phase sea lampreys include those that survive treatment (residual populations) and an untreated population in the Niagara River (Larson et al. 2003). Intermittent production from the Moira and Trent Rivers is low, and treatment of these large rivers would be uneconomical. No reduction in sea lamprey control in Lake Ontario is anticipated in the foreseeable future. Sea lamprey abundance is expected to remain low and stable, and the A-1 wounding rate is expected to remain at around two per 100 lake trout.

THE OFFSHORE PELAGIC FISH COMMUNITY

James N. Bowlby[1], Daniel L. Bishop[2], Marion E. Daniels[3], Thomas H. Eckert[4], James A. Hoyle[1], Brian F. Lantry[5], Robert O'Gorman[5], Randall W. Owens[5], and Ted Schaner[1]

For fish communities, the offshore habitat zone of Lake Ontario is described as waters deeper than 15 m, excluding embayments (Stewart et al. 1999). Offshore pelagic fish are those species that tend to inhabit the midwater thermal zones rather than the bottom of the lake. This relatively simple fish community is composed of two main planktivores, the non-native alewife and the native threespine stickleback, and their predators—a complex of salmonines, including Chinook salmon, coho salmon, rainbow trout, brown trout, and Atlantic salmon. The Atlantic salmon is a native species that has been reintroduced through stocking hatchery-reared fish, but the others are non-native and supported mainly by stocking. About 5 million salmonines have been stocked annually since 1993. Of the four species of ciscoes that historically were the dominant native planktivores in the lake, only the lake herring is still present, but in very low numbers (Christie 1973).

The fish-community objectives for Lake Ontario are: "The offshore pelagic fish community will be characterized by a diversity of salmon and trout; Chinook salmon as the top predator; abundant populations of rainbow trout (steelhead); fishable populations of coho salmon and brown trout; populations of stocked Atlantic salmon at levels consistent with investigating the feasibility of restoring self-sustaining populations; amounts of naturally produced (wild) salmon and trout, especially rainbow trout, that are consistent with fishery and watershed plans; and a diverse prey-fish community with the alewife as an important species" (Stewart et al. 1999).

---

[1] Lake Ontario Management Unit, Ontario Ministry of Natural Resources, Glenora Fisheries Station, 41 Fish Hatchery Lane, R.R. 4, Picton, Ontario, Canada, K0K 2T0

[2] New York Department of Environmental Conservation, 1285 Fisher Avenue, Cortland, New York, U.S.A., 13045

[3] Lake Ontario Management Unit, 300 Water Street, Peterborough, Ontario, Canada, K9J 8M5

[4] New York Department of Environmental Conservation, Cape Vincent Fisheries Research Station, 541 East Broadway, Cape Vincent, New York, U.S.A., 13618

[5] U.S. Geological Survey, Great Lakes Science Center, Lake Ontario Biological Station, 17 Lake Street, Oswego, New York, U.S.A., 13126

The status and trends of the major pelagic fish populations and how they relate to the fish-community objectives are presented below.

Alewife

The major prey of all salmonines in Lake Ontario is the alewife (Brandt 1986b; Lantry 2001). Alewife abundance since 1996 has oscillated within a range lower than in the 1980s and early 1990s (Fig. 29). This pattern of abundance was induced by periods of weak and strong year classes. Year-class strength was related to nearshore water temperatures during spawning and egg incubation, winter weather, lake productivity, and abundance of adult alewives (O'Gorman et al. 2004). Lower average abundance in the late 1990s likely reflected a reduction in the carrying capacity of the lake due to lower primary and secondary productivity (Mills et al. 2003) and dreissenid mussel mediated declines in native benthic macroinvertebrates. A reduction in the numbers of salmon and trout stocked in 1993, intended to reduce predation pressure on prey fish, was rendered less effective by the effects of decreased productivity and dreissenid colonization.

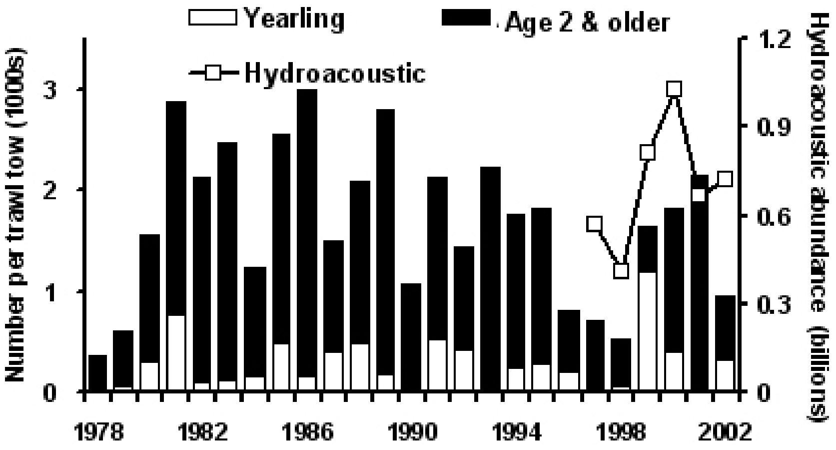

Fig. 29. Relative abundance of alewife during spring 1978-2002 in U.S. waters of Lake Ontario indexed as number per standard trawl tow in New York State Department of Environmental Conservation (NYSDEC) and U.S. Geological Survey (USGS) bottom trawls, and abundance of age-1 and older alewife in Lake Ontario estimated during 1997-2002 by NYSDEC, Ontario Ministry of Natural Resources, and USGS using hydroacoustics and mid-water trawls.

The alewife will likely continue to be the dominant planktivore in the pelagic community, but its abundance and body condition is expected to remain lower than in the 1980s. The pattern of abundance of yearling alewife shows a higher incidence of weak year classes in the early and mid-1990s compared to the 1980s (Fig. 29). Prior to the mid-1990s, the alewife population consisted of a mix of several prominent year classes, but, since then, the population has been dominated by two strong year classes (1995 and 1998). The 2001 year class was moderately strong at age 1, having been favored by unusually mild conditions in its first winter. Water temperatures in spring 2002 and hydroacoustic observations suggest production of a strong 2002 year class, but this cohort experienced a severe first winter (2002-2003) and could be reduced by overwinter mortality before reaching age 1. In the short term, the 2001 and 2002 year classes should enhance or at least maintain the population. Condition of adult alewives improved in the mid-1990s, possibly a density-dependent response to reduced abundance, but condition has since declined (O'Gorman et al. 2003). In 2002, alewife condition declined over the course of the summer, an abnormality seen only once before, in 1983, when the abundance of adults was nearly fivefold higher than in 2002. Because condition is usually a reflection of the food to fish ratio and because alewife numbers were low in 2002, the seasonal decline in condition in 2002 indicates that carrying capacity for alewife is now considerably lower than in the early 1980s.

Lake Herring (Cisco)

Lake herring abundance in Lake Ontario was low at the start of the 20th century, increased during the next three decades, and then declined to remnant population levels by the late 1940s (Christie 1973). The decline in the 1940s coincided with the introduction and proliferation of rainbow smelt. Lake herring abundance has declined further since the 1940s. Since the mid-1990s, few lake herring have been captured in assessment and commercial gear in Lake Ontario (Owens et al. 2003; Ontario Ministry of Natural Resources 2004).

Historically, lake herring was the dominant native prey species providing a thiaminase-free diet item for lake trout and Atlantic salmon. Restoration of lake herring would aid the restoration of self-reproducing populations of lake trout and Atlantic salmon, but significant obstacles must be overcome. Lake herring are planktivorous and likely compete with alewife, rainbow smelt, and threespine stickleback. Alewife and rainbow smelt are also larval-fish predators, potentially preying on lake herring. Also, the amphipod diporeia was a secondary component of the diet of lake herring (Pritchard, 1931), and the decline of lake herring in the 1990s coincided with the collapse of the diporeia population. Because of these obstacles, the lake herring is not expected to recover to its former position as dominant planktivore.

Threespine Stickleback

Catches of threespine sticklebacks in trawls increased after 1993 and currently fluctuate at levels far above those of the early 1990s (Fig. 30; Owens et al. 2003; Ontario Ministry of Natural Resources 2004). The species has also appeared more frequently in the diet of salmonines since the mid-1990s. The true abundance of threespine sticklebacks is difficult to estimate. Because of their small size, they are not effectively caught in standard survey trawls that have mesh sizes designed for capture of adult alewife and rainbow smelt, and, because of their pelagic distribution, they are never fully available to bottom trawls.

