Volume 2, Number 2 July 1996

Great Lakes Research Review

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GREAT LAKES RESEARCH CONSORTIUM
Sea Grant New York

GREAT LAKES FISHERIES

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Great Lakes Research Review

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The *Great Lakes Research Review* is published cooperatively by the Great Lakes Program, the Great Lakes Research Consortium, and New York Sea Grant. The publication is designed to provide an in-depth look at a particular topic or area of research related to the Great Lakes. Although not intended to be a peer reviewed journal, the publication offers a substantive overview of research being conducted in the Great Lakes Basin.

The Great Lakes Research Review is offered to inform researchers, policy-makers, educators, managers, and stakeholders about the research efforts taking place in New York, Ontario, and throughout the basin.

This is the second issue of our latest topic: **Great Lakes Fisheries**. Other previous issues focused on the fate and transport of toxic substances, and the effects of toxics. All highlight the work of researchers associated with the sponsoring organizations and others who were involved in the specified research area.

We gratefully acknowledge all of the contributing authors who willingly shared their research efforts for this publication. Our appreciation is extended to David MacNeill, Extension Specialist of the New York Sea Grant Extension Program for writing the guest commentary for this issue.

Questions concerning this issue may be addressed to Jack Manno, Executive Director, Great Lakes Research Consortium. Those interested in obtaining copies of previous issues may contact the Great Lakes Research Consortium.

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Great Lakes Research Review

GREAT LAKES FISHERIES

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Great Lakes Research Review
Volume 2 , No. 2 , July 1996

INTRODUCTION

Michael J. Connerton
Great Lakes Research Consortium

Welcome to the latest issue of the Great Lakes Research Review. The purpose of this publication is to bridge the gap between the investigators who have spent their careers on Great Lakes ecosystem research and others who may want the information in less technical terms. The lakes are surrounded by two countries, two provinces, eight states, hundreds of cities and towns, millions of stakeholders, and scores of organizations dedicated to research and conservation of the Great Lakes. The complexity of the human condition in the Basin parallels the intricate web of the ecosystem within the lakes. Managers, researchers, and citizens alike must gather and process data and draw conclusions about many issues, but what information they gather can determine what conclusions they draw. Depending on the source and type of information, inferences may be slightly different, contradictory even, and that causes uncertainty for all involved. This publication offers a solution to this problem by providing information that is understandable to researchers, policy-makers, educators, managers, and stakeholders about the research efforts taking place in New York, Ontario, and throughout the basin. It is our hope that this source will bring its readers together on the important subjects it addresses.

The articles presented in this volume deal with Great Lakes Fisheries, which has been one of the most formidable and controversial research areas. Lake Ontario is particularly discussed in this issue because it has posed a daunting challenge to fisheries managers due to the dramatic changes occurring from reduced lake productivity, and general uncertainties about the sustainability of the present fishery.

Managers have recently decreased salmonine stocking, an unpopular management decision among salmon fishermen. Public perception is that resource-users are not represented well enough in the decision-making process. Some of the varying opinions are discussed by David MacNeill is his guest commentary *Lay Perspectives on Lake Ontario Fisheries.* Results of informal surveys and numerous interactions with lay audiences by the New York Sea Grant reveal several common areas of public concern with the future and current directions of Lake Ontario fisheries.

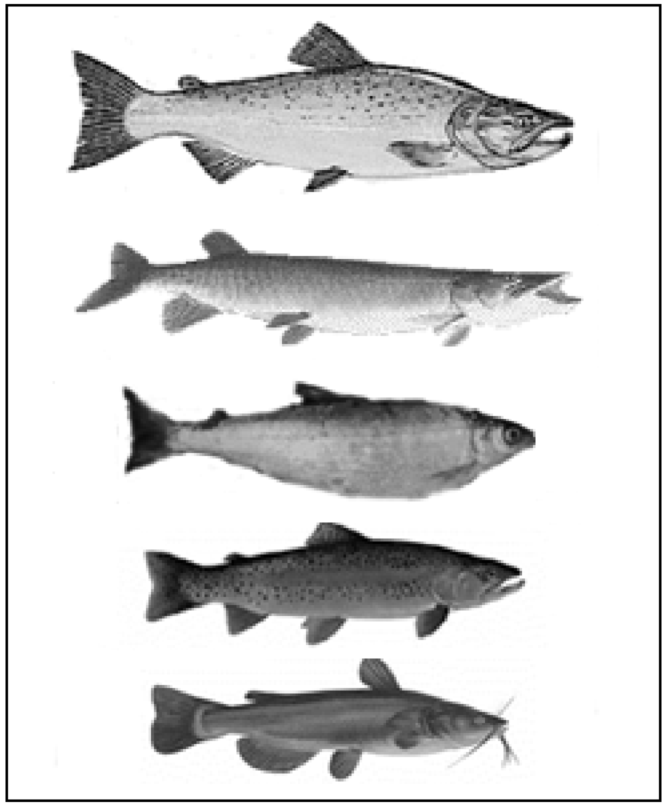
To address the conundrum of Lake Ontario, the New York State Department of Environmental Conservation (DEC) convened a panel of experts and asked for a review of the current status of the Lake Ontario salmonine fishery. Lars Rudstam and his colleagues present a summary of the findings in their article entitled, *A Review of the Current Status of Lake Ontario's Pelagic Fish Community: A Report from the 1996 Lake Ontario Technical Panel*. The panel reviewed information provided by the DEC and by letters from the charterboat industry and other concerned citizens. In addition, several panel members presented their own findings on aspects of the lake ecosystem. Using all of these sources, the panel drew their conclusions and stated that "managing the Lake Ontario pelagic fish-

> ***... Lake Ontario is particularly discussed in this issue because it has posed a daunting challenge to fisheries managers due to the dramatic changes occurring from reduced lake productivity, and general uncertainties about the sustainability of the present fishery.***

eries through salmonine stocking is a balance between return and risk". Managers must weigh the possibility of increased risk to the chinook fishery against the increased salmonine return from higher levels of stocking.

One of the tools that Great Lakes fisheries researchers utilize to assess pelagic stocks is underwater acoustics. Acoustics combined with bioenergetics models are useful for evaluating the relationship of trophic supply and predator demand. This is discussed by Steve Brandt and Darryl Hondorp in their article, *Spatially Explicit Models of Fish Growth Rate: Tools for Assessing Habitat Quality.* They present a brief, intelligible description of the techniques, and give examples of applications in current research.

The complexity of the Lake Ontario system does not stop with alewife, salmonids, or smelt. Another significant component to Lake Ontario fisheries is the lake whitefish because of its status as the most important commercial species. Recently, the whitefish has experienced a resurgence producing a population at least twice as great as at any time since 1900, according to *Resurgence of Lake Whitefish in Lake Ontario in the 1980's* by Casselman, Hoyle, and Brown. The recovery resulted from increased survival of young due to winterkill of predators (alewife and white perch). The article points out that Eastern Lake Ontario has two major spawning stocks and renewal of these stocks signals the re-establishment of a more diverse, self-sustaining cold water fish community.



From Top to Bottom: chinook salmon, muskellunge, lake whitefish, rainbow trout, black bullhead (not drawn to scale).

The two last articles of this issue take the reader out of Lake Ontario and into the St. Lawrence River. Werner et al. present their findings from a study on habitat utilization by another important species, the muskellunge. Their article, *Vegetative Characteristics Of Muskellunge Spawning And Nursery Habitat In The 1000 Islands Section Of The St. Lawrence River*, emphasizes that the nature and distribution of aquatic vegetation in spawning and nursery areas are important components affecting the survival of muskies during the initial period of their life. The dynamics of the vegetation and the changing needs of the young muskellunge present a challenge to fisheries managers who want to establish self-sustaining populations of muskellunge.

In the final article, we shift away from the ecology of fishes to physiology and toxicology. Thomas Moon and his colleagues report on data indicating that different fish respond differently to contaminants, and warn against using levels of enzyme activation as indices of contaminants. The inability of rainbow trout to detoxicate accumulated PCBs could ultimately impact the survival of this species in a highly contaminated area and their suitability for human consumption, whereas the black bullhead may not be so affected.

It is fitting that this issue of the Review, which skims the surface of Great Lakes fisheries topics, ends in the St. Lawrence River. The St. Lawrence is the doorway from the Lakes to the Atlantic Ocean and through it passes oceangoing vessels, anadromous fish, and other aquatic life. The Lakes may end in the river, but the St. Lawrence is just the beginning for many issues in the Basin. Exotic organisms that arrive in the river from vessels passing through have the potential to completely transform the ecosystem of the Basin. Not only could the various plant and fish communities be disrupted or destroyed by the invaders, but the subsequent changes could lead to a decrease in water quality and human usage. Recent introductions (i.e., zebra mussels, European ruffe, predaceous cladocerans) have already caused problems and pose a significant challenge to managers and stakeholders alike. Due to the importance of this predicament, the next issue of the Review will be dedicated to articles addressing *exotic species*.

Anyone interested in submitting an article or would like more information regarding authors' guidelines should contact the Consortium at the address listed in the front cover of this publication.

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Guest Commentary

Lay Perspectives on Lake Ontario Fisheries

D.B. MacNeill
New York Sea Grant Extension Program

Dramatic changes to the Lake Ontario ecosystem, resulting from reduced lake productivity, and the resulting uncertainties for the future sustainability of the salmonine sportfishery have generated concern among researchers, fisheries managers, and lay sportfisheries stakeholders including recreational anglers and the charterboat captains. Throughout the past decade, various educational and outreach efforts have been conducted by NY Sea Grant, academia, NYSDEC, OMNR, USFWS, and NBS to facilitate improved understanding of the complexities of Lake Ontario fisheries for lay resource users. Many of these activities for the lay community were the result of collaborations between management and research and were mediated by extension. These forums included highly-respected fisheries professionals as technical resources.

The prime objective of this outreach, particularly on the extension side, has been to reduce polarity between fisheries stakeholder groups, facilitated through improved dialogue. Vehicles for this holistic approach include annual State of Lake Ontario seminars, factsheets, *Charterlines* (a quarterly newsletter for the charterboat industry published by NY Sea Grant) and NYSDEC taskforce meetings. Although this proactive outreach has been successful in improving the understanding of many facets of Lake Ontario ecosystem, many members of the lay community are in disagreement with various conclusions from scientific community and question the solutions to maintain an economically viable sportfishery. The low acceptance of the information by many members of the lay community has been a source of frustration for all concerned.

The NYSDEC convened two scientific review panel meetings in an attempt to define the current state of knowledge of the lake ecosystem in 1992 and 1996, followed by publication of panel reports. These panel meetings were initiated following some disturbing fisheries trends. Following the first meeting, a public participation process component was developed by NYSDEC to solicit input from resource user groups as to their desired vision of the sportfishery over a range of management scenarios. The consensus of public opinion was that management actions should be encouraged that promote a sustainable alewife population so as to maintain an economically viable sportfishery, with chinook salmon occupying a prominent role as a target species.

The resulting decision by NYSDEC to reduce predatory demand by 50% on the declining alewife stock through a stepwise stocking reduction of the key predatory species, chinook and lake trout, was implemented in 1992. This decision was met with resistance from many resource users as to the appropriateness of the action due to questions on the integrity of the scientific data because of perceived flaws in the data collection and interpretation process. In addition, many lay stakeholders felt that the stocking reduction was implemented without sufficient consideration of socioeconomic impacts of this management action.

During the second iteration of the scientific review of Lake Ontario in 1996, two respected leaders in the charterboat community were included as observers. Both leaders viewed the review process and the credential of the scientific participants favorably. Despite their credence in the validity of the second review process and in the report supporting the course of fisheries management and research in Lake Ontario, many lay stakeholders still question the propriety of certain management actions (i.e., stocking reductions) and research directions, yet the proportion of lay stakeholders accepting the notion of depressed lake productivity has actually increased.

Results of informal surveys and numerous interactions with lay audiences by NY Sea Grant Extension reveal several specific areas of concern as to the future and current directions of Lake Ontario fisheries. As a preface to presenting these lay perspectives, again obtained largely from anglers and charterboat captains, it should be noted that this summary is neither a critique or admonishment of Lake Ontario research, management or outreach policy, nor is it an opportunity to ridicule the lay stakeholders. It is merely a reminder that public acceptance of scientific information is often compromised by the remote and abstract nature of science.

SUMMARY OF LAY PERSPECTIVES

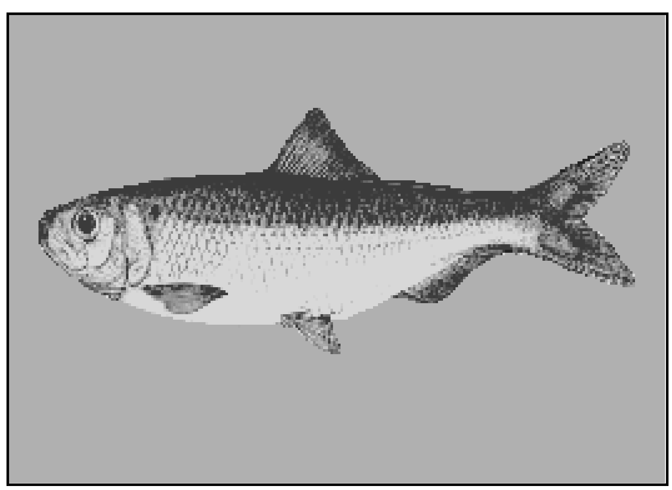
- ***Although declining lake productivity may be occurring and is the most likely the reason for ecosystem (fisheries) changes, the fisheries impacts could have been anticipated, mitigated and may yet be remediated.*** Scientists have known since the late 1970s that phosphorus levels have been declining and would result in a reduction in the lake's fisheries production. Questions such as, why was nothing done to mitigate the inevitable decline in fish production, could this situation have been prevented, and what was the economic trade-off between improved water quality and reduced revenues from a declining fishery, are common. The aggressive lake restoration efforts are an indication that during this restoration program, little consideration was given towards maintaining an economically viable sportfishery. By relaxing the phosphorous abatement program, lake fertility could be adjusted to maintain the sportfishery at its peak levels of production.