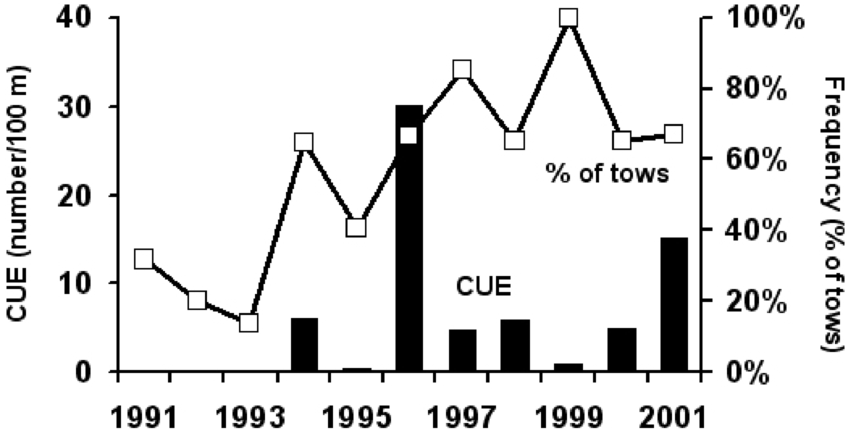

Fig. 30. Abundance of threespine sticklebacks, indexed as number caught per 100-m trawl tow (catch per unit effort, open boxes) and frequency of occurrence (%, solid boxes) in midwater trawls during lakewide summer hydroacoustic surveys of Lake Ontario, 1991-2001.

Chinook Salmon

Harvest rates (number per angler-hour) of salmon and trout in the boat-angler fishery of Lake Ontario indicate that Chinook salmon is the most-abundant salmonine in the pelagic fish community (Bowlby 2003; Eckert 2003b). Its abundance is mainly dependant on stocking, and about 2.2-million fingerlings have been stocked annually in most years from 1997 to 2002 (Fig. 31). Harvest rates in the boat fishery (Fig. 32) indicate the abundance of Chinook salmon increased in the early 1980s as stocking increased. A stocking reduction in 1993 initially produced a corresponding decline in abundance. However, since 1995, abundance has been significantly higher than predicted by stocking levels (Bowlby et al. 1998; Bowlby 2003). Increased natural reproduction and density-dependent survival may be responsible for this higher-than-expected abundance (Bowlby and Daniels 2003).

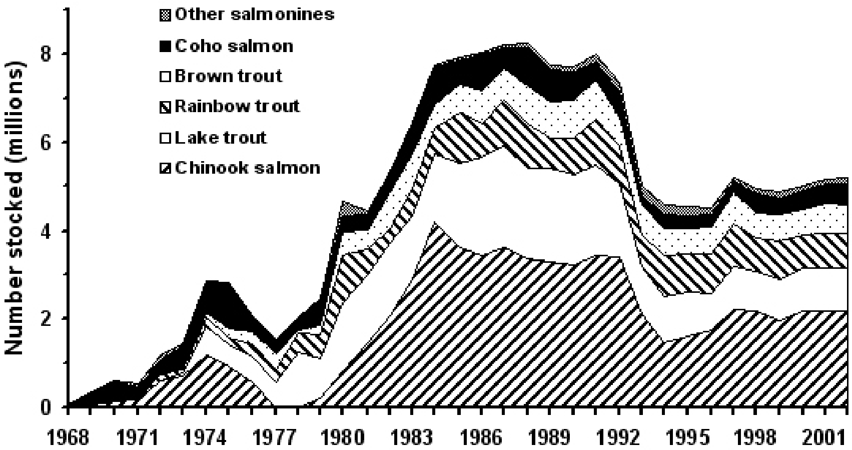

Fig. 31. Number of salmon and trout stocked annually in Lake Ontario, 1968-2002 (includes only fish ≥3 g). Other salmonines are Atlantic salmon, splake, and brook trout.

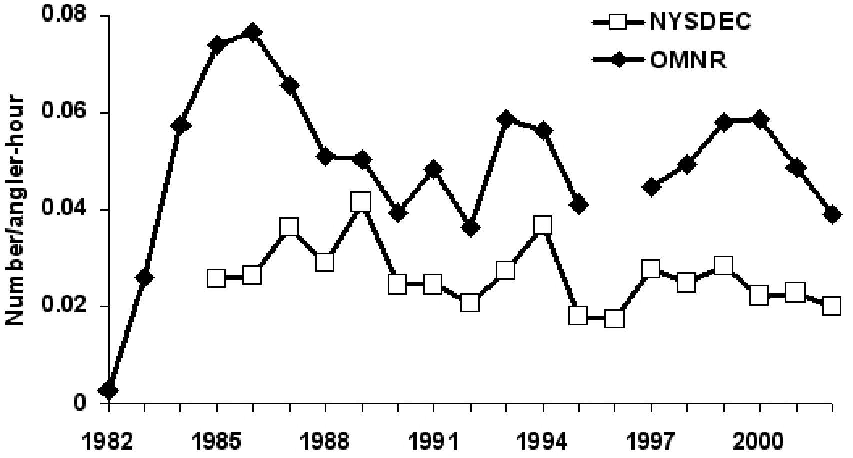

Fig. 32. Harvest rate (number/anger-hour) of Chinook salmon by boat anglers in Lake Ontario during April-September 1982-2002, estimated from surveys by the Ontario Ministry of Natural Resources (Bowlby 2003) and New York State Department of Environmental Conservation (Eckert 2003b).

Natural reproduction of Chinook salmon increased abruptly in 1997 in Ontario tributaries (Fig. 33), including the Salmon River (Bishop and Johnson 2003). Bowlby et al. (1998) estimated about 400,000 pre-smolt fingerlings occupied Ontario tributaries in the spring of 1997. During 1997-2001, natural reproduction has contributed an estimated 473,000 fingerlings annually to Lake Ontario (Bowlby and Daniels 2003).

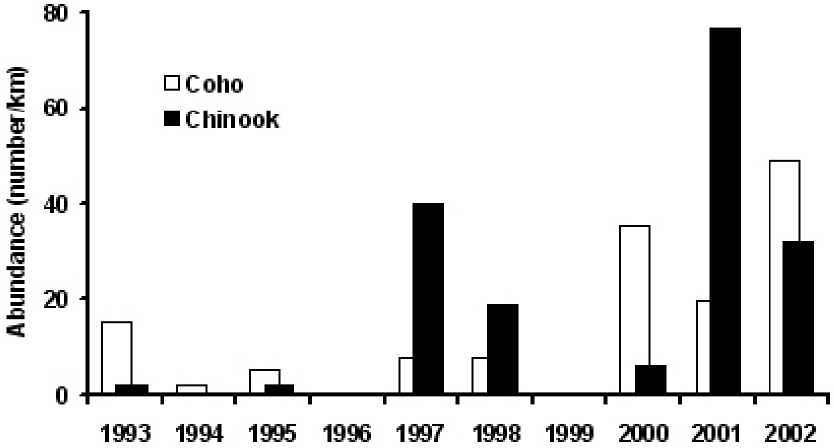

Fig. 33. Estimated relative abundance of wild Chinook and coho salmon juveniles based on summer (24 July-25 September) single-pass electrofishing surveys at 28-31 index sites in Ontario tributaries to Lake Ontario, 1993-2002. No surveys were conducted in 1996 and 1999.

Size-at-age (TL) of mature male and female Chinook salmon returning to the Salmon and Credit Rivers was relatively consistent during 1986-2002 (Fig. 34), but body condition declined (Bowlby and Daniels 2003). Condition of Chinook salmon and alewife are correlated ($r = 0.73$, $p = 0.007$) (Bowlby and Daniels 2003), so the decline in condition of alewife since the mid-1990s is likely responsible for the decline in Chinook salmon body condition.

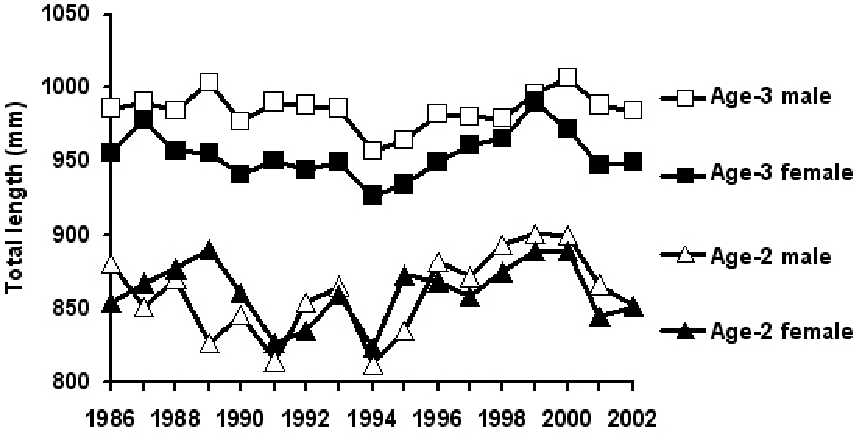

Fig. 34. Size of Chinook salmon in Lake Ontario, 1986-2002, as indexed by total length at age of mature male and females in spawning runs in the Salmon River (New York) and Credit River (Ontario). Data from the two rivers were combined because there was no significant difference (ANOVA) in length at age for either sex between rivers.