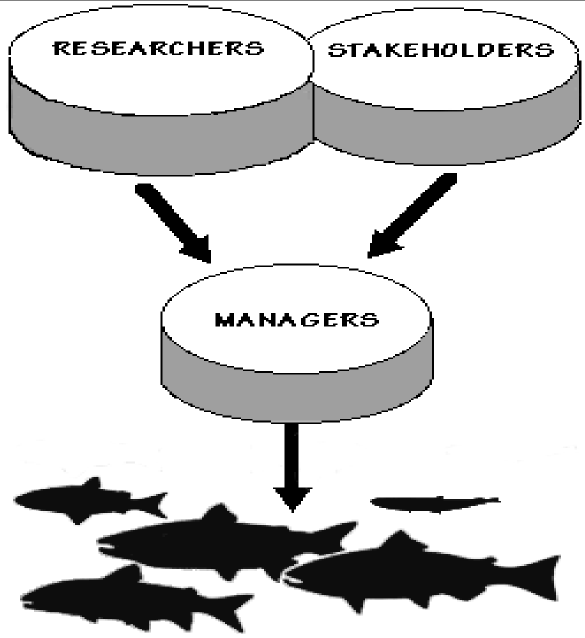
- ***Anecdotal information from fishermen is as valid or equivalent to scientific observations in fisheries management decisions.*** Sportfishery stakeholders desire to be part of a fisheries management team with researchers and managers. Because of the extensive time spent out on the lake, the lay community routinely observe phenomena that are not often seen by biologists. The stakeholders feel that they do not have a sufficient voice in the decision-making process and that their anecdotal information should be included as data relevant to fisheries management. Management decisions should also be geared more towards the improvement of the economic value of the sportfishery.

- ***Ecosystem modeling as a management tool is flawed.*** The perception exists from many lay stakeholders that researchers can often lose credibility when they try to analyze and predict ecosystem dynamics from model simulations. Models have no real-world utility since they are based on pie-in-the-sky assumptions and the parameters incorporated into models are inaccurate. Some researchers are too remotely associated to the lake and have little practical grasp of the dynamics, because they spend excessive time modeling on the computer and not enough time in the field. Models have no practical use in fisheries management of research and should not be used in the decision-making process.

- ***The alewife sampling program is flawed.*** The current alewife sampling program does not provide an accurate picture of abundance due to a lay perception of low spatial overlap between sampling locations and actual distribution. Trawl catches are not representative of the population since trawling is restricted to offshore areas when many alewives are observed closer to shore by anglers. Trawl samples capture only smaller fish since large fish are only found in predator stomachs. Hydroacoustic assessment is based on technology that is too new to be reliable and should not be incorporated into management decisions until the bugs are worked out.

The alewife , which was introduced to the Great Lakes in the 1870's has become a siginificant component in the Lake Ontario ecosystem.





The desire by the lay stakeholders to become more closely involved with researchers and managers should be capitalized upon...

- The importance of chinook salmon to the sportfishery is not sufficiently recognized. Chinook is the bread and butter species of the sportfishery, and generates considerable revenue. By reducing chinook stocking, fewer fish are surviving due to heavy predation by predators and lampreys resulting in lower sport catches. This may also be due to a poorer hatchery product. Current chinook survival estimates are too crude to have any validity in managing the fishery and bias chinook impact on the forage base. Recent increases in chinook harvest and growth in Lake Michigan indicate that chinook are less dependent on alewives as prey. Stocking should be reinstated to original levels, acknowledging that there is some unknown risk to the forage base. This is justified by larger alewives present in 1995, an indication that alewives are healthy and can withstand additional predation. Stocking reductions were based on insufficient evidence of chinook survival, errors in alewife abundance and also reflect a bias of fisheries managers towards restoration of native species at the cost of a put-grow-and-take fishery, with chinook as the sacrificial lamb in this strategy. Excessive effort has been spent on lake trout restoration and as a sportfishery species, it is difficult to promote because of the slot limit size regulation, high post-angling mortality during the summer and high contaminant burdens. Paradoxically, there is concern that should lake trout restoration be eliminated, would sea lamprey control follow suit?

IMPLICATIONS

The implications of these perspectives are that dialogue between all interest groups should be maintained, if not intensified due to the continually changing ecosystem and potentially escalating conflicts between stakeholder groups that may result. The desire by the lay stakeholders to become more closely involved with researchers and managers should be capitalized upon, but not necessarily in the decision-making process. (The recent financial support donated to the NBS by the lay stakeholders is an example of the level of commitment to ongoing fisheries assessment programs and the willingness to cooperate that exists, despite differing views.) For example, information on species catches, and predator stomach contents from trained lay volunteers could be of value to researchers and managers. At ongoing educational forums, more emphasis could be placed on explaining the mechanics of sampling, differentiating anecdotal vs. scientific information, outlining the process of scientific investigation, and designing are search project, in addition to regular information updates. Case histories of other similar fisheries issues outside Lake Ontario should also be included. Effort could also be expended in the development of a contingency plan or a proactive agency response from management and outreach in the event of a catastrophic ecosystem change.

For researchers, reminders of several fisheries research priorities emerge. Examples of these target areas could include improving the precision of acoustic forage assessment programs, improving post-stocked salmonine survivorship estimates, monitoring salmonine feeding habits, quantifying effects of increasing light penetration in the lake on predator-prey interactions, their distribution and vulnerability to capture by trawls (forage fish) and angling (salmonines), and improving the parameters used in ecosystem modeling assessing socioeconomic impacts of the sportfishery. Researchers should also be encouraged to continue their participation in future public forums.

Dave MacNeill has been an extension specialist in Great Lakes Sportfisheries with the NY Sea Grant Extension Program since 1987. He is the editor of Charterlines and a technical advisor to the National Zebra Mussel Information Clearinghouse and a contributing author to the Clearinghouse newsletter, Dreissena. Dave has also served as a technical liaison to the NYSDEC Lake Ontario Fisheries Taskforce and the Cormorant Taskforce.

Great Lakes Research Review
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A Review of the Current Status of Lake Ontario's Pelagic Fish Community. Report of the 1996 Lake Ontario Technical Panel.

Technical Panel:

Stephen Brandt, Great Lakes Center, Buffalo State College.
Joseph DePinto, Great Lakes Program, SUNY at Buffalo.
Michael Hansen, National Biological Service, Great Lakes Science Center, Ann Arbor, MI.
Kyle Hartman, Great Lakes Center, Buffalo State College.
Edward Mills, Cornell University.
Robert O'Gorman, National Biological Service, Oswego Station.
Peter Rand, University of British Columbia, Vancouver, BC.
Howard Riessen, Buffalo State College.
Lars Rudstam, Cornell University.
Randal Snyder, Buffalo State College.
Donald Stewart, SUNY College of Environmental Science and Forestry, Syracuse.

Original Report Edited by:
Lars Rudstam, Cornell University Biological Field Station
900 Shackelton Point Road, Bridgeport, NY 13030

This article contains the executive summary of a full report submitted to the NYSDEC. The full report includes the background data on which the conclusions in this summary are based. For a copy of the full report, interested parties should contact Mr. Robert E. Lange, New York State Department of Environmental Conservation, 50 Wolf Road, Room 552, Albany, New York 12333-4753.

INTRODUCTION

In 1993, the New York State Department of Environmental Conservation (NYSDEC) and the Ontario Ministry of Natural Resources (OMNR) decided to decrease the predatory demand from salmonines in Lake Ontario by 50%. This was done by decreasing the stocking rate of lake trout and chinook salmon. Chinook salmon stocking was decreased from 2.6 million (1992) to 1 million annually in 1994 and 1995 (1.7 million were stocked in 1993) and lake trout stocking was reduced from 2 million in 1992 to 1 million per year in 1993 -1995. This decision was based on findings by a technical panel (Jones and Stewart 1992) and an extensive public participation process during 1992. The goal of this stocking reduction was to protect the alewife population and thereby decrease the risk of a disease outbreaks and declines in salmonines, particularly chinook salmon, similar to that observed in Lake Michigan during the 1980s.

However, the decision to cut salmonine stocking did not have unanimous public support. Therefore, NYSDEC asked for a review of the current status of Lake Ontario by a panel of experts during January and February 1996. Specific tasks asked of this panel was to:

1) Evaluate the earlier report "Status of the Lake Ontario Pelagic Fish Community and Related Ecosystem in 1992" (Jones and Stewart 1992) and comment on the extent to which the conclusions remain relevant in 1996.
2) Evaluate the status of prey populations, especially

alewife and rainbow smelt, in Lake Ontario, with emphasis on their capability to sustain the level of predator demand implied by current stocking rates.

3) Evaluate the anticipated effects, on a risk basis, of increases in predator demand that would result from stocking increases with regard to the status of the prey populations and trout and salmon production.

The panel was given the freedom to change these objectives as it considered appropriate. NYSDEC promised to provide any information requested by the panel and assistance with distribution, printing, etc., of this report, but the content was not to be edited, proofread or altered by NYSDEC. This was explicitly stated as part of the panel's initial instructions. Thus, this report and the conclusions and suggestions put forth are the opinions of the Technical Panel and not necessarily of NYSDEC.

The Technical Panel met for two days (Jan. 10-11) and for one additional day (Feb. 3) at the Cornell Biological Field Station (Bridgeport, NY). The panel consisted of eleven scientists from Cornell University, Buffalo State College, SUNY at Buffalo, SUNY-ESF at Syracuse, the National Biological Service, and the University of British Columbia. In addition, Mr. Robert Lange and two representatives from the charter industry (Mr. Larry Watts and Mr. Frank Sanza) were present as observers.

The panel reviewed information provided by NYSDEC and by letters from charter industry representatives and other concerned citizens. In addition, several panel members presented information on aspects of the Lake Ontario ecosystem (O'Gorman on alewife and smelt abundance, Mills on zooplankton and nutrients, Rudstam on mysids, DePinto on nutrient loading and an ecosystem model, Rand and Stewart on salmonine diets and results from a new alewife population model).

BACKGROUND TO THE PROBLEM

The decision to reduce salmonine stocking in Lake Ontario was based on the following observations (Jones and Stewart 1992):

A.) Phosphorus levels, primary production, and zooplankton production had declined through the 1980s.

B.) Biomass of alewife and smelt, the primary forage fish of salmonines, had declined through 1992.

C.) The number of larger alewife and smelt sizes selected by salmonines had decreased proportionally more than the rest of these populations.

D.) Even though alewife and smelt populations declined, the condition of remaining fish did not improve.

E.) Abundance of salmonines remained high with a peak in 1986 indicating continued high predatory demand from these fish. Bioenergetics models predicted that decrease in chinook stocking would be the most efficient way to decrease predatory demand by the salmonine population in Lake Ontario.

F.) The severity of a disease outbreak (bacterial kidney disease, BKD) in the chinook salmon population of Lake Michigan was attributed to decreased food availability increasing the stress on the fish.

These observations led the 1992 technical panel to concluded that the 1992 stocking levels of salmonines in Lake Ontario was not sustainable. The panel argued that the alewife population was affected by both lower productivity and by high predatory demand from salmonines. Given these stresses, the alewife (and smelt) populations were considered at risk, especially if decimated by an unusually cold winter (as observed in 1976/77). A further decline or a collapse of the alewife population was considered likely to increase the risk of a disease outbreak in Lake Ontario, comparable to the outbreak of BKD in chinook salmon in Lake Michigan, with resulting declines in salmonines, especially chinooks. These expectations were supported by a model of fish species interactions in the lake (SIMPLE, Jones et al. 1993). To protect the alewife population and, by extension, the chinook and coho salmon populations (the two species that most heavily rely on alewife), NYSDEC and OMNR decreased stocking of chinook salmon by 60% and lake trout by 50%, staged over two years (1993 and 1994).

Critics of the NYSDEC and OMNR decision has put forth a list of counter-arguments. These arguments can be grouped in three categories:

1.) Evidence that the current alewife population cannot support the 1992 salmonine stocking levels are weak at best.

2.) The risk of depleting the alewife population is outweighed by the benefits of maximizing chinook production and by the possibility of higher numbers of valuable native species (yellow perch, walleye, whitefish, lake herring, natural lake trout reproduction).

3.) Stocking cuts, if needed, should be made on species other than chinook salmon.

Suggestions put forth by concerned citizens for consideration by the panel were (grouped to contrast the observations by the 1992 panel):

A.) Phosphorus content in Lake Ontario decreased in the 1970 and early 1980s without further decrease in the late 1980s and 1990s. Thus, alewife production was high enough to support the high salmon stocking levels in the late 1980s and early 1990s even at the current low phosphorus loading rates.

B.) Alewife populations may have declined but abundances are high enough to support the higher stocking levels.

C.) The signs of predation effects on the alewife and smelt population (decreased number of larger and older fish) is due to natural mortality of the large alewife that were produced by excellent growth rates of fish surviving the 1976/77 winter die-off and not to predation by salmonines.

D.) The continuing low condition of alewife indicates that the fish is still too abundant and that the alewife population (and production) would benefit from increased mortality (improving conditions for remaining fish).

E.) There are large uncertainties in the calculation of predatory demand by salmonines associated with large uncertainties for early mortality of chinook salmon. This questions both the validity of calculations showing chinook salmon to be the major contributor to salmonine predatory demand and the validity of calculations that predators consume up to 30-40% of annual alewife production.

F.) The decrease in chinook salmon in Lake Michigan was due primarily to BKD and not to decreased alewife stocks, a position held by some Michigan DNR biologists.

Two additional points made by concerned citizens were discussed:

G.) Decreased salmonine stocking rates will increase mortality from sea lamprey.

H.) Compensatory responses to lower alewife populations by both the alewives themselves (increased condition and recruitment) and by the salmonines (decreased growth rates, increased mortality possibly through cannibalism) makes the system more resilient than assumed in the previous models. This would decrease the risk associated with higher stocking levels.

Some of these opposing views can be resolved by careful interpretation of available information, others can not. Still others are beyond the scope of this report. For example, we will follow the opinion of the Great Lakes Fish Disease Control Committee from March 1993 (letter to the Great Lakes Fisheries Commission) that BKD is a stress related disease and that an "ecosystem imbalance" causes BKD to be expressed in chinook salmon. This panel does not have the expertise to evaluate the alternative even though it is important. If chinook salmon declines in Lake Michigan were not caused by stress due to low forage abundance, the arguments put forth later in this report may be invalid.

EXECUTIVE SUMMARY OF PANEL FINDINGS

We now have an additional three years of data not available to the 1992 task force. Of these three years, 1993 is unaffected by decrease stocking, 1994 should have a 13% decrease and 1995 a 25% decrease in predatory demand if mortality rates of salmonines stayed the same as in prior years. The reason the full 50% reduction would not yet be fully realized is that there is a time lag from the time stocking cuts are implemented to the time these cuts affect the total salmonine population. For example, because most chinook salmon mature at age 3 and 4, it will take about four years for stocking cuts to affect the whole population of chinooks. The time lag for lake trout is longer because lake trout may live more than 10 years.

The panel considered both the validity of the conclusions of the 1992 technical panel that suggested stocking cuts and the arguments put forth opposing such cuts.

The panel considered both the validity of the conclusions of the 1992 technical panel that suggested stocking cuts and the arguments put forth opposing such cuts. This report is our evaluation of the merits of these positions evaluated using the additional data available from 1993-1995 and through two new models: an ecosystem model by Jain and DePinto (1996) and a risk analysis of a food web model centered around alewife population dynamics (Rand, Stewart, and O'Gorman, ms. in prep.).