Coho Salmon

Most coho salmon in Lake Ontario are stocked (Fig. 31), but the OMNR stopped stocking coho salmon in 1991, and NYSDEC has reduced stocking 59% since 1992. The elimination of stocking in Ontario waters likely contributed to the sharp decline in angler harvest rates during the 1990s (Fig. 35). The OMNR resumed coho salmon stocking in 1997 mainly with a NYSDEC strain, but, for unknown reasons, harvest rates have not increased to expected levels. Natural reproduction of coho salmon in Ontario tributaries has more than quintupled since 1998, and their relative abundance now approximates that of age-1 Chinook (Fig. 33), which also are not common in streams. Therefore, the contribution of wild coho to the population in Lake Ontario is modest at best.

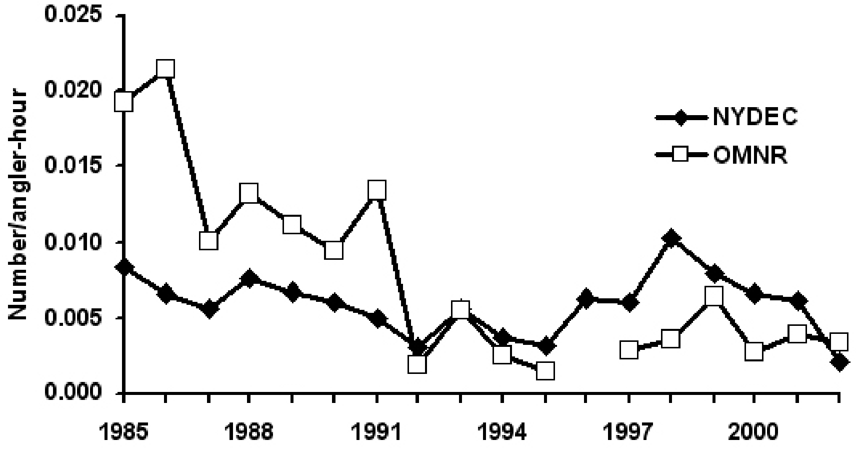

Fig. 35. Harvest rate (number/angler-hour) of coho salmon by boat anglers in Lake Ontario during April-September 1985-2002 estimated from surveys by the Ontario Ministry of Natural Resources (Bowlby 2003) and New York State Department of Environmental Conservation (Eckert 2003b).

Condition of adult (age 2) coho salmon from the spawning run in the Salmon River has been highly variable since 1984 (Fig. 36). Bowlby et al. (1994) noted that condition of coho salmon, rainbow trout, and lake trout declined in the mid-1980s when Chinook salmon population estimates were highest, suggesting a possible interspecific density-dependent response.

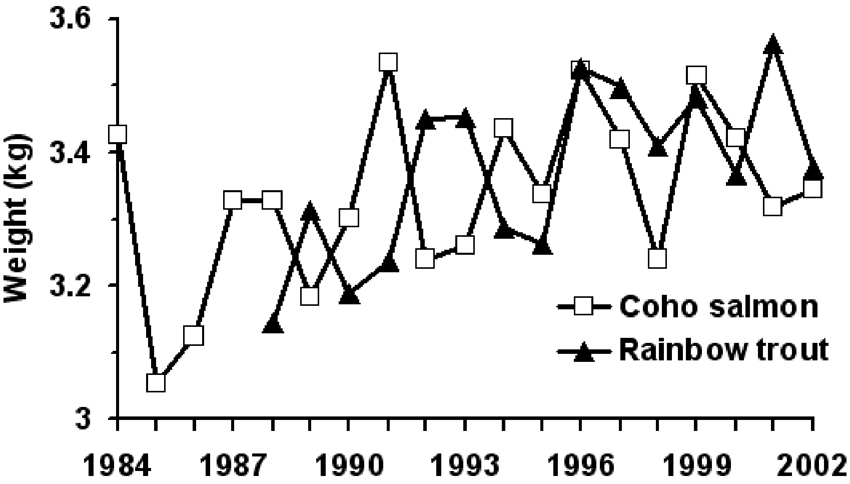

Fig. 36. Body condition (estimated weight of a 700-mm fish from length-weight regressions) of adult coho salmon and rainbow trout in spawning runs in the Salmon River (New York), 1984-2002.

Rainbow Trout

The overall abundance of rainbow trout, based on angler harvest rates, declined during 1985-2001 (Fig 37). In 1992, angler harvest rates increased in New York waters but decreased in Ontario waters (Fig. 37). Stocking of rainbow trout was relatively constant during 1985-2001 averaging close to 900,000 juveniles per year (Fig. 31). Spring air temperature and rainbow trout harvest rates are correlated and may explain year-to-year differences between harvest rates in Ontario and New York waters (Bowlby and Daniels 2003). Spring temperatures affect the timing of spawning and out-migration of rainbow trout from tributaries, and the formation of thermal bars near stream mouths affects rainbow trout migration patterns and distribution in the lake (Haynes et al. 1986). The proportion of wild rainbow trout in the catch averaged 29% during 1989-1995 (Bowlby and Stanfield 2001). The contribution from Ontario versus New York streams is unknown, but most wild rainbow trout are thought to originate from Ontario tributaries because suitable nursery habitat is more abundant in these rivers.

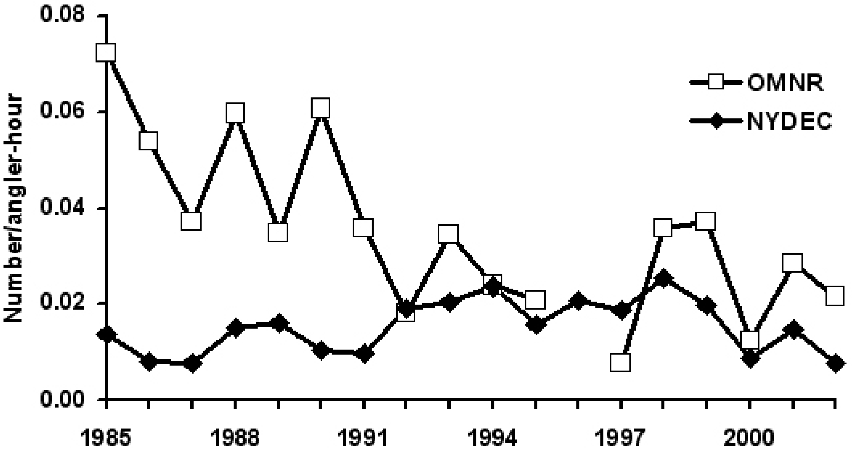

Fig. 37. Harvest rate (number/angler-hour) of rainbow trout by boat anglers in Lake Ontario during April-September 1985-2002 estimated from surveys by the Ontario Ministry of Natural Resources (Bowlby 2003) and the New York State Department of Environmental Conservation (Eckert 2003b).

Unlike the condition of Chinook salmon, condition of adult rainbow trout in spawning runs in New York's Salmon River has increased during 1988-2002 (Fig. 36) even though the diet of both species is mainly alewife (Lantry 2001). The pattern of rainbow trout condition more closely matched that of coho salmon (Fig. 36).

Brown Trout

Most brown trout in Lake Ontario are the result of stocking spring yearlings directly into the lake. The number stocked annually (Fig. 31) has been relatively consistent since 1982. The strains used are also used for inland stocking and have minimal migratory tendencies. Fall spawning runs of fish from the lake are common in New York and Ontario tributaries but are largely confined to lower reaches where spawning and rearing habitat is limited. Brown trout stocked in Ontario and New York tributaries appear to remain in the streams with little tendency to migrate into the lake. There was no trend in angler harvest rates during 1985-2002 (Fig. 38)—rates were much higher in New York waters due to greater numbers stocked.

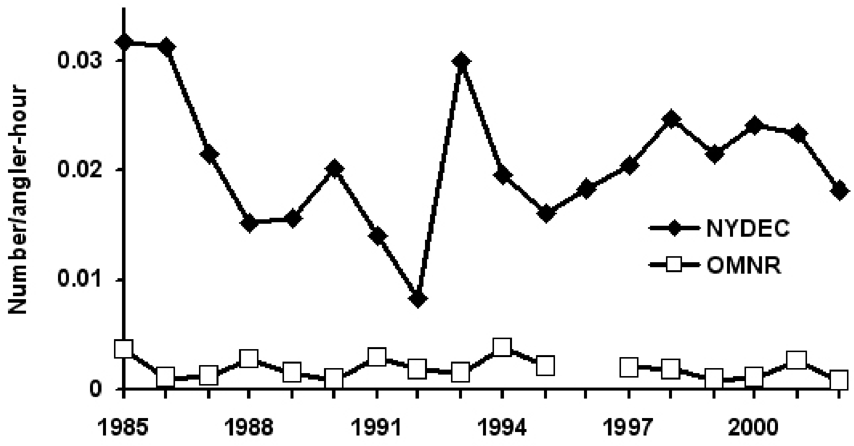

Fig. 38. Harvest rate (number/angler-hour) of brown trout by boat anglers in Lake Ontario during April-September 1985-2002 estimated from surveys by the Ontario Ministry of Natural Resources (Bowlby 2003) and New York State Department of Environmental Conservation (Eckert 2003b).

Atlantic Salmon

The Atlantic salmon, a top predator in Lake Ontario’s pelagic fish community at the time of European colonization, was extirpated prior to 1900 (Christie 1973). Stocking was initiated in 1983 in an attempt to restore naturally reproducing populations (Fig. 39). Until the mid-1990s, most Atlantic salmon were stocked as yearlings—lesser numbers of fall fingerlings were stocked in most years as were a few (<500) adults.

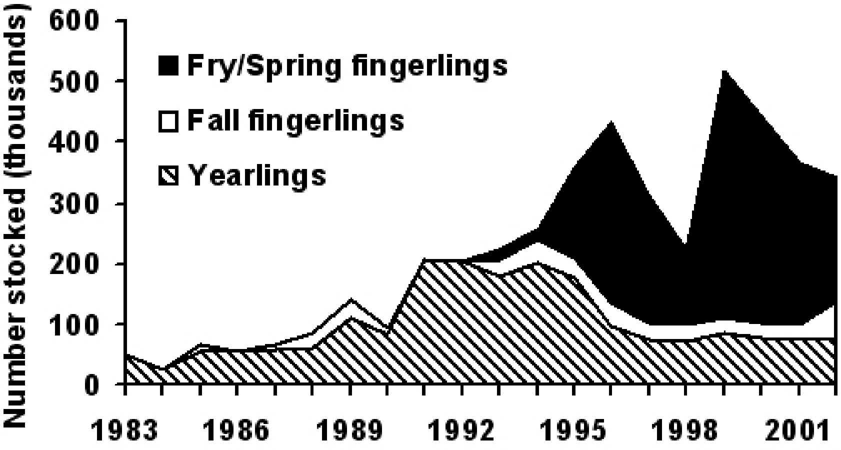

Fig. 39. Numbers of juvenile Atlantic salmon stocked in Lake Ontario by the New York State Department of Environmental Conservation and Ontario Ministry of Natural Resources, 1983-2002.

In 1995, the OMNR discontinued stocking yearlings and increased fry stocking as part of a restoration plan and to study competition for stream habitat between juvenile Atlantic salmon and rainbow trout in 12 historical Atlantic salmon streams (Stanfield and Jones 2003). During 1996-2002, the U.S. Fish and Wildlife Service began similar studies with Atlantic salmon fry and spring fingerlings in 22 New York streams (Johnson and Wedge 1999; Johnson 2004). Although competition with rainbow trout appeared to reduce the survival of juvenile Atlantic salmon, densities of fall fingerlings in Ontario streams exceeded a restoration benchmark of five fish per 100 m^{-2} at over half of the sites surveyed. Thus, these streams were deemed suitable for Atlantic salmon restoration (Stanfield and Jones 2003).

Very few Atlantic salmon are harvested by anglers in Lake Ontario. Annual harvest rates during 1985-2002 were $<0.0004 \cdot \text{angler-hour}^{-1}$ in all years except 1994. Harvest rates underestimate Atlantic salmon abundance, because they are often misidentified and a minimum size limit of 63 cm (25 in) increases their rarity in the harvest.

The future for Atlantic salmon is uncertain as their abundance currently relies on low stocking rates (relative to stocking of other trout and salmon) in a small number of streams. Issues of thiamine deficiency and access to suitable stream habitat remain key challenges to achieving self-sustaining populations.

Future Outlook for Salmonines

Although some Lake Ontario salmonines reproduce successfully, the replacement of native lake herring with exotic alewife in the offshore pelagic food web may pose a significant obstacle to their self-sustainability. Alewife is the main diet item for many of these fishes, but alewife abundance and condition has declined (O'Gorman et al. 2003). In addition, high levels of thiaminase in alewife have caused thiamin deficiency in some Lake Ontario salmonines (Fitzsimons et al. 1999), which has partially impaired natural reproduction. Thiamin deficiency remains one of the greatest concerns for restoration of naturally reproducing Atlantic salmon and lake trout. Restoration of native ciscoes (lake herring, bloater and kiyi) could alleviate the thiamin-deficiency problem, but obstacles to their restoration include competition and predation by alewife and rainbow smelt, predation by salmonines, and declining abundance of diporeia. The dependence of the pelagic salmonine community on stocking will likely continue for the foreseeable future.

LITERATURE CITED

Baillie, J.L. 1947. The double-crested cormorant nesting in Ontario. Can. Field-Nat. **61**: 119-26.

Baldwin, N.S., Saalfeld, R.W., Ross, M.A., and Buettner, H. J. 1979. Commercial fish production in the Great Lakes. Great Lakes Fish. Comm. Tech. Rep. No. 3.

Beeton, A.M., Sellinger, C.E., and Reid, D.E. 1999. An introduction to the Laurentian Great Lakes ecosystem. *In* Great Lakes fisheries policy and management: a binational perspective. *Edited by* W.W. Taylor and C.P. Ferreri. Mich. State U. Press, East Lansing, MI. pp. 3-54.

Benoit, H.P., Johannsson, O.E., Warner, D.M., Sprules, W.G., and Rudstam, L.G. 2002. Assessing the impact of a recent predatory invader: the population dynamics, vertical distribution and potential prey of *Cercopagis pengoi* in Lake Ontario. Limnol. Oceanogr. **47**: 626-635.

Bishop, D.L., and Johnson, J.H. 2003. 2002 Salmon River wild young-of-the-year Chinook salmon seining program. *In* Section 8 of the 2002 annual report of the Lake Ontario and St. Lawrence River Units. Minutes of the 2003 meeting of the Lake Ontario Committee, Great Lakes Fishery Commission. Ann Arbor, MI.

Bowen, K., Millard, E.S., Dermott, R.M., Johannsson, O.E., and Munawar, M. 2003. Lower trophic level comparison of nearshore and offshore habitats in the Bay of Quinte, Lake Ontario. Report #12. Project Quinte annual report 2001. Bay of Quinte remedial action plan. Kingston, ON, Canada.

Bowlby, J.N. 2003. The boat fishery for salmon and trout in western Lake Ontario. *In* Section 8, of the 2001 annual report of the Ontario management Unit, Ontario Ministry of Natural Resources, Picton, ON, Canada. Available from http://www.glfc.org/lakecom/loc/mgmt_unit/index.html.

Bowlby, J.N., and Daniels, M. 2003. Lake Ontario salmon and trout. *In* Section 2 of the 2002 annual report of the Lake Ontario management Unit. Ontario Ministry of Natural Resources, Picton, ON, Canada. Available from http://www. glfc.org/lakecom/loc/mgmt_unit/ index.html.

Bowlby, J.N., and Hoyle, J.A. 2002. Abundance, recruitment and mortality rates of walleye in eastern Lake Ontario. *In* Part III (special studies) of the 2002 annual report of the Lake Ontario Management Unit. Minutes of the Ontario Ministry of Natural Resources, Picton, ON, Canada. Available from http://www.glfc.org/lakecom/loc/mgmt_unit/ index.html.

Bowlby J.N., and Stanfield, L.W. 2001. Exploitation of wild rainbow trout populations in Wilmot Creek and the Ganaraska. *In* Section 10 of the 2000 annual report of the Lake Ontario Management Unit. Ontario Ministry of Natural Resources, Picton, ON, Canada. Available from http://www.glfc.org/lakecom/loc/ mgmt_unit/index.html.

Bowlby, J.N., Mathers, A., Hurley, D.A., and Eckert, T.H. 1991. The resurgence of walleye in Lake Ontario. Great Lakes Fish. Comm. Spec. Pub. 91-1. pp. 169-205.

Bowlby, J.N., Johnston, D., Orsatti, S., Rawson, M., and Schaner, T. 1994. Pelagic piscivores. *In* Section 2 of the 1993 annual report of the Lake Ontario Fisheries Unit. Ontario Ministry of Natural Resources, Picton, ON, Canada.

Bowlby, J.N., Daniels, M., and Schaner, T. 1998. Pelagic piscivores. *In* Section 2 of the 1997 annual report of the Lake Ontario Fisheries Unit. Ontario Ministry of Natural Resources, Picton, ON, Canada.

Brandt, S.B. 1986a. Ontogenetic shifts in habitat, diet, and diel-feeding periodicity of slimy sculpin in Lake Ontario. Trans. Am. Fish. Soc. **115**: 711-715.

Brandt, S.B. 1986b. Food of salmon and trout in Lake Ontario. J. Great Lakes Res. **12**: 200-205.

Brandt, S.B., Mason, D.M., MacNeill, D.B., Coates, T., and Gannon, J.E. 1987. Predation by alewives on larvae of yellow perch in Lake Ontario. Trans. Am. Fish. Soc. **116**: 641-645.

Burnett, J.A.D., Ringler, N.H., Lantry, B.F., and Johnson, J.H. 2002. Double-crested cormorant predation on yellow perch in the eastern basin of Lake Ontario. J. Great Lakes Res. **28**: 202-211.