Our review of the data available to the panel lead us to a number of conclusions. These are summarized below. The background data for each of these statements are provided in the full report submitted to the NYSDEC. For copies of the full report, readers should contact the NYSDEC at the address provided at the beginning of this article.

Phosphorus loading and lower trophic levels

1.) Phosphorus loading levels have been fairly constant since the beginning of the 1980s. Lake concentrations of total phosphorus (both spring and summer) declined at least through 1992 (last year of data) although most of this decline had occurred prior to 1985.
2.) Summer chlorophyll-a levels (measure of algae biomass) declined through the 1980s from about 4 µg/L in 1968-80 to about 2 µg/L in 1990-92. Spring total chlorophyll-a concentration remained between 0.7 and 2.2 µg/L since 1970.
3.) In 1995, nearshore chlorophyll-a levels were much lower relative to total phosphorus content of the water than either the offshore or the embayments.

Comments: There is always a lag between decreases in phosphorus loading and the full effect on in-lake phosphorus levels. This lag is associated with water residence time and with release of phosphorus stored in the sediments. We argue this is the reason for the approximately five year time lag between decrease in phosphorus loading and decrease in in-lake total-phosphorus levels.

The lack of an effect of lower spring total-phosphorus levels on spring chlorophyll-a concentration indicates that the spring bloom is not phosphorus limited. The spring bloom consists of diatoms which may be limited by silica and not phosphorus. Since the spring bloom largely settles on the bottom and feeds the benthic community, there should be less effect of lower nutrient levels on benthic than on pelagic secondary production through the 1990s.

The slow decline in in-lake phosphorus levels after 1985 compared to the beginning of the 1980s is likely the reason for the different views expressed earlier. Thus, there is merit to both points of view, phosphorus content has continued to decline, but this decline has been small since 1985.

The low ratio between algae and phosphorus in nearshore waters compared to what we expected from general relationships may be due to grazing by dreissenid mussels (zebra and quagga mussels). The full effect of dreissenids has likely not yet been realized.

We do not expect large changes in phosphorus loading, total phosphorus concentrations or offshore chlorophyll-a levels during the 1990s. The relationship between phosphorus loading and reductions in lake phosphorus concentrations should now be stabilized. Increasing dreissenid mussels could affect the nearshore phosphorus levels but will likely have little effect on lake-wide algal biomass due to the large volume of water relative to bottom area (which limits dreissenids) in Lake Ontario.

Fish populations

4.) The alewife population has declined steadily and reached very low levels in 1994, possibly the result of an unusually severe winter in 1993/94. A large proportion of the current population is from the 1991 year class.
5.) The dip in the alewife population in 1994 was accompanied by increases in size of zooplankton, increases in growth and condition of alewife, and increased reproduction of various species including lake trout and yellow perch.
6.) The growth of chinook and coho salmon declined in 1994, likely a response to lower food abundance. Age at maturity of chinook salmon continued to increase indicating slower average growth of individuals. However the condition of these species did not decline in 1994.
7.) The continued low abundance of larger alewife and smelt, the most preferred prey of salmonines, indicate high predation pressure on these fish populations. This is not the result of disappearing larger alewife produced after the increased growth rates in 1978 following the winter die-off of 1976-77.
8.) The smelt population now consists of primarily one year class indicative of a highly exploited population. Even so, the biomass of smelt increased in 1995 compared to 1994.
9.) The alewife population posted a small increase in 1995. This increase was accompanied by a modest decline in adult condition and increased size of chinook and coho salmon relative to 1994; zooplankton size did not change.
10.) Decreasing stocking rates will not increase the number of salmonines dying from lamprey attacks. However, the proportion of salmonines affected will increase if the salmonine populations decline.

Comments: We believe these data, particularly for 1994 and 1995 to be very important. First, the 1994 numbers suggest that the alewife population can become low enough so that both their prey (zooplankton) and their predators (salmonines) are affected. We believe this is the first time where negative effects on chinook salmon growth rates have been observed as a result of low alewife abundance (although coho salmon growth has been shown previously to be dependent on abundance of juvenile alewife in Lake Ontario; O'Gorman et al. 1987). In these situations, there might be a risk for increased diseases in the salmonine populations. Second, despite low

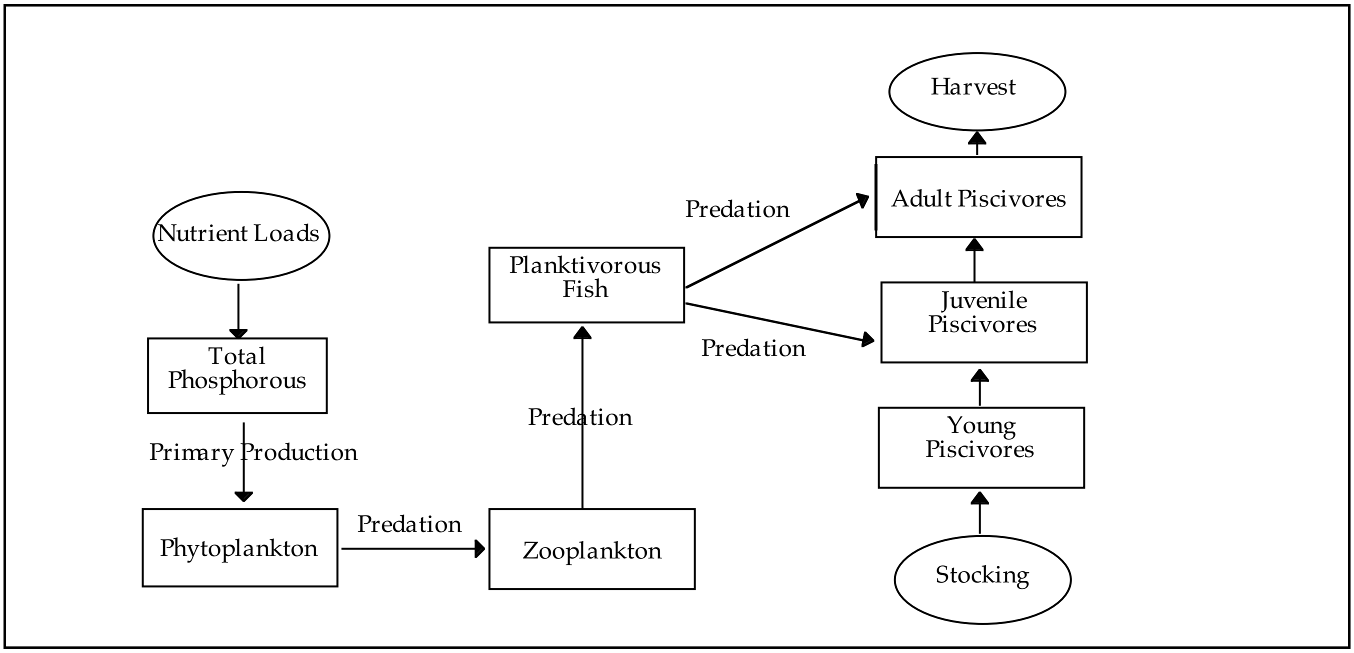

Figure 1. Simplified Lake Ontario ecosystem model (adapted from Jain and DePinto 1996)

abundance and high predation pressure, alewife biomass appears to have increased in 1995. This suggests that the alewife population is more resilient than previously thought. Compensatory mechanisms may allow alewife recovery even from low abundance levels. Three potential mechanisms may have operated in 1994: 1) lower predation rates by individual salmonines (indicated by lower growth rates); 2) increased mortality of salmonines; and 3) increased production of zooplankton causing better condition of alewife. Alternatively, the implemented stocking cuts may have caused a decline in predatory demand by salmonines (although predicted effects in 1994 were relatively minor). The main indication of high exploitation and a cause for concern for both the alewife and smelt populations is the dominance of one age class in both populations.

Model predictions

The panel reviewed two models that are being developed for improving our understanding of the Lake Ontario ecosystem and alewife population dynamics. The first model is a ecosystem model developed by Jain and DePinto (1996) which has been used to explore the combined effects of stocking rates and phosphorus loading (Figure 1). The second is a food web model by Rand, Stewart, and O'Gorman (ms. in prep.) centered around the dynamics of the alewife population. This model can be used to estimate the risk associated with different stocking levels of salmonines and will be referred to as the RISK model. This risk was evaluated as the proportion of 15 year simulations that included at least one year when the alewife population is low enough to cause chinook salmon to be prey limited.

11.) The ecosystem model (Jain and DePinto 1996) suggests that increasing the phosphorus load will have a stronger effect on salmonine biomass than an increase in stocking rates at the current state of the lake. Thus, the model predicts diminishing returns per stocked fish as stocking levels increase above the 1995 levels.

12.)The RISK model indicates that the alewife population is more resilient than previously thought and that the likelihood of a complete alewife collapse is small.

13.)With current zooplankton production and stocking rates, the risk of reaching prey limitations for chinook salmon is about 60%. This risk increases to over 80% if stocking levels are returned to the 1992 levels.

Comments: Any model predictions depend on the quality of input data (including limited data to translate trawl catches to lake-wide alewife abundance) and the ability of the model to correctly represent major interactions. Given these caveats, the tentative conclusion is that there is a relatively high probability that the alewife population will become low enough to cause chinook salmon to be prey limited even at current stocking rates. This risk increases if we return to the 1992 stocking levels (60% at current levels and over 80% at 1992 levels). On the other hand, predation by salmonines is not likely to cause a complete collapse the alewife population.

Future data needs

We believe we are moving forward towards understanding the interactions between salmonines, alewife and nutrient loading in Lake Ontario, but we still have a ways to go. Some additional important data may be available shortly, in particular refined acoustic estimates of forage fish biomass from 1991 though 1995. Accurate whole lake biomass estimates are critical for the interpretations posed here and for resolving some of the differing opinions listed earlier. We need to know that the 1994 alewife biomass is indeed low compared to previous years and that the 1995 biomass is indeed higher than 1994 to solidify the conclusions in this report. The preliminary acoustic biomass for 1993 and 1994 show continued decline in the forage fish biomass that is not mirrored in the trawl data. The fall 1994 biomass estimate is particularly low (23 kt; Schneider and Schaner 1995). We need an explicit analysis comparing the trawl abundance data with acoustic biomass estimates. The panel cautions against any long term management decision without full consideration of those data.

The cause and extent of alewife winter kills is not well understood. The risk associated with salmonine stocking is to a large degree related to the severity of winter alewife die-offs. Thus, we need to know if there are any interactions between size/condition of alewife and winter kills and what proportion of the alewife population is typically affected.

...managing the Lake Ontario salmonine fisheries through salmonine stocking is a balance between return and risk.

The lack of annual estimates of salmonine mortality rates is another problem, especially for estimating predatory demand. We need to know the importance of cannibalism for juvenile salmonine survival, and how cannibalism is correlated with abundance of other prey such as alewife. Higher cannibalism at low alewife abundance decrease predatory demand by the salmonines by increasing salmonine mortality.

In addition, the panel considers differences in annual growth rates, condition and energy content of salmonines, and changes in diets of salmonines to be important indicators of the status of the salmonine-alewife interaction in the lake. We recommend that steps be taken to initiate data collections to follow these indicators. Analysis of archived salmonine scales would give some information on changes in annual growth rates over time. Cooperation with charter captains is a potentially fruitful approach to collect diet data. For data on salmonine annual growth, condition and energy content, gill net surveys may be more appropriate as fish in poor condition are less likely to be caught by anglers than fish in good condition.

CONCLUDING REMARKS

Based on the information available to us at present, the panel concludes:

1.) The increase in alewife biomass in 1995 despite relatively high stocking rates and a low alewife population in 1994 indicates that the alewife population is more resilient than previously suggested. This is also the inference from the RISK model which indicates that the risk of collapsing the alewife population with either stocking scenario is minimal.

2.) Returning to higher stocking levels will increase the risk for prey limitation and decreased salmonine growth rates and therefore increase the risk for disease outbreaks, especially in chinook salmon. Our best estimates suggest a 30% higher risk for this to occur with the higher stocking rates than with the current stocking rates.

3.) Returning to higher stocking rates should improve chances for increases in native fish species by keeping the alewife population low. However, it should be remembered that species that could increase include both native species considered valuable (i.e., whitefish, lake herring, lake trout, yellow perch, walleye) and less valued species (both native such as burbot and exotics such as gobids and ruffe). For example, burbot has increased dramatically in Lake Michigan and will likely compete with lake trout and other salmonines (Rudstam et al. 1995). Surprises should be expected when fish communities change.

Managing the Lake Ontario salmonine fisheries through salmonine stocking is a balance between return and risk. The current lowered stocking rates will likely maintain fast growing chinook and coho salmon based on alewife as the primary forage fish. A return to higher stocking rates will likely increase the salmonine population, but also increase the risk of disease outbreaks and slower growth rates. Unfortunately, the risk may be large because an outbreak of, for example BKD in chinook salmon, is very hard to eradicate from the system (as the Lake Michigan example has shown) and may be detrimental to the fishery for a long time period.

Managing for maximum yield in a fishery is a risky enterprise as indicated by the many examples of collapsed marine fisheries. It is often not possible to know the maximum sustainable yield until it has been exceeded. This is one of the reasons Hilborn and Walters (1992) consider managing for maximum yield to be a prescription for disaster. This is likely also true if we exploit a population (in this case alewife) through stocking predators. We do not envy the managers that have to weigh the possibility of increased salmonine returns against the possibility of increased risk for the chinook fishery in Lake Ontario. Maximizing yield and minimizing risk are not compatible management options.

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Spatially-Explicit Models of Fish Growth Rate: Tools for Assessing Habitat Quality

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INTRODUCTION

A key concern of Great Lakes fisheries researchers is the estimation of trophic supply (How much prey is there?) and predator demand (How much prey is needed by the predator?). The management history of the fisheries resources in the lakes has centered on the problems that arise when these variables fall into disequalibrium. For example, the introduction of Pacific salmonids into Lake Michigan in the mid-1960's was an attempt to control alewife (*Alosa pseudoharengus*) populations and restore predator-prey balance in a system that lacked a top predator. Today, after thirty years of intense stocking, predator demand in the same system threatens to exceed prey supply and cause disequalibrium once again. Clearly, effective fisheries management in the Great Lakes depends on defining the relationship between trophic supply and predator demand.

One method that can be used to estimate trophic supply is underwater acoustics. Acoustic systems transmit short (3-5 s^{-1}) bursts of directed, high-frequency (70 kHz - 420 kHz) sound into the water using a beam-forming transducer. The sound propagates in the water as a pressure wave; any object whose density differs from that of the water, fish for example, reflects a fraction of the pressure wave back to the transducer. At the transducer, the reflected pressure wave, or 'echo,' is converted to a voltage which is then stored using an analogue recording device (such as a DAT recorder or VCR) or a computer. With the aid of acoustic data processing software, the recorded voltages can be converted to fish density.