Carlson, D.M. 1995. Lake sturgeon waters and fisheries in New York State. J. Great Lakes Res. **21**: 35-41.

Casselman, J.M. 2001. Dynamics of American Eel, *Anguilla rostrata*, Resources: declining abundance in the 1990s. Extended abstract of a paper presented to Advances in Eel Biology, Tokyo, Japan, September 28-30, 2001.

Casselman, J.M. 2002. Effects of temperature, global extremes, and climate change on year-class production of warmwater, coolwater, and coldwater fishes in the Great Lakes Basin. *In* Fisheries in a changing climate. *Edited by* N.A. McGinn. Proc. Am. Fish. Soc. Symp. 32. pp. 39-59.

Casselman, J.M., and Lewis, C.A. 1996. Habitat requirements of northern pike *Esox lucius*. Can. J. Fish. Aquat. Sci. **53**(Suppl. 1): 161-174.

Casselman, J.M., Brown, D.M., Hoyle, J.A., and Eckert, T.H. 2002. Effects of climate and global warming on year-class strength and relative abundance of smallmouth bass in eastern Lake Ontario. Am. Fish. Soc. Symp. **31**: 73-90.

Castonguay, M., Hodson, P.V., Couillard, C.M., Eckersley, M.J., Dutil, J-D., and Verreault, G. 1994. Why is recruitment of the American eel, *Anguilla rostrata*, declining in the St. Lawrence River and Gulf? Can. J. Fish. Aquat. Sci. **51**: 479-488.

Charlebois, P.M., Corkum, L.D., Jude, D.J., and Knight, C. 2001. The round goby (*Neogobius melanostomus*) invasion: current research and future needs. J. Great Lakes Res. **27**: 263-266.

Charlton, M.N., Le Sage, R., and Milne, J.E. 1999. Lake Erie in transition: the 1990s. *In* State of Lake Erie (SOLE): past, present and future. Edited by M. Munawar, T. Edsall, and I.F. Munawar. Backhuys Publishers, Leiden, The Netherlands.

Chotkowski, M.A., and Marsden, J.E.. 1999. Round goby and mottled sculpin predation on lake trout eggs and fry: field predictions from laboratory experiments. J. Great Lakes Res. **25**: 26-35.

Christie, W.J. 1963. Effects of artificial propagation and the weather on recruitment in the Lake Ontario whitefish fishery. J. Fish. Res. Board Can. **20**: 597-646.

Christie, W.J. 1972. Lake Ontario: Effects of exploitation, introductions, and euthrophication on the salmonid community. J. Fish. Res. Board Can. **29**: 913-929.

Christie, W. J. 1973. A review of the changes in the fish composition of Lake Ontario. Great Lakes Fish. Comm., Tech. Rep. 23.

Christie, W.J., Scott, K.A., Sly, P.G., and Strus, R.H. 1987. Recent changes in the aquatic food web of eastern Lake Ontario. Can. J. Fish. Aquat. Sci. **44**(Suppl. 2): 37-52.

Cook, P.M., Zabel, E.W., and Peterson, R.E. 1997. The TCDD toxicity equivalence approach for characterizing risks for early life-stage mortality in trout. *In* Chemically induced alterations in functional development and reproduction of fishes. *Edited by* R.M. Rolland, M. Gilbertson, and R.E. Peterson. Proceedings from a session at the Wingspread Conference Center, 21-23 July 1995, Racine, WI. Society of Environmental Toxicology and Chemistry (SETAC), Pensacola, FL, pp. 9-27.

Dermott, R. 2001. Sudden disappearance of the amphipod *Diporeia* from eastern Lake Ontario, 1993-1995. J. Great Lakes Res. **27**: 423-433.

Dermott, R. 2003. Diporeia in Lake Ontario. *In* State of Lake Ontario (SOLO): past, present and future. *Edited by* M. Munawar. Backhuys Publishers, Leiden, The Netherlands.

Dermott, R.M., and Legner, M. 2002. Dense mat-forming bacterium Thioploca ingrica (Beggiatoaceae) in eastern Lake Ontario: implications to the benthic food web. J. Great Lakes Res. **28**: 688-697.

Eckert, T.H. 2002a. Summary of 1976-2001 warm water assessment. *In* Section 4 *of the* 2001 annual report of the Lake Ontario and St. Lawrence River Units. Minutes of the 2002 meeting of the Lake Ontario Committee, Great Lakes Fishery Commission. Ann Arbor, MI.

Eckert, T.H. 2002b. Lake Ontario fishing boat census 2001. *In* Section 2 of the 2001 annual report of the Lake Ontario and St. Lawrence River Units. Minutes of the 2002 meeting of the Lake Ontario Committee, Great Lakes Fishery Commission. Ann Arbor, MI.

Eckert, T.H. 2003a. Summary of 1976-2002 warm water assessment. *In* Section 4 of the 2002 annual report of the Lake Ontario and St. Lawrence River Units. Minutes of the 2003 meeting of the Lake Ontario Committee, Great Lakes Fishery Commission. Ann Arbor, MI.

Eckert, T. H. 2003b. Lake Ontario fishing boat census 2002. *In* Section 2. of the 2002 annual report of the Lake Ontario and St. Lawrence River Units. Minutes of the 2003 meeting of the Lake Ontario Committee, Great Lakes Fishery Commission. Ann Arbor, MI.

Eckert, T.H. and Pearsall, W. 2002. Comparison of warm water gillnetting: Pultneyville and eastern basin. *In* Section 22 of the 2001 annual report of the Lake Ontario and St. Lawrence River Units. Minutes of the Lake Ontario Committee, Great Lakes Fishery Commission. Ann Arbor, MI.

EIFAC/ICES Working Group on Eels. 2001. Report of the EIFAC/ICES Working Group on Eels, St. Andrew, NB., 28 August-1 September 2000. Advisory Committee on Fisheries Management, International Council for the Exploration of the Sea. ICES CM 2001/ACFM: 03, Copenhagen.

Elrod, J.H., and O'Gorman, R. 1991. Diet of juvenile lake trout in southern Lake Ontario in relation to abundance and size of prey fishes, 1979-1987. Trans. Am. Fish. Soc. **120**: 290-302.

Elrod, J.H., Schneider, C.P., and Ostergaard, D.E. 1993. Survival of lake trout stocked in U.S. waters of Lake Ontario. N. Am. J. Fish. Manage. **13**: 775-781.

Elrod, J.H., O'Gorman, R., Schneider, C.P., Eckert, T.H., Schaner, T., Bowlby, J.N., and Schleen, L. 1995. Lake trout rehabilitation in Lake Ontario. J. Great Lakes Res. **21**(Suppl. 1): 83-107.

Farquhar, J.F., McCullough, R.D., and Mazzocchi, I.M. 2002. Cormorant management activities in Lake Ontario's eastern basin. NY State Dept. Env. Cons. Spec. Rep., Albany, NY.

Fitzsimons, J.D. 1995. Assessment of lake trout spawning habitat, egg deposition and survival in Lake Ontario. J. Great Lakes Res. **21**(Suppl. 1): 337-347.

Fitzsimons, J.D. 1996. The significance of man-made structures for lake trout spawning in the Great Lakes: are they a viable alternative to natural reefs? Can. J. Fish. Aquat. Sci. **53**:142-15.

Fitzsimons, J.D., Brown, S.B., Honeyfield, D.C., and Hnath, J. 1999. A review of early mortality syndrome (EMS) in Great Lakes salmonids: relationship with thiamine deficiency. Ambio. **28**: 9-15.

Fitzsimons J.D., Lantry, B.F., and O'Gorman, R. 2003. A review of lake trout (Salvelinus namaycush) restoration in Lake Ontario from an early life history perspective. *In* State of Lake Ontario (SOLO): past, present, and future. *Edited by* M. Munawar. Ecovision World Monograph Series. Backhuys Publishers, Leiden, The Netherlands.

French, J.R.P., III, and Jude, D.J. 2001. Diets and diet overlap of nonindigenous gobies and small benthic native fishes co-inhabiting the St. Clair River, Michigan. J. Great Lakes Res. **27**(3): 300-311.

Great Lakes National Program Office. 2003. USEPA Great Lakes National Program's fish indicator [online]. Available from http://www.epa.gov/glnpo/glindicators/fish.html.

Great Lakes Fishery Commission. 1997. A joint strategic plan for management of Great Lakes fisheries. Great Lakes Fish. Comm., Ann Arbor, MI. http://www.glfc.org/pubs_out/docs.htm

Hall, R.H., Pauliukonis, N.K., Mills, E.L., Rudstam, L.G., Schneider, C.P., Lary, S.J., and Arrhenius, F. 2003. A comparison of total phosphorus, chlorophyll *a*, and zooplankton in embayment, nearshore, and offshore habitats of Lake Ontario. J. Great Lakes Res. **29**: 54-69.

Hart, J.L. 1931. The food of the whitefish *Coregonus clupeaformis* (Mitchill) in Ontario waters, with a note of the parasites. Contr. Can. Biol. Fish. No. 20, **6**: 447-454.