Predator demand can be estimated with bioenergetic models. These models use a mass balance approach to trace the energy in consumed food through the pathways of growth, respiration and waste export. Coupled with information on size-specific growth and mortality rates, the expected consumptions of individual predators or entire populations can be calculated over a variety of time scales. This modeling approach has been used to estimate annual consumption of zooplankton biomass in Lakes Michigan and Ontario (Hewett and Stewart 1989; Rand et al. 1995) and prey fish consumption by Lake Michigan salmonids (Stewart and Ibarra 1991; Brandt et al. 1991). If estimated values of consumption are provided, the bioenergetics model can also predict fish growth rates.

However, estimates of prey supply and predator demand alone are insufficient to determine if current prey stocks actually satisfy predator needs. Consider the situation in Figure 1, Plate A. This figure depicts a generalized

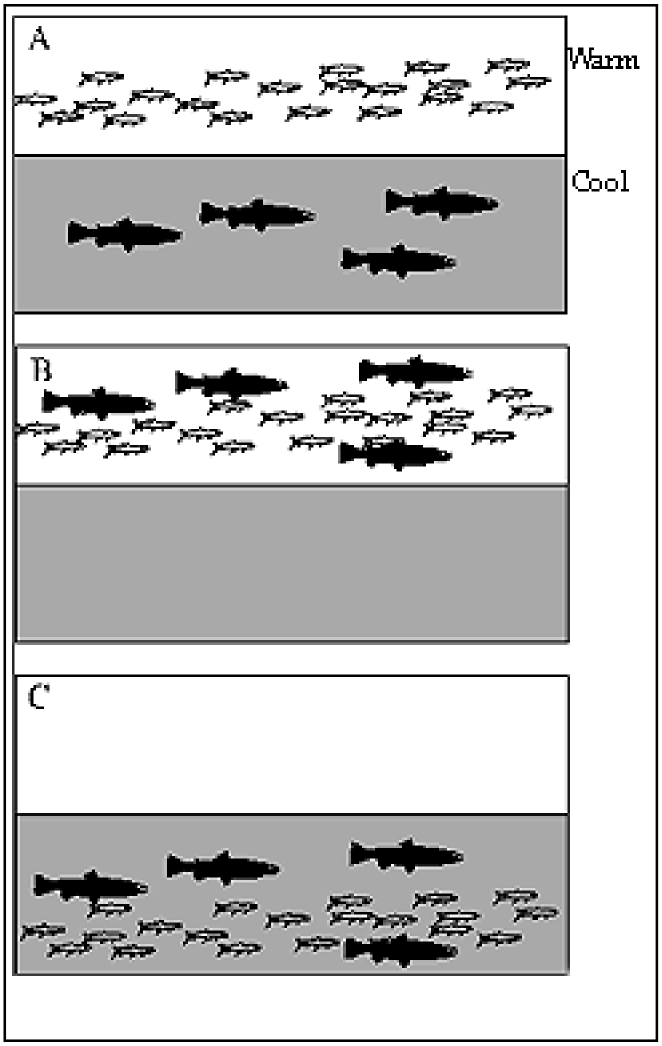

Figure 1. Plates A-C show an environment with two thermal habitats (warm and cool) and different distributions of predators and prey. Plate C is the optimal state for the cool-water predator.

cool water predator in its preferred habitat and a prey population inhabiting warm surface waters. Under these circumstances, the predator population cannot exploit the available food resources because the predator and prey populations are spatially segregated. Thus, managers and researchers must know if predator and prey distributions overlap or, in other words, if prey populations are accessible to the predator.

Now consider Figure 1, Plates B and C. Figure 1, Plate B shows the cool-water predator foraging on the prey population in the warm surface waters. Figure 1, Plate C depicts the same event in cooler water, the preferred habitat of the predator. In these figures, temperature is the critical factor. Rates of consumption (i.e., predator demand), as well as growth and other metabolic processes, vary with temperature; therefore, estimates of predator demand will not match actual consumption if thermal conditions are unaccounted for in the calculation.

Predator growth provides the means to relate prey supply and predator demand. If prey are sufficiently abundant *and* accessible, and if the temperature conditions are favorable, the predator will experience positive growth. If not, then zero or negative growth (i.e., weight loss) ensues. However, in most aquatic systems, prey distributions are non-uniformly distributed in space and time, and water temperatures can vary significantly between the surface and the bottom. Thus, the resulting environment is patchy and non-uniform with regards to growth conditions for the predator. Lasker's (1978) work demonstrated the importance of patchy conditions to foraging predators when he showed that the growth and survival of larval anchovy (*Engraulis mordax*) in the California Current was correlated with the existance of patches of optimally-sized food particles. More generally, Lasker's research demonstrated that important biological and ecological processes can depend on growth conditions at a scale not represented by system-wide averages of prey density or temperature (Brandt et al. 1992).

Spatially-explicit modeling (Brandt et al. 1992, Brandt and Kirsch 1993; Goyke and Brandt 1993; Luo and Brandt 1993; Mason et al. 1995) is one way to assess the effects of complex or patchy environments on fish growth. This new modeling approach combines bioenergetics models with the spatial information inherent in underwater acoustic data. Acoustic surveying is an automated technique for recording the position of fish in the water column and allows the spatial details of prey and predator distributions to be incorporated into the model. The resulting spatially-explicit bioenergetics model describes the potential growth rate of a specific predator along the acoustic transect.

MODEL FRAMEWORK

Conceptual Overview

In a spatially-explicit model of fish growth rate, space is modeled as an explicit attribute of the environment by subdividing the water column into a series of rows and columns that define a grid (Fig. 2, Brandt and Kirsch 1993; Goyke and Brandt 1993). Each cell in the grid is characterized by a specific temperature, a prey density, and a prey size distribution. A foraging model converts the size-specific prey densities in each cell to a predator consumption level. A species-specific bioenergetics model then

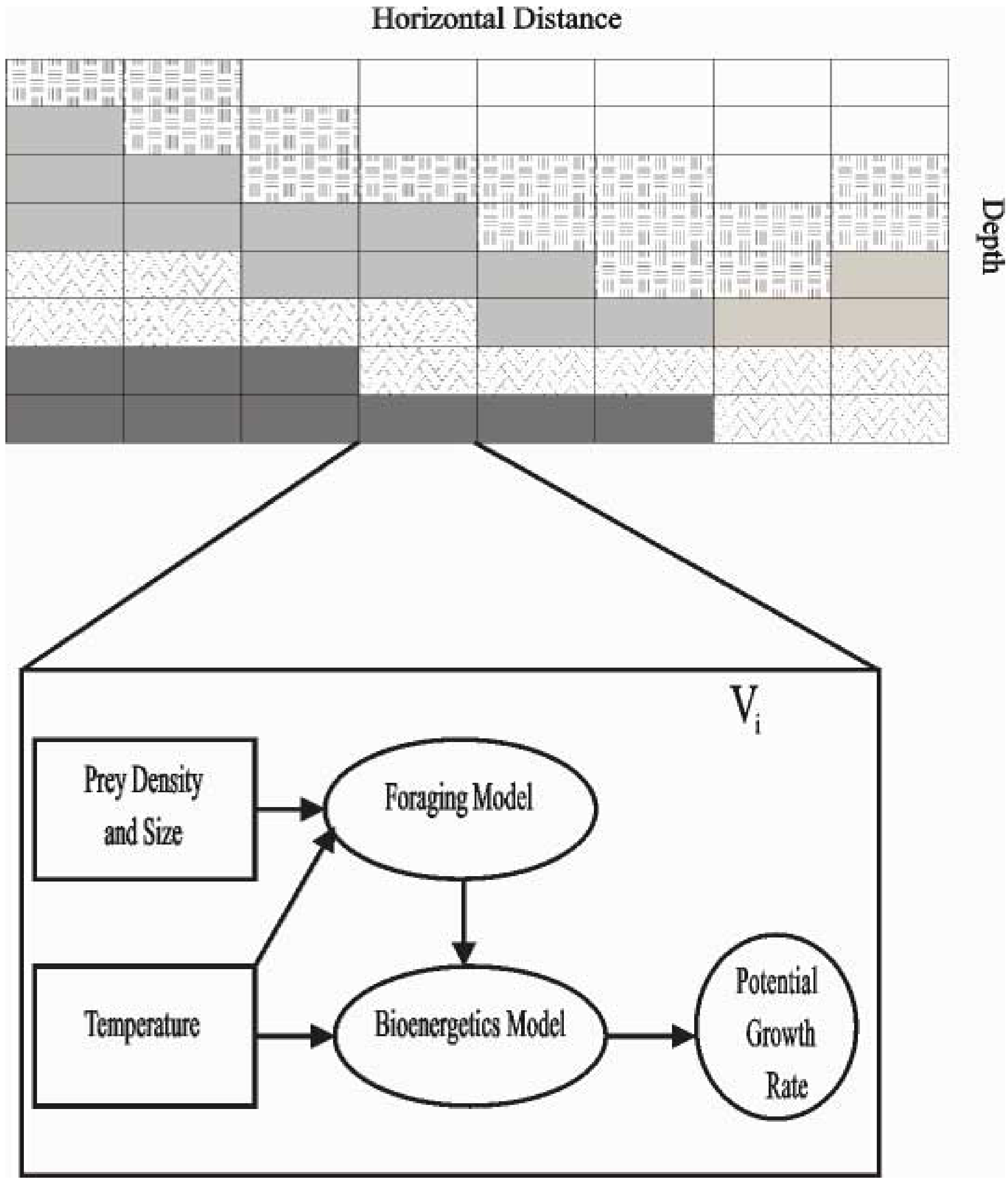

Figure 2. The spatial modeling framework. The shading of the cells depicts microhabitats with different temperatures and prey densities.

calculates a potential growth rate (in g g^{-1} d^{-1}) from the predicted consumption. The potential growth rate is the growth rate a predator might achieve if placed in a given cell and describes the habitat quality of that cell for the predator.

Underwater Acoustics

Underwater acoustics technology is the backbone of the model framework and provides high-resolution, spatial maps of prey densities and distributions through continuous sampling of the entire water column (Clay and Medwin1977; Brandt et al. 1991; MacLennan and Simmonds 1992). Fish densities and locations can be determined from the reflected waves, or echoes, using multibeam transducers (Burczynksi and Johnson 1986; Foote et al. 1986), multiple-frequency systems (Holliday et al. 1989), or statistical analyses (e.g., Clay 1983). Also, the size (length, biomass) of the target can be obtained from the strength of the echo. The actual cells of the grid are created by horizontal and vertical pooling of the acoustic data. Cell size determines the resolution of the model and is governed by boat speed, water depth, acoustic hardware settings, and acoustic data processing software. Data from CTD casts is used to assign a temperature to each cell.

Within a cell, prey fish are assumed to be randomly distributed. The consumption of prey by the predator in each cell is calculated from an equation based on the encounter rate between predator and prey and the probability that the predator recognizes, attacks, captures and ingests the prey item. The encounter rate is defined as a function of the prey density in the cell, predator reactive distance, and the swimming speeds of the predator and prey (Gerritsen and Strickler 1977).

Bioenergetics Model

After the consumption by a predator has been calculated, a species-specific bioenergetics model is run in each cell. The bioenergetics model traces the energy of the consumed food through the various metabolic pathways of an individual fish and calculates potential growth rate according to the following equation:

$$G_i = C_i - (R_i + F_i + U_i)$$

where G_i is the potential growth rate (PGR) of a predator in specific volume of water, i, C_i is the potential consumption of the predator in cell i, R_i is the metabolic losses, F_i is egestion and U_i is excretion. The model is also size-specific; therefore, increasing or decreasing the size of the desired predator will change estimates of PGR. The model format follows that of the "Wisconsin" bioenergetics model (Kitchell et al. 1977; Hewett and Johnson 1987, 1992).

Figure 3 shows the final output of a spatially-explicit bioenergetics model. The shading of a cell corresponds to the growth rate that a specific predator might achieve if placed in that cell and allowed to feed. Since fish growth rates are extremely sensitive to alterations in water temperature and food supply (Bartell et al. 1986), Figure 3 also demonstrates the spatial variability of habitat quality available in the environment.

EXAMPLES

Goyke and Brandt (1993) used a spatially-explicit model to explore chinook salmon (*Oncorhychus tshawytscha*) and lake trout (*Salvelinus namaycush*) growth rates in Lake Ontario along a mid-lake transect (Oak Orchard, NY - Cobourg, Ontario) during the spring, summer and fall. The PGR of each cell was recorded in order to calculate the mean PGR for the transect and the percentage of cells characterized by positive or negative growth rates. Also, maps of PGR's were constructed for the salmonines and displayed as in Figure 3. Their results indicated that the average PGR for these species in the spring and summer was negative because the overlap of sufficient prey biomass with optimal water temperatures was minimal. The data also showed that although only a small fraction of the transect volume supported positive growth (3.2-26.3 %), the overlap of predators with regions of positive growth was high (49.6-71.7 % of all predators). These results suggest that salmonine growth in Lake Ontario, like the growth and survival of Lasker's (1978) anchovies, is tied to patches of favorable growth conditions—not system-wide averages of prey density and temperature used by most growth models.

Goyke and Brandt (1993) also found that lake trout PGR's were much lower than those for chinook salmon in Lake Ontario. Acoustic measures of prey distributions demonstrated that the hypolimnion, the traditional habitat of the lake trout, was largely devoid of prey fish. These observations imply that sufficient prey densities may not occupy the preferred temperature range of adult lake trout. The model results suggested that the reintroduction of a native deep-water coregonid (e.g., bloater) might be useful for enhancing food availability to lake trout populations.

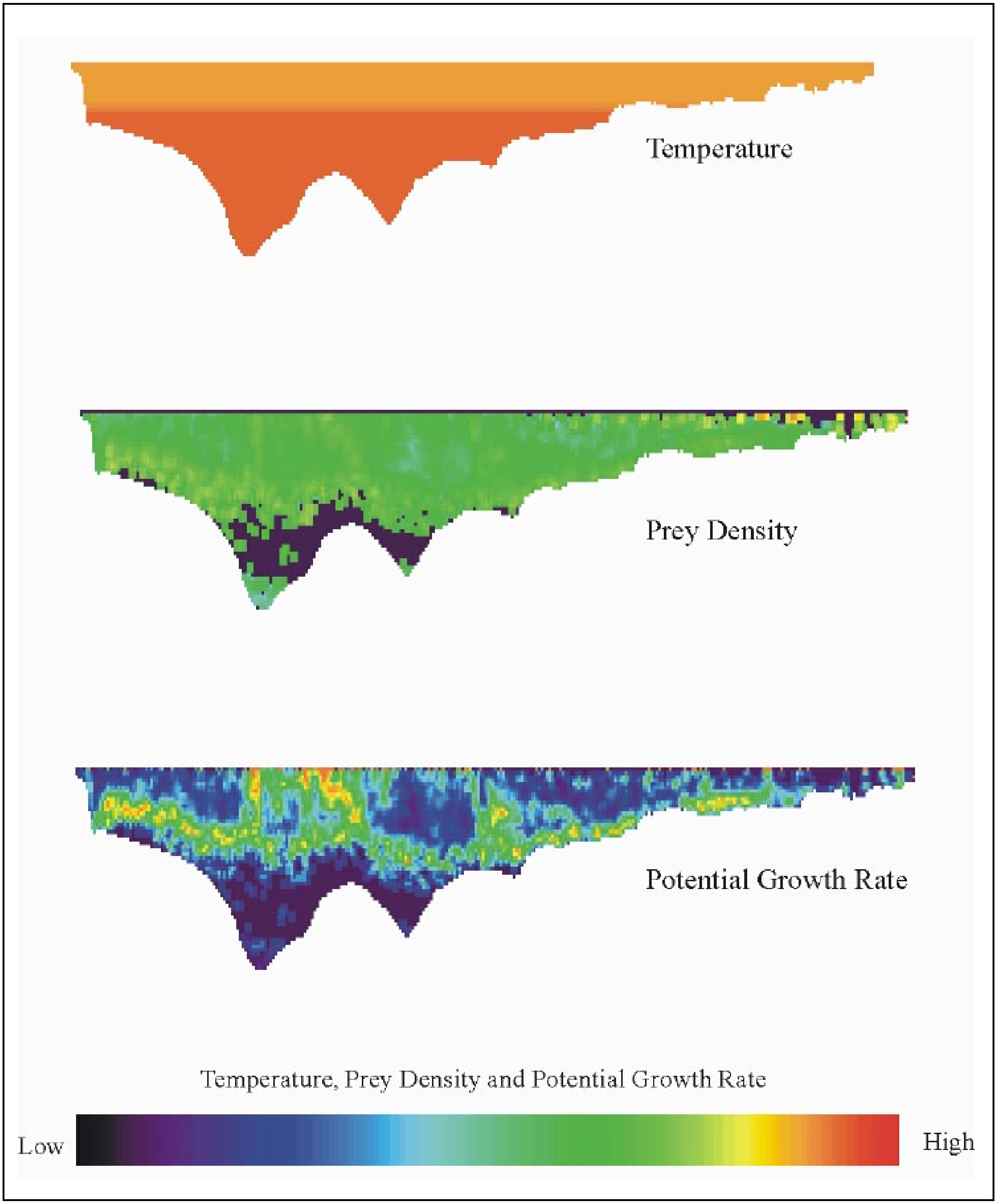

Figure 3. An example of spatially-explicit modeling from Chesapeake Bay. The bottom panel is a map of potential growth rates (g g^{-1} d^{-1}) for a 1.9 kg striped bass (Morone saxatilis).

Mason et al. (1995) used a spatially-explicit bioenergetics model to compare the habitat quality of Lakes Michigan and Ontario for salmonines. Lakes Michigan and Ontario have similar predator assemblages (primarily chinook salmon and lake trout), but different prey bases. In lake Michigan, bloater (*Coregonus hoyi*) dominates the prey fish community, and alewife (*Alosa pseudoharengus*) are the second most numerous species. In Lake Ontario, alewife are numerically dominant with rainbow smelt the next most numerous species. Despite differences in the prey fish communities, the model results indicated that habitat quality (as determined by mean PGR) for salmonines in the two lakes were similar. However, the percentage of the transect volume that supported positive salmonine growth in Lake Michigan (26% for chinook salmon, 21% for lake trout) generally exceeded that provided by Lake Ontario (19% for both species). For chinook salmon, this difference resulted from variability in the spatial distributions of the principle prey species. In Lake Ontario, most of the prey fish biomass (predominately alewife) was concentrated in water warmer than 20°C, a temperature range that does not support high chinook salmon growth rates. Conversely, most of the prey fish biomass in Lake Michigan (mostly bloater and alewife) was located in water water less than 20°C, a temperature range that supports higher chinook salmon growth rates.

APPLICATIONS

The spatially-explicit modeling approach can also be used to assess habitat quality for species that do not currently inhabit the ecosystem. Any potential predator can be modeled if a bioenergetics model exists for that species, and if an acoustic template of prey distributions can be provided. A method of parameterizing each cell with a temperature is also needed. This application of the spatially-explicit modeling approach will allow Great Lakes researchers and managers to evaluate the potential success of purposeful species introductions and the potential impacts of invader species.

Another application of spatially-explicit modeling is the assessment of competition between species. Maps of PGR can be constructed for potential competitors and the percentage of overlap between areas of positive growth can then be compared. A significant amount of overlap would indicate the potential for intense competition between the species and highlight areas where resource partitioning is likely ocurring. A map of PGR for a species absent from the system might also be compared with acoustically-derived maps of fish distributions in the system. Again, a high degree of overlap would indicate the potential for competitive interactions.

The spatially-explicit modeling approach also allows managers and researchers to examine the effects of changing environments (both physically and biologically) on trophic supply and predator demand. For example, there are five to six different species of salmonids stocked into New York waters of Lake Ontario each year (Eckert et al. 1994), and each places its own unique demand on the prey resources of the lake. The proportion of each species stocked is not constant; therefore,the total predator demand on the system changes as the proportion of each species stocked changes. If the distributions of each species in the predator assemblage can be differentiated, the spatially-explicit modeling approach can be used to relate prey supply to each predator's need and the total predator demand. Likewise, the model can also be used to evaluate the effects of shifts in prey fish composition on the predator-prey balance. In the early 1980's, Lake Michigan's dominant prey fish changed from alewife, an epilimnetic planktivore, to bloater (*Coregonus hoyi*), a hypolimnetic benthivore (Jude and Tesar 1985). The spatially-explicit bioenergetics model provides a mechanism (i.e., acoustic survey) to account for the altered distribution of the primary prey species and the ensuing effects on prey accessibility (see Mason et al. 1995) — other growth models do not. The spatially-explicit modeling approach can also relate changes in physical variables, such as temperature, to predator demand because the bioenergetic model is run using the temperature of a particular cell, not a system-wide average of temperature.

CURRENT RESEARCH

One example of current research is centered on the development of spatially-explicit growth models for the trophic level beneath piscivorous predators: the planktivorous fish. Whereas acoustics data provided the spatial information on prey densities and distributions for previous models, optical plankton counters (OPC's) are capable of providing similar information on zooplankton. Thus, models can be constructed for planktivorous fish foraging in a prey field composed of zooplankton. Currently, a spatially-explicit model of alewife growth is being developed for Lake Ontario. The completed model will be used to assess the habitat quality of Lake Ontario for alewife growth and to evaluate the effects of foraging mode, zooplankton abundance and alewife distributions on model predictions.

A second area of current research centers on the temporal variability of potential growth rates. Data for this research were generated by collecting acoustic data along a set of transects arranged in a grid (Figure 4). Each individual transect is 1 nautical mile long and is resampled every 6 hrs. Data collection is continuous and the entire grid is sampled 5 times in 30 hours. A main goal of this effort is to determine how maps of PGR change over time scales of a single day or less and, more generally, to understand the time frame over which a PGR map may be considered representative.

A third area of current research is focused on the development of spatially-explicit models of walleye (*Stizostedion vitreum vitreum*) growth rates in Lake Erie. The west, central, and eastern basins of Lake Erie are distinctive in terms of depth, thermal structure, and fish assemblages. Thus, walleye growth rates in each basin will likely differ. The spatially-explicit model of walleye PGR's will be used to test this hypothesis and to predict potential migration patterns between the basins. The model will also be used to evaluate the impacts of the zebra mussel (*Dresseina polymorpha*) invasion, such as increased water clarity, on walleye growth.

CONCLUSION

In this article we emphasized that the spatial variability in temperature and prey distributions creates a patchy, non-uniform environment with regards to habitat quality for the predator. We have also stressed that system-wide averages of temperature and prey density may be meaningless for describing ecosystem function because predator growth and predator-prey balance can be greatly affected by environmental conditions at the level of the patch. Finally, we have shown that spatially-explicit bioenergetics models can account for the effects of environmental heterogeneity on estimates of habitat quality and are useful tools for evaluating the relationship of trophic supply and predator demand.

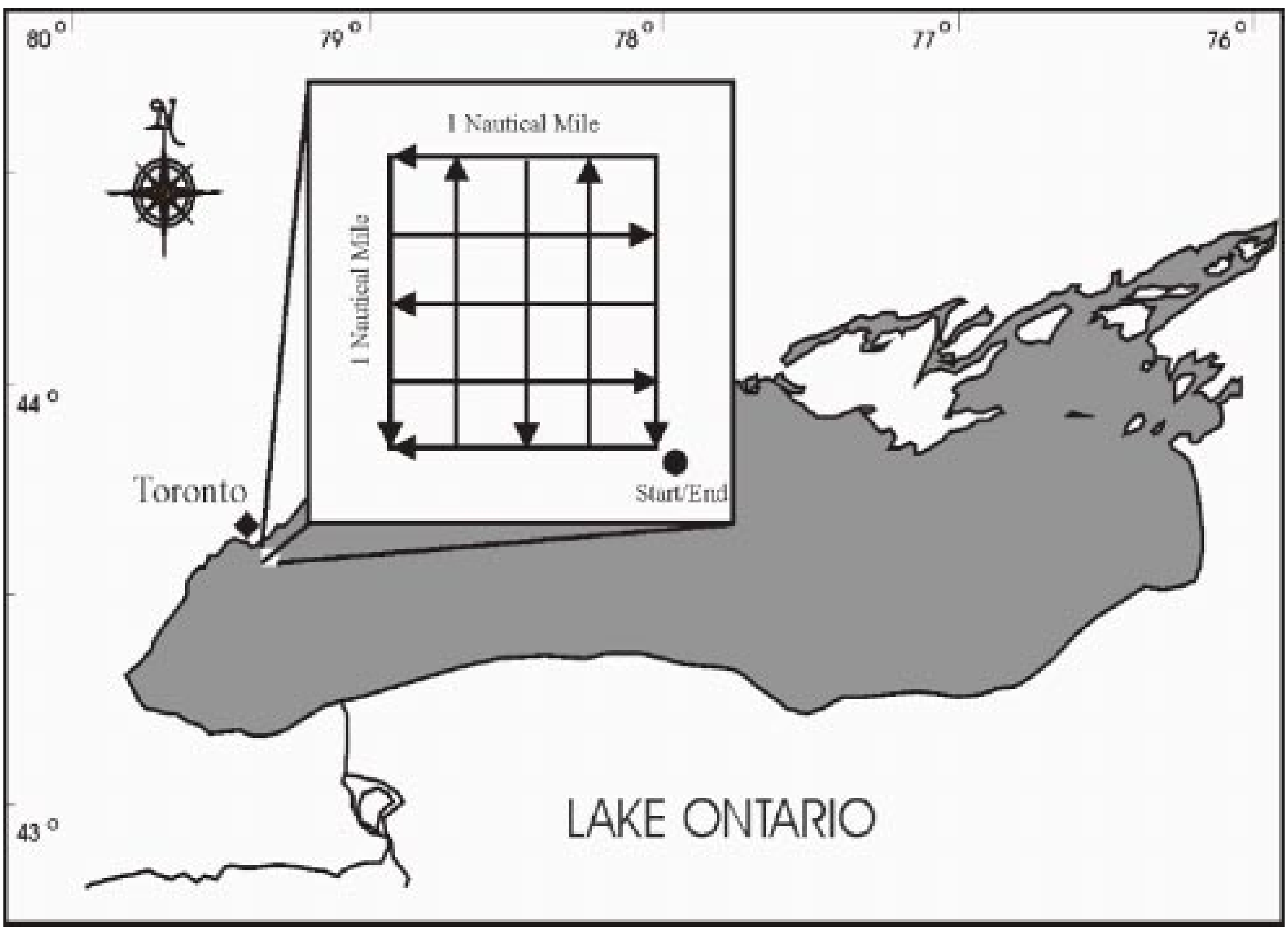

Figure 4. The grid site. The arrows show how each transect is sampled. Data collection begins and ends in the SE corner.

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Resurgence of lake whitefish, *Coregonus clupeaformis*, in Lake Ontario in the 1980s

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ABSTRACT

Lake whitefish (*Coregonus clupeaformis*) is Lake Ontario's most important commercial species. The population was low in 1900, increased substantially in the early 1920s, then declined steadily up to the 1930s. Only a remnant population existed from the 1960s to the 1970s. A major resurgence commenced in the late 1970s and early 1980s, producing a population at least twice as great as at any time since 1900. Eastern Lake Ontario has two major spawning stocks—"lake" (south shore, Prince Edward County) and "bay" (Bay of Quinte). Quantitative scale characteristics have been developed that discriminate these stocks 90% of the time.

The present resurgence began in 1977 and progressed more quickly in the lake (approx. 2 years). The lake stock contributed to the resurgence of the bay stock and in the late 1980s expanded to the west end of Lake Ontario, re-establishing another spawning stock. This resurgence resulted from increased recruitment because of more favourable severe winter conditions (1976-77 and 1977-78) and from increased fry survival associated with winterkill of predators (i.e., alewife and white perch) and with increased lake trout predation on predators (i.e., rainbow smelt). This resurgence signals the re-establishment of a more diverse, self-sustaining cold-water fish community in Lake Ontario.

INTRODUCTION

Lake whitefish (*Coregonus clupeaformis*) is an important species in the cold-water fish community of eastern Lake Ontario. Because of the economic value of this commercial species, early studies were conducted on life history and growth (Hart 1930, 1931). Although some important studies on recruitment have been conducted (Christie 1963), the status of the population has not been published since the early 1970s (Christie 1973, 1974). Early studies have always assumed that the population in eastern Lake Ontario was composed of two stocks—a "lakeshore", or "lake", stock that spawned on the south shore of Prince Edward County and a "bay" stock that spawned in the Bay of Quinte.

During the 1960s and 1970s, only a remnant population of lake whitefish existed in Lake Ontario. During the 1980s, however, both stocks showed a major resurgence. We examine this resurgence in order to understand the factors associated with the species' remarkable resilience. We will review the historic status of the species and specifically (1) examine indices of abundance, based on commercial and research gillnetting, as well as a young-of-the-year bottom trawling; (2) examine year-class strength of spawning fish and consider factors associated with the resurgence; (3) present techniques for separating the stocks and reviewing their status; and (4) consider future prospects for the species.

Historical Perspective

For many years, lake whitefish was the most important commercial species in Lake Ontario (Christie 1973). At the turn of the century, whitefish stocks were at critically low levels (Baldwin et al. 1979). The population increased substantially, however, peaking in the early 1920s, followed by a steady decline until the 1930s. From the 1930s to the 1960s, stock density was well below carrying capacity (Christie 1973). Alternate-year fry stocking was conducted from the mid-1940s to the mid-1950s in an attempt to supplement recruitment. This experimental management was, however, generally unsuccessful (Lapworth 1956; Christie 1963). Although there was evidence that the whitefish stock oscillated in relation to weather conditions, studies in the 1950s and 1960s indicated that exploitation stress induced by commercial fishing was strong enough to, on its own, explain a decline in the species (Christie 1968).