Hatch, J.J. 1995. Changing populations of double-crested cormorants. Colon. Waterbirds **18** (Spec. Pub. 1): 8-24.

Hatch, J.J., and Weseloh, D.V. 1999. Double-crested cormorant. *In* The birds of North America, No. 441. *Edited by* A. Poole and F. Gill. Cornell Laboratory of Ornithology, Ithaca, NY and the Academy of Natural Sciences, Philadelphia, PA.

Haynes, J.M., Nettles, D.C., Parnell, K.M., Voiland, M.P., Olson, R.A., and Winter, J.D. 1986. Movements of rainbow steelhead trout (*Salmo gairdneri*) in Lake Ontario and a hypothesis for the influence of spring thermal structure. J. Great Lakes Res. **12**: 304-313.

Hayes, J.M., Stewart, T.W., and Cook, G.E. 1999. Benthic macroinvertebrate communities in southwestern Lake Ontario following invasion of Dreissena: continuing change. J. Great Lakes Res. **25**: 828-838.

Hoyle, J.A. 2005. Status of lake whitefish (*Coregonus clupeaformis*) in Lake Ontario and the response to the disappearance of *Diporeia* spp. *In* Proceedings of a workshop on the dynamics of lake whitefish (*Coregonus clupeaformis*) and the amphipod *Diporeia* spp. in the Great Lakes. *Edited by* L.C. Mohr and T.F. Nalepa. Great Lakes Fish. Comm. Tech. Rep. 66. pp. 47-66.

Hoyle, J.A., and Schaner, T. 2002. Lake Ontario nearshore fish community. *In* Part I (fish communities) of the 2001 annual report of the Lake Ontario Management Unit. Ontario Ministry of Natural Resources, Picton, ON, Canada. Available from http://www.glfc.org/lakecom/loc/mgmt_unit/index.html.

Hoyle, J.A., Casselman, J.M., Dermott, R., and Schaner, T. 2003. Resurgence and decline of lake whitefish (*Coregonus clupeaformis*) stocks in eastern Lake Ontario following *Dreissena* mussel invasion. In State of Lake Ontario (SOLO): past, present, and future. Edited by M. Munawar. Ecovision World Monograph Series. Backhuys Publishers, Leiden, The Netherlands.

Hurley, D.A. 1986. Fish populations of the Bay of Quinte, Lake Ontario, before and after phosphorus control. *In* Project Quinte: point-source phosphorus control and ecosystem response in the Bay of Quinte, Lake Ontario. *Edited by* C.K. Minns, D.A. Hurley, and K.H. Nichols. Can. Spec. Publ. Fish. Aquat. Sci. **86**: 201-214.

Hurley, D.A., and W.J. Christie. 1977. Depreciation of the warm-water fish community in the Bay of Quinte, Lake Ontario. J. Fish. Res. Bd. Canada. **34**:1849-1860.

Ihssen, P.E., Evans, D.O., Christie, W.J., Reckahn, J.A. and DesJardine, R.L. 1981. Life history, morphology, and electrophoretic characteristics of five allopatric stocks of lake whitefish (*Coregonus clupeaformis*) in the Great Lakes region. Can. J. Fish. Aquat. Sci. **38**: 1790-1807.

Johannsson, O.E. 1995. The response of *Mysis relicta* population dynamics and productivity to spatial and temporal gradients in Lake Ontario. Can. J. Fish. Aquat. Sci. **52**: 1509-1522.

Johannsson, O.E. 2003. Lake Ontario zooplankton: role in energy transfer and response to management actions. *In* State of Lake Ontario (SOLO): past. present, and future. *Edited by* M. Munawar. Backhuys Publishers, Leiden, The Netherlands.

Johannsson, O.E., Millard, E.S., Ralph, K.M., Myles, D.D., Graham, D.M., Taylor, W.D., Giles, B.G., and Allen, R.E. 1998. The changing pelagia of Lake Ontario (1981 to 1995): a report of the Department of Fisheries and Oceans long-term biomonitoring (bioindex) program. Can. Tech. Rep. Fish. Aquat. Sci. No. 2243.

Johannsson, O. E., Mills, E.L., and O'Gorman, R. 1991. Changes in the nearshore and offshore zooplankton communities in Lake Ontario. Can. J. Fish. Aquat. Sci. **48**: 1546-1557.

Johannsson, O.E., Bowen, K.L., and Gerlofsma, J. 2003. Zooplankton. *In* Report #12. Project Quinte annual report 1999-2000. Bay of Quinte remedial action plan, Kingston, ON, Canada.

Johnson, J.H. 2004. Comparative survival and growth of Atlantic salmon from egg stocking and fry releases. North Am. J. Fish. Manage. **24**: 1409-1412.

Johnson, J.H., and Wedge, L.R. 1999. Interspecific competition in tributaries: prospectus for restoring Atlantic salmon in Lake Ontario. Great Lakes Res. Rev. **4**: 11-17.

Johnson, J.H., LaPan, S.R., and Bendig A. 2000a. Diet composition and fish consumption of double-crested cormorants from three St. Lawrence River colonies in 1999. *In* Section 17 of the 1999 annual report of the Lake Ontario and St. Lawrence River Units. Minutes of the 2000 meeting of the Lake Ontario Committee. Great Lakes Fishery Commission. Ann Arbor, MI.

Johnson, J.H., Ross, R.M., and Farquhar, J.F. 2000b. The effects of egg oiling on fish consumption by double-crested cormorants on Little Galloo Island, Lake Ontario. U.S. Geological Survey. NY State Dept. Env. Cons. Spec. Rep., Albany, NY.

Johnson, J.H., LaPan, S.R., Edmonds, B., and Bendig, A. 2001a. Diet composition and fish consumption of double-crested cormorants from three St. Lawrence River colonies in 2000. *In* Section 17 of the 2000 annual report of the Lake Ontario and St. Lawrence River Units. Minutes of the 2001 meeting of the Great Lakes Fishery Commission. Ann Arbor, MI.

Johnson, J.H., Ross, R.M., and Farquhar, J.F. 2001b. The effects of egg oiling on fish consumption by double-crested cormorants on Little Galloo Island, Lake Ontario in 2001. NY State Dept. Env. Cons. Spec. Rep., Albany, NY.

Johnson, J.H., Klindt, R.M., Edmonds, B., and Bendig, A. 2002a. Diet composition and fish consumption of double-crested cormorants from three St. Lawrence River colonies in 2001. *In* Section 17 of the 2001 annual report of Lake Ontario and St. Lawrence River Units. Minutes of the 2002 meeting of the Great Lakes Fishery Commission. Ann Arbor, MI.

Johnson, J.H., Ross, R.M., McCullough, R.D., and Edmonds, B. 2002b. Diet composition and fish consumption of double-crested cormorants from the Little Galloo Island colony of eastern Lake Ontario in 2001. *In* Section 14 of the Lake Ontario and St. Lawrence River Units. Minutes of the 2002 meeting of the Great Lakes Fishery Commission. Ann Arbor, MI.

Johnston, T.A. 1997. Within-population variability in egg characteristics of walleye (*Stizostedion vitreum*) and white sucker (*Catostomus commersoni*). Can. J. Fish. Aquat. Sci. **54**: 1006-1014.

Jude, D.J., Reider, R.H., and Smith, G.R. 1992. Establishment of Gobiidae in the Great Lakes basin. Can. J. Fish. Aquat. Sci. **49**: 416-421.

Ketola, H.G., Bowser, P.R., Wooster, G.A., Wedge, L.R., and Hurst, S.S. 2000. Effects of thiamine on reproduction of Atlantic salmon and a new hypothesis for their extirpation in Lake Ontario. Trans. Am. Fish. Soc. **129**: 607-612.

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

Klumb, R.A., Rudstam, L.G., Mills, E.L., Schneider, C.P., and Sawyko, P.M. 2003. Importance of Lake Ontario embayments and nearshore habitats as nurseries for larval fishes with emphasis on alewife (*Alosa pseudoharengus*). J. Great Lakes Res. **29**: 181-198.

Kuns, M.M., and Sprules, W.G. 2000. Zooplankton production in Lake Ontario: a multistrata approach. Can. J. Fish. Aquat. Sci. **57**: 2240-2247.

Lakewide Management Plan. 1998. Lake Ontario lakewide management plan biennial report. Stage 1: problem definition. May 1998 [online]. Available from http://www.epa.gov/glnpo/lakeont/index.html.

Lakewide Management Plan. 2002a. Lake Ontario lakewide management plan biennial report. April 2002 [online]. Available from http://www.epa.gov/glnpo/lakeont/2002highlight/ontLamp2002web.pdf.

Lakewide Management Plan. 2002b. Implementing indicators 2003: a technical report [online]. Available from http://www.epa.gov/glnpo/solec/solec_2002/Implementing _Indicators.pdf.