In the early 1960s, the "bay" stock collapsed, and more extensive regulations were applied to the commercial fishery. The stock did not recover immediately, however, because predation by smelt and white perch was strong enough to keep the stock depressed (Christie 1973). In addition, cultural eutrophication on the Bay of Quinte may have acted to negatively affect productivity of the "bay" stock.

Commercial Importance

Commercial harvest of the species in Lake Ontario peaked in the early 1920s (Christie 1963; Baldwin et al. 1979). From 1930 to the early 1960s, harvest was sustained at approximately 420,000 lb annually, although near the end of that period, harvest was probably sustained only by technological advances and gear. In 1950, nylon gillnets came into general commercial use. Commercial harvest plummeted to record-low levels in the 1970s. Regulatory action was taken to protect the species. In the early 1970s, the stock was protected at spawning time, and harvest quotas were implemented in 1980.

Modern commercial fisheries regulations were initiated in 1984. The principal feature of the program was to move toward management on a stock-specific basis. In that year, a harvest quota of 100,000 lb was established for the "lake" whitefish spawning stock, which was showing signs of recovery. A small quota of 2,300 lb was set for the "bay" stock, mainly to assist managers in stock assessment. In addition to quotas, harvest was restricted by season and gear. Commercial licences were bought out in the mid-1980s to reduce commercial effort. Since 1984, stock-specific harvest quotas have been increased as the two stocks recovered. Season and gear restrictions have been adjusted to minimize incidental catch, mainly around large, mature lake trout, as well as walleye.

Since the recovery of the lake whitefish population in the 1980s and 1990s, the species has once again become the most important in the Lake Ontario commercial fishery. Since 1990, the lake whitefish harvest has exceeded that of all other commercial species. In 1994, the harvest was over 450,000 lb; 75% of the 600,000-lb quota was taken at a value of over $400,000. At the present time, the lake whitefish quota and harvest are composed of about 2/3 "lake" spawning stock and 1/3 "bay" spawning stock.

INDICES OF ABUNDANCE

Although it is difficult to find unbiased long-term indices of abundance, there are several measures of whitefish abundance in Lake Ontario that could be used to detect change over both the long- and short-term periods. The commercial catch provides a very long-term

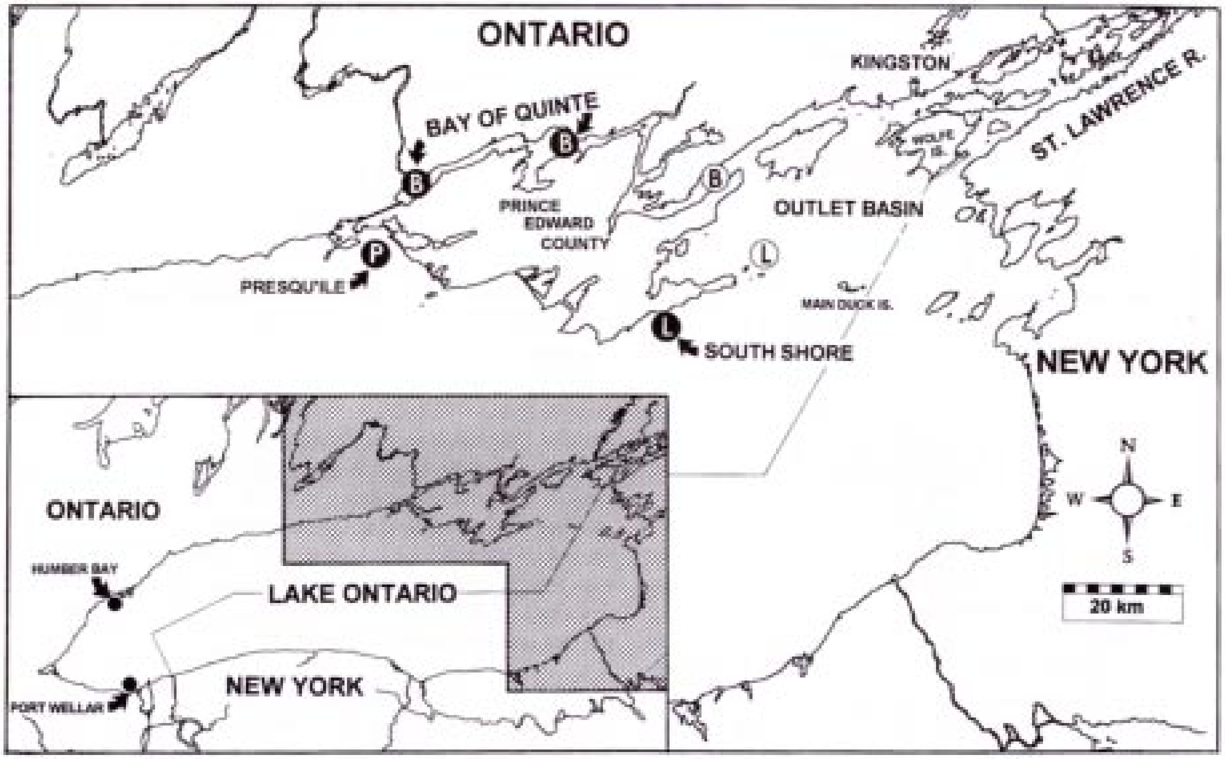

Figure 1. Lake Ontario, illustrating the principal sites that are important in the resurgence of the lake whitefish population in the 1980s. Samples of mature lake whitefish were taken at spawning time from the "lake" stock (dark circle—L) on the south shore of Prince Edward County in eastern Lake Ontario and from the "bay" stock (dark circle—B) in the upper Bay of Quinte. Index gillnetting was conducted in the Canadian waters of the outlet basin. Trawling sites in the two nurseries are indicated (lake—light circle—L; bay—light circle—B). Other samples of lake whitefish have been collected in recent years off Presqu'Ile (dark circle—P) and at Humber Bay and Port Wellar in the west end of Lake Ontario. Trawling sites are also marked.

measure; however, for the past three decades, regulatory changes and quota setting have affected its utility as an unbiased index of abundance. On the other hand, since the early 1960s, standard index netting (originated by Research and more recently amalgamated into the Assessment program) has been conducted annually during the summer period in the outlet basin of Lake Ontario (Fig. 1). This program utilizes stations and gear that once were used routinely by the commercial whitefish fishery. By comparing these two indices during the 1960s, it is possible to adjust the commercial catch so that it provides an index that is comparable to the Research gillnet index; a combination of the two provides a long-term index of relative abundance from the early 1900s to the present.

In the 1970s, Research established standard bottom trawling stations in the outlet basin and in the Bay of Quinte. At two of these stations (Fig. 1) (in the vicinity of Timber Island—43°57′80"N, 76°48′10"W —and Conway—44°05′50"N, 76°54′20"W) in late summer and early autumn (August to October) over the years, young-of-the-year lake whitefish have been caught consistently enough to provide a useful index of abundance of young lake whitefish. These two sites appear to be the late summer-autumn nurseries of young whitefish of the two stocks. The trawl catches are valuable indices because they provide a direct measure of year-class strength.

Commercial-Research Index

The commercial-Research gillnet index indicates that lake whitefish in eastern Lake Ontario are at record-high levels. Although large numbers of lake whitefish were taken commercially in the early 1920s, in the early 1990s, whitefish were two to three times more abundant (Fig. 2). This resurgence commenced in the late 1970s and early 1980s and increased at a time when the population was at record-low levels. This suggests that survival in the 1970s must have been extremely high to allow such a rapid re-establishment of the population. The index is not extended to the present (Fig. 2) because a major rational-

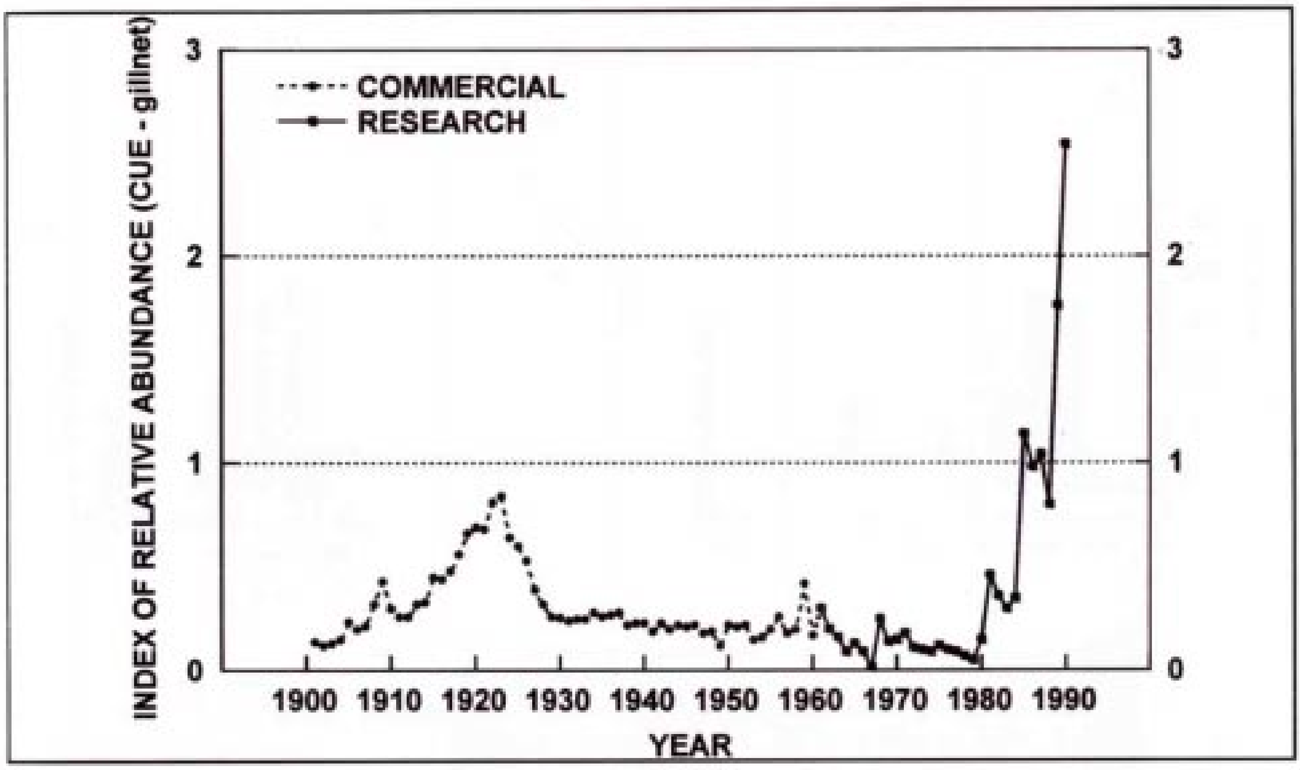

Figure 2. Index of relative abundance of lake whitefish from 1900 to 1991, Eastern Basin and Bay of Quinte, Lake Ontario. Index for the past three decades obtained from catch of lake whitefish abundance (numbers per 100 m^2 of experimental gill net) in Research index gillnets (Casselman and Scott 1992). Index prior to that time is of commercial catch (Baldwin et al. 1979) scaled to Research catches to provide a comparable index extending from the turn of the century.

ization of the indexing program in 1992 resulted in some changes that require converting data; specifically, multifilament gillnets were replaced with monofilament nets and the number of indexing stations in the outlet basin was reduced from six to two. Quantitative integration of this trend-through-time data series is currently under way. Nevertheless, whitefish abundance remained high and stable from 1991 to 1995, and it appears that whitefish numbers may have reached a state of temporary equilibrium.

Young-of-the-Year Index

In the early 1970s, when bottom trawling was first conducted in the outlet basin in the lee of the south shore of Prince Edward County (Timber Island) and in the mouth of the Bay of Quinte (Conway) (Fig. 1), few, if any, young-of-the-year whitefish were observed (Fig. 3). Prior to 1981, only two young-of-the-year lake whitefish were caught at the “bay” site, one each from the 1973 and 1974 year classes. At the “lake” site, small catches (Fig. 3) of the 1977 to 1979 year classes were taken. Small and moderate year classes of bay and lake stocks were observed in 1981. Since 1981, young-of-the-year lake whitefish catches in the bay nursery indicate that moderately strong year classes were probably produced annually with the possible exception of 1985. In the lake nursery, however, no strong year classes were observed between 1981 and 1986. Since then, the stocks have produced a strong year class in either 1986 or 1987. Subsequently, year-class production has been stronger, more so for the bay stock, but rather sporadic, with both stocks producing a relatively strong year class in 1994 (Fig. 3).

YEAR-CLASS STRENGTH AND FACTORS AFFECTING RESURGENCE

In recent years, the commercial catch has been sampled at the peak of the spawning period, providing a measure of relative abundance of the contributing year classes. We present here relative year-class strength of three samples of mature whitefish taken on spawning grounds in the “lake” and in the “bay” (Fig. 1). For com-

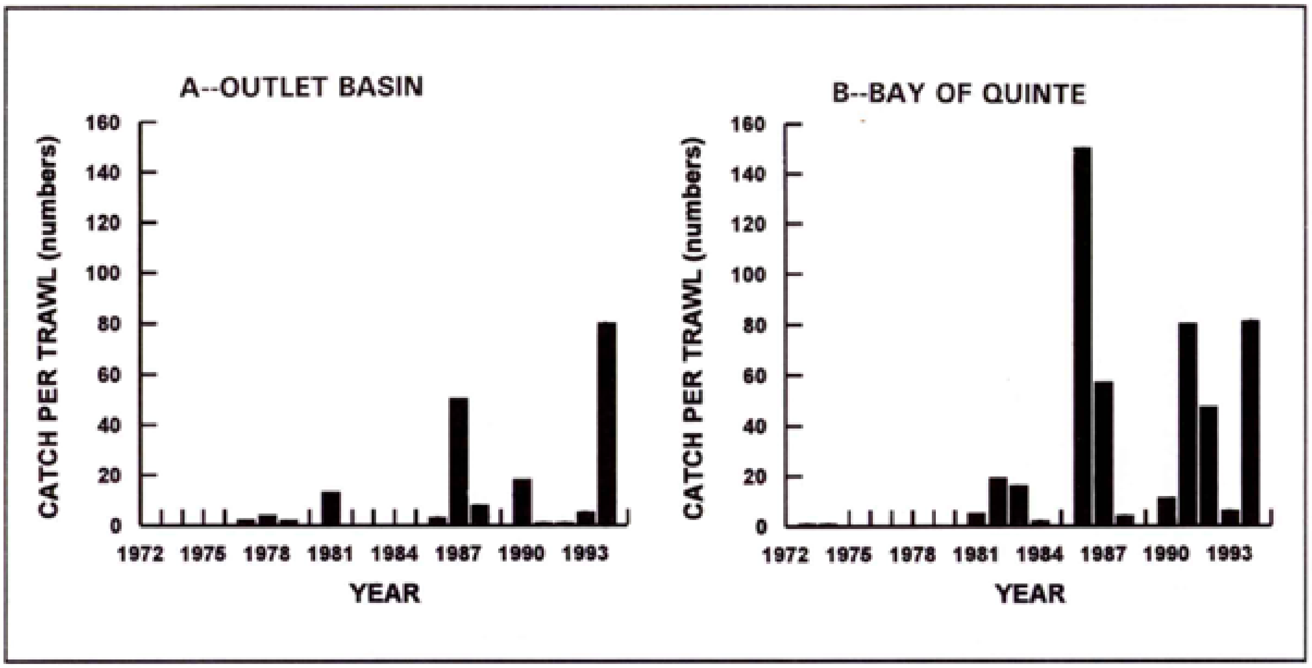

Figure 3. Catch of young-of-the-year lake whitefish in standard trawl drags conducted in late summer and early autumn in A) the outlet basin, Timber Island, and B) the Bay of Quinte, Conway. Catch is adjusted to number of lake whitefish caught in a 12-minute haul. Vessel was refitted in 1989; trawling was not conducted.

parative purposes, we use the first of the samples that were collected in 1988 and two samples subsequently taken in 1990 and 1992, each containing more than 100 fish (Fig 4). The earliest sample indicates that whitefish resurgence probably began in 1977 in both the lake and the bay. In the lake, this was followed by a gradual resurgence; however, in the bay, a similar resurgence did not occur until 2 years later, commencing in 1979. The 1979 and 1980 year classes were strongest in the lake and could be tracked into 1990 and 1992. Of these early year classes, 1982 was strongest in the bay. This could be tracked in subsequent samples. The earliest sample (1988) contained some very old individuals; both the 1962 and 1967 year classes were present in the lake sample.

Prior to this, Christie (1973) had shown that the 1957 year class was strong in the lake and that the 1962 year class had considerable strength. It appears that these remnant year classes from the 1960s may have been contributed to the recent resurgence. It is rather remarkable to see such old individuals (21 and 25) spawning and contributing to the re-establishment of the stock. Subsequent samples indicate that the 1983 to 1985 year classes were probably relatively weak but that the 1986 and 1987 year classes were strong. This agrees with the trawl index (Fig. 3). These catches also indicate that recruitment has become consistent and less sporadic than in former years.

Several factors probably contributed to this major resurgence, including both favourable weather conditions and reduced predation on early life stages. In Lake Ontario, weather conditions and the abundance of potential predator populations such as alewife are integrally related, making it difficult to factor out their relative importance. It has been shown that cold autumns, early ice cover, and cold, prolonged winters are correlated with strong year classes and increased survival of lake whitefish eggs and fry (Christie 1963; Freeberg et al. 1990); similar results have been shown for lake trout, another fall-spawning, cold-water species (Casselman 1995). The exact mechanism has been speculated to involve protection of the eggs during the incubation period (Christie 1963); however, Casselman (1995) demonstrated that this was related to fewer heat units, slower development, reduced premature hatch, and better synchrony between hatch and emergence in spring. It is more than coincidental that the winters of 1976-77 and 1977-78 were among the most severe in recent times. On the other hand, a massive die-off of alewife, a potential predator on larval whitefish, occurred during the first of these two winters, and the following winter, another potential predator, white perch, underwent a selective winterkill (Christie 1973).

In the late 1970s, extensive stocking of lake trout was begun in Lake Ontario to re-establish the species. This large piscivore substantially reduced alewife and rain-

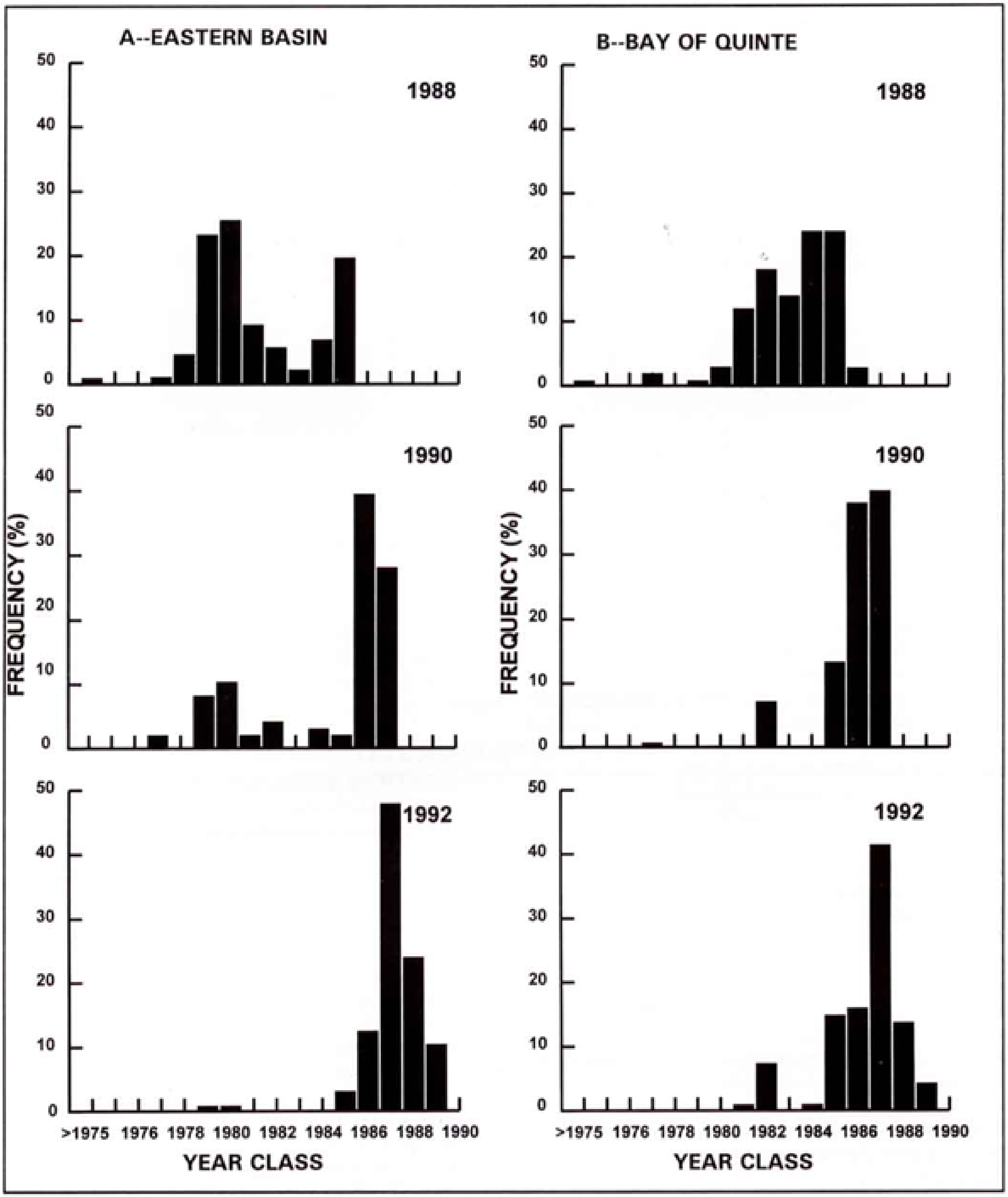

Figure 4. Relative year-class strength (%) of mature lake whitefish sampled in 1988, 1990, and 1992, determined from collections made at spawning time from A) eastern basin, south shore of Prince Edward County, and B) the Bay of Quinte, upper bay. Age was interpreted from acetate replicates of otolith sections. Each sample contained at least 100 fish. Fish belonging to year classes prior to and including 1974 were combined.

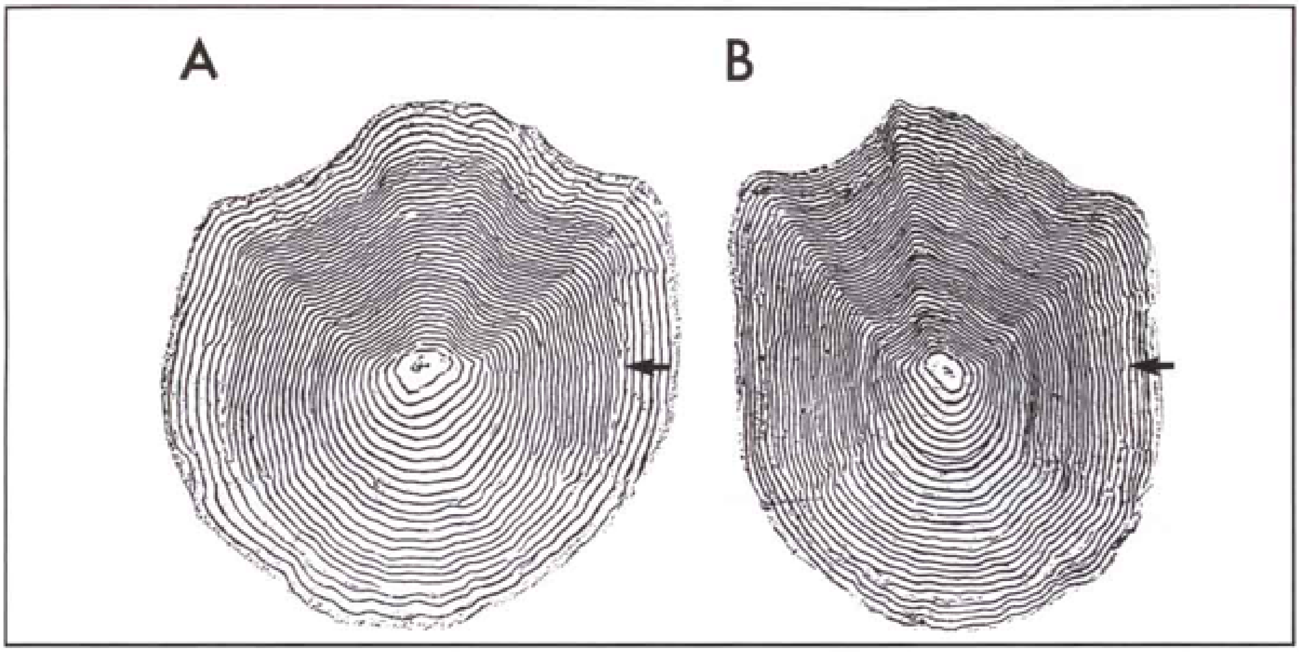

Figure 5. Typical scales from yearling lake whitefish taken by trawls from A) Bay of Quinte, Conway, on June 23, 1988; FL 173 mm; B) outlet basin of Lake Ontario, station 3, Timber Island, July 8, 1988; FL 191 mm. Scales show typical checks and growth characteristics of fish of bay and lake origin. Arrows indicate location of the check associated with the first annulus. Magnification = 20x.

bow smelt abundance (Christie et al. 1987; Casselman and Scott 1992). Lake trout selectively preyed upon large smelt. Smelt are considered to be a substantial predator of larval coregonines (e.g., Christie 1973; Loftus and Hulsman 1986). Casselman and Scott (1992) showed a strong correlation between increase in lake trout abundance, decrease in the number of large smelt, and the major increase in lake whitefish; therefore, predators of larval whitefish were greatly reduced.

Although other factors may play a role in this resurgence, they would be minor compared with the effects of favourable winter conditions that occurred in the late 1970s and the associated dramatic decrease in potential predators and low predator stocks perpetuated by large piscivores such as walleye and stocked lake trout. The coincidental resurgence of walleye in the Bay of Quinte indicates that decreased predation may also have been a major factor in the re-establishment of other species in embayments in eastern Lake Ontario. Whatever the factors, they appear to have contributed coincidentally throughout the Great Lakes, since this resurgence in lake whitefish also occurred at approximately the same time in Lakes Michigan, Huron, and Erie. In Lake Ontario, the lake stock appeared to have expanded prior to the bay stock. Indeed, there is evidence that the lake stock contributed to the re-establishment of the bay stock (Casselman and Brown, unpubl. data). Cultural eutrophication in the bay may have also contributed to suppress the productivity of the bay stock because whitefish resurgence also commenced just as phosphorus reduction began.

STOCK STATUS AND SEPARATION TECHNIQUES

As indicated earlier, there is evidence that the bay and lake stocks have responded differently over the years. Up to 1960, the bay stock was considered to be three times larger than the lake stock (Christie 1973), and in the 1960s, possibly because of cultural eutrophication, the bay stock virtually collapsed. The status of the stock, however, could be evaluated only by assessing relative abundance on the spawning grounds. This was difficult to assess and impossible to quantify because no specific techniques were available to separate these putative stocks of whitefish.

A research study was begun recently to separate stocks of whitefish, based on quantitative attributes in their calcified structures. Scale characteristics associated with the first annulus provided a valid quantitative technique that involved measuring the spacing of the circuli on either side of the check associated with the first annulus (Fig. 5) (Brown and Casselman 1992). The misclassification rate has been determined to be 10% for each stock. Therefore, it is now possible to correctly assign stock origin to "bay and "lake" whitefish 90% of the time. Quite generally, whitefish associated with the bay nursery have narrow-spaced circuli on the scales prior to the winter cessation, and wide-spaced circuli are deposited when growth commences in the spring (Fig. 5A). Such a prominent change is not visible at the first annulus of the scales of young-of-the-year and yearling whitefish living in eastern Lake Ontario (Fig. 5B). Quantitative criteria have been developed to assign origin, and software has been incorporated into CSAGES (Casselman and Scott 1994) that makes discrimination technically straightforward, quantitative, and objective (Brown and Casselman 1992). In addition, other quantitative stock-separation techniques are being developed. Once these can be widely applied, management of the species can be more specific, because it is apparent that production of these two major stocks may not always be in synchrony.

Application of these criteria to a subsample of the 1990 spawning stock indicated that after corrections were made for misclassifications, 12% of the fish from the two samples had developed in alternate nursery habitat, indicating considerable mixing between these two spawning stocks. These calcified structure stock-discrimination techniques are practical and accurate and are especially useful, given that other morphological and biochemical attributes have not been able to separate these two sympatric whitefish stocks of eastern Lake Ontario (Ihssen et al. 1985).

In the late 1980s and early 1990s, lake whitefish increased in abundance on the west side of Prince Edward County near Presqu'Ile (Fig. 1). An examination of the scales of these fish indicated that the majority (79%) originated in the lake nursery associated with the spawning grounds on the south shore of Prince Edward County (Fig. 1) (Brown and Casselman, unpubl. data). It appears that whitefish in this area may have resulted from the expansion of the whitefish population associated with the south shore of Prince Edward County in the mid-1980s. Scale attributes in later life were, however, different; whether they originated in this area or from the south shore spawning grounds has not been determined.