Lantry, B.F., and Stewart, D.J. 2000. Population dynamics and ecological efficiency of rainbow smelt (*Osmerus mordax*) in Lakes Ontario and Erie: a modeling analysis of cannibalism effects. Can. J. Fish. Aquat. Sci. **57**: 1594-1606.

Lantry, B.F., Eckert, T.H., Schneider, C.P., and Chrisman, J.R. 2002. The relationship between the abundance of smallmouth bass and double-crested cormorants in the eastern basin of Lake Ontario. J. Great Lakes Res. **28**(2): 193-201.

Lantry, B.F., Eckert, T.H., O'Gorman, R., and Owens, R.W. 2003. Lake trout rehabilitation in Lake Ontario, 2002. *In* Section 5 of the 2002 annual report of the Lake Ontario and St. Lawrence River Units. Minutes of the 2003 Lake Ontario Committee, Great Lakes Fishery Commission. Ann Arbor, MI.

Lantry, J.R. 2001. Spatial and temporal dynamics of predation by Lake Ontario trout and salmon. M.Sc. Thesis. State University of New York College of Environmental Science and Forestry, Syracuse, NY.

Larson, G.L., Koonce, J.F., Christie, G.C., Johnson, D.A., Mullett, K.M., and Sullivan, W.P., 2003. The history and optimization of sea lamprey control in Lake Ontario. J. Great Lakes Res. **29**(Suppl. 1): 637-654.

Laxson, C.L., McPhedran, K.N., Makarewicz, J.C., Telesh, I.V., and MacIsaac, H.J. 2003. Effects of the non-indigenous cladoceran *Cercopagis pengoi* on the lower food web of Lake Ontario. Freshwater Biol. **48**: 2094-2106.

Levels Reference Study Board. 1993. Levels reference study: Great Lakes-St. Lawrence River basin. Inter. Joint Comm., March 1993.

Limno-Tech, Inc. 2004. LOTOX2 model documentation: in support of development of load reduction strategies and a TMDL for PCBs in Lake Ontario. Technical report, New England Interstate Water Pollution Control Commission, Lowell, MA; EPA-Region 2, New York, NY.

Litten, S., and Donlon, J. 1998. Enhanced toxics sampling for trace organic chemicals in Lake Ontario. NY State Dept. Env. Cons. Division of Water, Spec. Rep., Albany, NY. 31 p.

Lozano, S.J., Scharold, J.V., and Nalepa, T.F. 2001. Recent declines in benthic macroinvertebrate densities in Lake Ontario. Can. J. Fish. Aquat. Sci. **58**: 518-529.

Luross, J.M., Alaee, A., Sergeant, D.B., Cannon, C.M., Whittle, D.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.

Mazzocchi, I.M. 2002. Double-crested cormorant and VHF telemetry on Lake Ontario, 2001. *In* Section 21 of the 2001 annual report of the Lake Ontario and St. Lawrence River Units. Minutes of the 2002 meeting of the Lake Ontario Committee of the Great Lakes Fishery Commission. Ann Arbor, MI.

McAllister, D.E. 1961. The origin and status of the deepwater sculpin, *Myoxocephalus thompsonii*, a nearctic glacial relict. Bull. Nat. Mus. Can. Contrib. Zool. **172**: 44-65.

McCullough, R.D., and Einhouse, D.W. 1999. Lake Ontario-eastern basin creel survey, 1998, section 4. *In* Final report: to assess the impact of double-crested cormorant predation on smallmouth bass and other fishes in the eastern basin of Lake Ontario. Cooperative report, New York State Department of Environmental Conservation and USGS Biological Resources Division. NY State Dept. Env. Cons. Spec. Rep., Albany, NY., February 1, 1999.

Millard, E.S., and Burley, M. 2003. Photosynthesis, Chlorophyll *a* and light attenuation. Report #12. Project Quinte annual report, 2001. Bay of Quinte remedial action plan. Kingston, ON, Canada.

Millard, E.S., Myles, D.D., Johannsson, O.E., and Ralph, K.M. 1996. Phytoplankton photosynthesis at two index stations in Lake Ontario 1987-92: assessment of the long-term response to phosphorus control. Can. J. Fish. Aquat. Sci. **53**: 1092-1111.

Millard, E.S., Johannsson, O.E., Neilson, M.A., and El-Shaarawi, A.H. 2003. Long-term, seasonal and spatial trends in nutrients, chlorophyll *a* and light attenuation in Lake Ontario. *In* The state of Lake Ontario ecosystem (SOLO)‒past, present and future. *Edited by* M. Munawar. Backhuys Publishers, Leiden, The Netherlands. pp. 97-132.

Miller, R.L. 1998. The double crested cormorant in New York. NY State Dept. Env. Cons. Spec. Rep., Albany, NY.

Mills, E.L., Casselman, J.M., Dermott, R., Fitzsimons, J.D., Gal, G., Hoyle, J.A., Johannsson, O.E., Lantry, B.F., Makarewicz, J.C., Millard, E.S., Munawar, M., Munawar, I.F., O'Gorman, R., Owens, R.W., Rudstam, L.G., Schaner, T., and Stewart, T.J. 2003. Lake Ontario: food web dynamics in a changing ecosystem (1970-2000). Can. J. Fish. Aquat. Sci. **60**: 471-490.

Minns, C.K., and Hurley, D.A. 1986. Population dynamics and production of white perch, *Morone americana*, in the Bay of Quinte, Lake Ontario. *In* Project Quinte: point-source phosphorus control and ecosystem response in the Bay of Quinte, Lake Ontario. *Edited by* C.K. Minns, D.A. Hurley, and K.H. Nichols. Can. Spec. Publ. Fish. Aquat. Sci. **86**: 215-223.

Mullett, K.M., Heinrich, J.W., Adams, J.V., Young, R.J., Henson, M.P., McDonald, R.B., and Fodale, M.F. 2003. Estimating lake-wide abundance of spawning-phase sea lampreys (*Petromyzon marinus*) in the Great Lakes: extrapolating from sampled streams using regression models. J. Great Lakes Res. **29**(Suppl. 1): 240-252.

O'Gorman, R., and Burnett, J.A.D. 2001. Fish community dynamics in northeastern Lake Ontario with emphasis on the growth and reproductive success of yellow perch (*Perca flavescens*) and white perch (*Morone americana*), 1978 to 1997. J. Great Lakes Res. **27**:367-383.

O'Gorman, R., and Stewart, T.J. 1999. Ascent, dominance, and decline of the alewife in the Great Lakes: food web interactions and management strategies. *In* Great Lakes policy and management: a binational perspective. *Edited by* W.W. Taylor and P. Ferreri. Michigan State U. Press, East Lansing, MI. pp. 489-514.

O'Gorman, R., Elrod, J.H., and Schneider, C.P. 1998. Reproductive potential and fecundity of lake trout strains in southern and eastern Lake Ontario, 1977-94. J. Great Lakes Res. **24**: 131-144.

O'Gorman, R., Elrod, J.H., Owens, R.W., Schneider, C.P., Eckert, T. H., and Lantry, B. F. 2000. Shifts in the depth distributions of alewives, rainbow smelt, and age-2 lake trout in southern Lake Ontario following establishment of dreissenids. Trans. Am. Fish. Soc. **129**: 1096-1106.

O'Gorman, R., Owens, R.W., Lantry, B.F., and Eckert, T.H. 2003. Status of major prey fish stocks in the U.S. waters of Lake Ontario, 2002. *In* Section 12 of the 2002 annual report of the Lake Ontario and St. Lawrence River Units. Minutes of the 2003 meeting of the Lake Ontario Committee, Great Lakes Fishery Commission. Ann Arbor, MI.

O'Gorman, R., Lantry, B.F., and Schneider, C.P. 2004. Effect of stock size, climate, predation, and trophic status on recruitment of alewives in Lake Ontario, 1978-2000. Trans. Am. Fish. Soc. **133**: 855-867.

Ontario Ministry of Natural Resources. 2004. Lake Ontario fish communities and fisheries. *In* 2003 annual report of the Lake Ontario Management Unit. Ontario Ministry of Natural Resources, Picton, Ontario [online]. Available from http://www.glfc.org/lakecom/loc/mgmt_unit/index.html.

Owens, R.W., and Bergstedt, R.A. 1994. Response of slimy sculpins to predation by juvenile lake trout in southern Lake Ontario. Trans. Am. Fish. Soc. **123**: 28-36.

Owens, R.W., and Dittman, D.E. 2003. Shifts in the diets of slimy sculpin (*Cottus cognatus*) and lake whitefish (*Coregonus clupeaformis*) in Lake Ontario following the collapse of the burrowing amphipod, *Diporeia*. Aquat. Ecosys. Health Manage. **6**: 311-322.

Owens, R.W., and Weber, P.G. 1995. Predation on *Mysis relicta* by slimy sculpins (*Cottus cognatus*) in southern Lake Ontario. J. Great Lakes Res. **21**: 275-283.

Owens, R.W., O'Gorman R., Eckert, T.H., and Lantry, B.F. 2003. The offshore fish community in southern Lake Ontario, 1972-1998. *In* The state of Lake Ontario: past, present, and future. *Edited by* M. Munawar. Ecovision World Monograph Series, Aquatic Ecosystem Health and Management Society, Burlington, ON, Canada. pp. 407-442.

Owens, R.W., O'Gorman, R., Eckert, T.H., Lantry, B.F., and Dittman, D.E. 2005. *In* Proceedings of a workshop on the dynamics of lake whitefish (*Coregonus clupeaformis*) and the amphipod *Diporeia* spp. in the Great Lakes. *Edited by* L.C. Mohr and T.F. Nalepa. Great Lakes Fish. Comm. Tech. Rep. 66. pp. 141-155.

Payne, N. 1963. The life history of the yellow walleye, (*Stizostedion vitreum vitreum*) (Mitchill), in the Bay of Quinte. M.Sc. Thesis, University of Toronto, ON, Canada. 40 p.

Pearce, W.A., Braem, R.A., Dustin, S.M., and Tibbles, J.J. 1980. Sea lamprey (*Petromyzon marinus*) in the lower Great Lakes. Can. J. Fish. Aquat. Sci. **37**: 1802-1810.

Peck, J.W. 1986. Dynamics of reproduction by hatchery lake trout on a man-made spawning reef. J. Great Lakes Res. **12**:293-303.

Price, I.M., and Weseloh, D.V. 1986. Increased numbers and productivity of double-crested cormorants, *Phalacrocorax auritu*s, on Lake Ontario. Canadian Field-Nat. **100**: 474-482.

Pritchard, A.L. 1931. Taxonomic and life history studies of the ciscoes of Lake Ontario. Univ. Toronto Stud., Biol. Ser. **41**: 1-78.

Ridgeway, M.S., Hurley, D.A., and Scott, K.A. 1990. Effects of winter temperature and predation on the abundance of alewife (*Alosa pseudoharengus*) in the Bay of Quinte, Lake Ontario. J. Great Lakes Res. **16**:11-20.

Ross, R.M., and Johnson, J.H. 1995. Seasonal and annual changes in the diet of double-crested cormorants: implications for Lake Ontario's fishery. Great Lakes Res. Rev. **2**: 1-9.

Ross, R.M, Johnson, J.H., McCullough, R.D., and Edmonds, B. 2003. Diet composition and fish consumption of double-crested cormorants from the Pigeon and Snake Island colonies of eastern Lake Ontario in 2002. *In* Section 16 of the 2002 annual report of the Lake Ontario and St. Lawrence River units. Minutes of the 2003 meeting of the Lake Ontario Committee, Great Lakes Fishery Commission. Ann Arbor, MI.

Schaner, T., Bowlby, J.N., Hoyle, J.A., and Stewart, T.J. 2002. Model simulation of walleye in eastern Lake Ontario. Part III. Special studies. *In* Lake Ontario fish communities and fisheries: 2001 annual report of the Lake Ontario Management Unit. Ontario Ministry of Natural Resources, Picton, ON, Canada. Available from http://www.glfc.org/lakecom/loc/mgmt_unit/index.html.

Schaner, T., and Lantry, B.F. 2002. Pelagic planktivores. *In* the 2001 annual report of the Lake Ontario and St. Lawrence River Units. Minutes of the 2002 meeting of the Lake Ontario Committee, Great Lakes Fishery Commission. Ann Arbor, MI.

Schneider, C.P., Schiavone, A., Jr., Eckert, T.H., McCullough, R.D., Lantry, B.F., Einhouse, D.W., Chrisman, J.R. Adams, C.M. Johnson, J.H., and Ross, R.M. 1999. Double-crested cormorant predation on smallmouth bass and other fishes of the eastern basin of Lake Ontario: overview, summary and recommendations. Section 1, pp. 1-6. New York State Department of Environmental Conservation Bureau of Fisheries and United States Geological Survey Biological Resources Division, 1999.

Scott, W.B. and W.J. Christie. 1963. The invasion of the lower Great Lakes by white perch, *Roccus americanus* (Gmelin). J. Fish. Res. Board. Can. **20**:1189-1195.

Seifried, K.E. 2000. Submerged macrophytes in the Bay of Quinte: 1972-2000. Department of Zoology, University of Toronto, Toronto, ON: 125.

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.

Stanfield, L.W., and Jones, M.L. 2003. Factors influencing rearing success of Atlantic salmon stocked as fry and parr in Lake Ontario tributaries. N. Am. J. Fish. Manage. **23**: 1175-1183.

Stewart, T.J., Lange, R.E., Orsatti, S.D., Schneider, C.P., Mathers, A., and Daniels, M.E. 1999. Fish-community objectives for Lake Ontario. Great Lakes Fish. Comm. Spec. Pub. 99-1.

Stewart, T.J., Hoyle, J.A, Mathers, A., and Schaner, T. 2002. Biological reference points for management of a declining walleye (*Stizostedion vitreum*) population in the Bay of Quinte and eastern Lake Ontario. Part III. Special Studies. *In* Lake Ontario fish communities and fisheries: 2001 annual report of the Lake Ontario Management Unit. Ontario Ministry of Natural Resources, Picton, ON, Canada. Available from http://www.glfc.org/lakecom/loc/mgmt_unit/index.html.

Stone, U.B. 1947. A study of the deep-water ciscoe fishery of Lake Ontario with particular reference to the bloater *Leucichthys hoyi* (Gill). Trans. Am. Fish. Soc. **74**: 230-249.

Stone, W.B., and Okoniewski, J.C. 2002. Botulism in Lake Erie. N.Y. State Conserv., June 2002. pp. 12-14.

Sullivan, W.P., Christie, G.C., Cornelius, F.C., Fodale, M.F., Johnson, D.A., Koonce, J.F., Larson, G.L., McDonald, R.B., Mullett, K.M., Murray, C.K., and Ryan, P.A. 2003. The sea lamprey in Lake Erie: a case history. J. Great Lakes Res. 2**9**(Suppl. 1): 615-636

Taraborelli, A.C., and Schaner, T. 2002. Diet of round goby in the Bay of Quinte, Lake Ontario. Part III. Special studies. *In* Lake Ontario Management Unit, 2002 Annual Report. Ontario Ministry of Natural Resources, Picton, Ontario. Available from http://www.glfc.org/lakecom/loc/mgmt_unit/index.html

Todd, T.N., and Smith, G.R. 1992. A review of differentiation in Great Lakes ciscoes. Pol. Arch. Hydro. 39, No. 3-4 (1992): 261-267.

U.S. Environmental Protection Agency. 2003. Survey of sediment quality in Lake Ontario, a summary of sampling results from September 1997. U.S. Environmental Protection Agency, Division of Environmental Science and Assessment, Edison, NJ.

U.S. Fish and Wildlife Service. 2001. Management of double-crested cormorants in the United States. Draft environmental impact statement. U.S. Department of the Interior, Washington DC.

Wells, L., and Beeton, A.M. 1963. Food of the bloater, *Coregonus hoyi*, in Lake Michigan. Trans. Am. Fish. Soc. **92**: 245-255.

Wells, L. 1969. Fishery survey of U.S. waters of Lake Ontario. Great Lakes Fish. Comm. Tech. Rep. 14. pp. 51-57.

Weseloh, D.V., and Collier, B. 1995. The rise of the double-crested cormorant on the Great Lakes: winning the war against contaminants. Great lakes fact sheet. Canadian Wildlife Service, Environment Canada and Long Point Bird Observatory.

Weseloh, D.V., and Ewins, P.J. 1994. Characteristics of a rapidly increasing colony of double-crested cormorants (*Phalacrocorax auritus*) in Lake Ontario: population size, reproductive parameters and band recoveries. J. Great Lakes Res. **20**: 443-456.

Weseloh, D.V.,and Pekarik, C. 1999. Numbers of double-crested cormorant nests in Lake Ontario colonies, 1995-1999. Canadian Wildlife Service, Downsview, ON.

Weseloh, D.V., Ewins, P.J., Struger, J., Mineau, P., Bishop, C.A., Postupalsky, S., and Ludwig, J.P. 1995. Double-crested cormorants of the Great Lakes: changes in population size, breeding distribution and reproductive output between 1913 and 1991. Colonial Waterbirds 18 (Spec. Pub. 1): 48-59.

Weseloh, D.V., Pekarik, C., Havelka, T., Barrett, G., and Reid, J. 2002. Population trends and colony locations of double-crested cormorants in the Canadian Great Lakes and immediately adjacent areas, 1990-2000: a manager's guide. J. Great Lakes Res. **28**:125-144.

Williams, D.J., Kuntz, K.W., L'Italien, S., and Richardson, V. 2001. Great Lakes surveillance program: organic contaminants in the Great Lakes, 1992 to 1998. Intra- and inter-lake spatial distributions and temporal trends. Environmental Conservation Branch—Ontario Region, Ecosystem Health Division, Environment Canada, report no. EHD/ECB-OR/01-01/I.