More recently, in 1993, whitefish have been taken in the west end of Lake Ontario. A sample collected in an index program off Humber Bay (Fig. 1) appears to have originated mainly from the 1992 year class, but the 1988, 1990, and 1991 year classes were also present in limited strength. Scales from fish of the 1988 year class appeared to have attributes typical of both the lake nursery associated with the south shore of Prince Edward County and later scale growth similar to samples from Presqu'Ile. But none of the young whitefish had scale characteristics specific to the lake stock originating in the east end of Lake Ontario. Ninety percent of the whitefish had scale attributes associated with the open lake, but none of the other whitefish had scale characteristics typical of the lake stock of Prince Edward County. It is speculated that the 1988 year-class fish may have dispersed from the south shore of Prince Edward County to Humber Bay in the west end of Lake Ontario and established a spawning stock. More recently (1995), whitefish with scale characteristics typical of western Lake Ontario have been taken in commercial fishing near Port Wellar (Fig. 1). These samples support the contention that lake whitefish spread to the west end of Lake Ontario in the late 1980s and early 1990s and have re-established another spawning stock.

FUTURE PROSPECTS

Lake whitefish, which have shown a major resurgence in the east end of Lake Ontario, are composed of two and possibly three spawning stocks, and in the west end of Lake Ontario, resurgence of at least one other major stock is occurring.

Stock separation techniques have been developed that make it possible to use archived scale samples to quantify the dynamics and interaction of whitefish stocks in Lake Ontario over the years. This will provide additional insights and precision into understanding the factors effecting fluctuations in abundance that have been observed. In addition, more specific commercial harvest and quota management can be applied to the various stocks.

Annual variations in weather conditions are considered to have been one of the most important factors regulating whitefish abundance. Nevertheless, the population is now large enough that, barring catastrophic change, adequate reproductive potential exists to maintain the population at present high levels. Commercial fishing, which was considered to have over-exploited whitefish in the 1950s and 1960s, now is considered to exert only a moderate effect on the population (exploitation rate of 5-10% of the bay stock). Lamprey wounding, which in the 1950s caused substantial mortality (Christie 1973), is now virtually nonexistent, hence is not important in regulating abundance. Decreased phosphorus loading and trends away from eutrophy and toward mesotrophy and oligotrophy mean that cultural eutrophication is not the limiting factor that it once was for the bay stock. In addition, predator species, such as smelt, alewife, and white perch, are currently at much reduced levels compared with former times and probably no longer are significant predators on larval whitefish. It remains to be determined whether zebra mussel colonization of the spawning shoals of the lake stock that began in 1992 and in the bay stock in 1993 (Hoyle 1993) will affect lake whitefish recruitment. Zebra mussel abundance has increased most rapidly in the bay and reached a density of 100,000 m^{-2} by 1994, as compared with only 10,000 m^{-2} in the lake spawning area on the south shore of Prince Edward County.

The resurgence of lake whitefish in eastern Lake Ontario and the Bay of Quinte, which began in the late 1970s and extended through the late 1980s, appears to have expanded to the west end of Lake Ontario. The increase in the size of the lake whitefish spawning population and its reproductive potential now results in more stable annual recruitment, which will probably remain strong with increased biomass. This resurgence of the whitefish population appears to signal the beginning of the re-establishment of a more diverse, natural, self-sustaining cold-water fish community in Lake Ontario.

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Vegetative Characteristics of Muskellunge (*Esox masquinongy*) Spawning and Nursery Habitat in the 1000 Islands Section of the St. Lawrence River

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INTRODUCTION

Populations of naturally reproducing muskellunge (*Esox masquinongy* Mitchill) are becoming less common throughout their range due, in large part, to poor reproductive success of the wild population (Dombeck et al 1986). Because of this, maintenance of the fishery often requires stocking. A more desireable alternative would be to restore natural reproduction to a level that would sustain a robust sport fishery. In order for fishery managers to restore self-sustaining populations of muskellunge, however, they must be able to identify appropriate spawning and nursery habitat which is difficult since a coherent picture of the spawning and nursery requirements of muskellunge has yet to be developed.

Craig and Black (1986) reported that muskellunge nursery habitat consisted of a narrow band of dense near shore emergents adjacent to a wider zone of less dense submergent vegetation. Hanson and Margenau (1992) indicated that muskellunge fingerlings preferred shallow areas (<10 ft.) over sand substrates, often with structural components of fallen trees and overhanging branches. Other habitat variables associated with muskellunge have included the presence of submergent vegetation such as pondweed (*Potamogeton* spp.), submergent and emergent species like water lilies (*Nuphar*, *Nymphea*), pickerel weed (*Pontederia*), arrowhead (*Sagittaria*), coontail (*Ceratophyllum* spp) and cattail (*Typha*) (Scott and Crossman 1973). Strand (1986) and Dombeck et al. (1984a) reported that muskellunge spawned over areas where muskgrass (*Chara*) was the dominant vegetation. Other investigators have reported the presence of muskgrass at muskellunge spawning and nursery sites.

Craig and Black (1986) surveyed 18 muskellunge nursery sites in Georgian Bay and concluded that sedges (*Najas flexilis*); muskgrass (*Chara*); and pondweed (*Potamogeton gramineus*) were the vegetation most closely associated with muskellunge nursery sites. Dombeck et al (1984a) also suggested that the lack of adhesive eggs and larvae without a cephalic cement gland cause muskellunge eggs to settle to the bottom which makes them vulnerable to the influence of conditions at the sediment water interface. If the water is eutrophic or has poor circulation then oxygen levels decline leading to mortality of eggs or early larvae.

Our experience in the St. Lawrence River suggests that the relationship between habitat and muskellunge egg, larval and early juvenile stages is a complex interaction between vegetation as a source of mortalilty of the egg and early larval stages and as shelter for the older more mobile individuals.

The objective of this study is to gather information on the location of muskellunge spawning and nursery sites in the 1000 Islands section of the U.S. portion of the St. Lawrence River and to compare the vegetative associations at these sites with sites which were not used by muskellunge for such activity.

Location Of Spawning And Nursery Sites

Potential sites in the study area were sampled throughout the spawning and nursery period at least once during the summers of 1987, 1988, and 1989 (Werner et al. 1990). Sites were sampled for spawning adults using trap and hoop nets. Nursery sites were sampled for young of the year fish with an 18m x 2m x 0.64 cm mesh bag seine. Twenty-one spawning sites and 37 nursery sites were identified out of 74 potential spawning and 88 potential nursery sites sampled (Table 1).

Based on this information, three studies were undertaken to explore the relationship between vegetative characteristics and muskellunge use of a site. Initially, 12 of the above sites were studied in detail. The vegetative composition of each was determined and then related to muskellunge use of the bay as a spawning or nursery site. The conclusions from this extensive study led to two other studies: one a detailed analysis of the vegetation at muskellunge sites focusing on the physical structure of the vegetation to try to understand the distribution of muskellunge among habitat types within a bay; the second study attempted to determine the mechanism by which an observed negative correlation between muskgrass and muskellunge use might be explained.

Potential spawning sites sampled	74.0
No. with musky	21.0
% with musky	28.4
Potential nursery sites sampled	88.0
No. with musky	37.0
% with musky	42.0

Table 1. Results of extensive sampling for muskellunge spawning and nursery sites in the 1000 Islands section of the St. Lawrence River.

Characterization of Spawning and Nursery Habitat

A. Extensive analysis of muskellunge spawning and nursery habitat

Twelve bays were selected to determine if there was any relationship between vegetation and muskellunge use of the bay for reproduction, seven were muskellunge sites and five were non-muskellunge sites. Vegetation was collected along three 50m transects at 10 meter intervals using 0.25m^2 quadrats and SCUBA gear. Individual stems were identified and counted in each quadrat and the depth and percent bottom cover were measured. Cover was estimated visually before vegetation was collected.

Vegetation collected throughout the season consisted of 17 genera belonging to 13 families. Four species occurred in all twelve sites: muskgrass, watermilfoil (*Myriophyllum*), elodea (*Elodea canadensis*), and wild celery (*Vallisneria americana*). None of the other species were as broadly distributed. Comparisons of stem densities

Table 2. Comparison of stem densities for the four most common species of submersed vegetation between sites used by muskellunge as a spawning or nursery site (Musky) and sites not used by muskellunge (Non-musky).

		Stem Density (No. M^{-2})		
Species	**Site Category**	**Mean**	**Std Error**	**P**
Chara sp.	Non-musky	10475.7	1949	0.032
Chara sp.	Musky	4609.6	1862	
Myriophyllum sp.	Non-musky	311.9	1785	0.922
Myriophyllum sp.	Musky	77.5	1603	
Elodea canadensis	Non-musky	598.4	1719	0.855
Elodea canadensis	Musky	159.0	1665	
Vallisneria americana	Non-musky	938.4	1855	0.939
Vallisneria americana	Musky	750.2	1603	

of these four species between muskellunge and non-muskellunge bays indicated that there were no significant differences between *Myriophyllum, Elodea and Vallisneria* densities between bays arranged by category (Table 2). There was a significant difference, however, for muskgrass. This species was at significantly lower densities in muskellunge sites than it was for those without muskellunge ($p = 0.03$) (Table 2).

The extent of cover along each transect was analyzed by calculating percent cover by site category (Table 3). Inspection of the confidence intervals suggests that the major differences occur at 30, 40, and 50 meters off shore in water greater than 1m deep where muskellunge sites have significantly less coverage by vegetation than non-muskellunge sites (Table 3). The inshore sites less than 20m from shore are similar in cover density. This analysis suggests muskellunge spawning and nursery sites have lower densities of muskgrass and lower vegetative cover in the off shore areas than non-muskellunge sites.

B. Microdistribution of Juvenile Muskellunge in Relation to Vegetation in Nursery Areas

Although general descriptions of nursery areas provide a valuable picture of muskellunge habitat requirements, they may not adequately represent specific habitat preferences on a smaller scale relevant to the individual. Habitat characteristics such as availability of cover in the upper portion of the water column may be as important as the species composition of the vegetation itself. Lapan (1985), Wahl and Stein (1989), Parsons (1959), Farrell (1991), and Osterberg (1985) describe muskellunge as being distributed in the upper portion of the water column. These observations imply that emergent and taller submergent species, in close proximity to the water surface, may offer muskellunge better habitat than shorter bottom cover species. Field observations of juvenile muskellunge in the St. Lawrence River have indicated that they are typically found with vegetation near the surface.

This portion of the study sought to identify several important factors associated with nursery habitat and test the hypothesis that muskellunge distribution is dependent on the spatial orientation of definable habitat parameters; such as height of vegetation, percent cover, and proximity of vegetation to the surface, as well as the density of muskgrass.

METHODS

Young of the year muskellunge were captured during the 1991, 1992, and 1993 field seasons using an 18m x 2m x 0.64cm mesh bag seine. Five musky and five non-musky bays were seined twice each August (in 1993, only one seining period was conducted). Both the numbers of hauls and depth were standardized.

The beginning and end of all seine haul paths were marked with fluorescent orange floats for later identification. Seine hauls were identified either as successful or unsuccessful hauls based on whether any muskellunge were netted. Using the point-quadrat method, three quadrat measurements were applied along each seine haul path to characterize the vegetation. The quadrat consisted of a $1m^2$ metal frame divided into four $0.25m^2$ sections.

Table 3. Summary of % vegetative coverage along transects in muskellunge and non-muskellunge sites.

Sites	N	mean	95% CI	shore (m)
Non-Muskellunge	9	6.11	1.39 - 10.82	10
	8	68.75	40.03 - 97.46	20
	9	69.44	39.48 - 99.39	30
	6	97.50	94.86 - 100.0	40
	7	97.85	95.70 - 99.99	50
	8	83.75	57.08 - 100.0	
Muskellunge	17	8.94	0 - 18.63	10
	18	58.77	39.67 - 77.86	20
	18	68.61	51.93 - 85.28	30
	18	72.77	54.94 - 90.59	40
	17	60.58	41.87 - 79.28	50
	18	61.38	43.62 - 79.13	

Variable (year)	Muskellunge	Non-Muskellunge	P
% Cover ('91)	77.1 (7.1)	67.2 (10.4)	0.433
% Cover ('92)	76.5 (6.9)	71.7 (6.3)	0.591
% Cover ('93)	89.1 (8.9)	69.6 (11.9)	0.232
Vegetative Height ('91)	32.8 (4.7)	32.9 (8.9)	0.387
Vegetative Height ('92)	29.8 (6.2)	38.6 (5.1)	0.144
Vegetative Height ('93)	36.7 (7.0)	45.1 (10.1)	0.186
Distance from Surface ('91)	54.6 (9.7)	46.0 (8.6)	0.256
Distance from Surface ('92)	45.5 (7.6)	48.9 (6.4)	0.499
Distance from Surface ('93)	66.5 (8.7)	58.0 (13.3)	0.588

Table 4. Comparison of vegetative parameters between designated muskellunge and non-muskellunge bays in the St. Lawrence River (1991-1993). [Mean (2 standard error).]

Within the quadrat, dominant aquatic plants were identified to genera, percent bottom coverage by vegetation, average plant height, and average distance of aquatic plants to the surface were measured. Water depth for each quadrat was determined by the sum of average height and distance to the surface. When emergent vegetation was encountered, distance to the surface was classified as zero and vegetative height equaled the depth of that particular quadrat.

Habitat parameter preferences by muskellunge fingerlings were determined through the comparison of muskellunge and non-muskellunge bays, and, most importantly, by comparing seine hauls that resulted in the capture of muskellunge to seine hauls that did not capture muskellunge. An analysis of variance with subsampling was employed to analyze all data (Peterson 1985) with alpha = 0.05. For comparisons of muskellunge and non-muskellunge bays, an experimental unit was defined as a bay and the sampling unit as the average of three quadrat measurements for a seine haul. All other comparisons defined seine hauls as the experimental unit and individual quadrats as sampling units. A X^2 analysis tested muskellunge response to muskgrass between muskellunge and non-muskellunge seine hauls. Observed relative abundance of muskgrass within successful and unsuccessful seine hauls were compared with the relative overall abundances found throughout all seine hauls. During this study both categories of bays were stocked with larval muskellunge (ca. 20mm TL), thus insuring that fish would be found in both bay types.

RESULTS

Comparisons by bay

The results suggest that on the less precise whole bay basis non-muskellunge bays possess similar habitat characteristics to muskellunge bays. Statistical analysis did not indicate significant differences in any of the habitat variables measured between muskellunge and non-muskellunge bays for any of the three years (Table 4).

Comparison by seine haul

Results of analysis between successful and unsuccessful seine hauls provides a more precise comparison of habitat preferences by quantifying habitat in the area in which muskellunge are actually found. Vegetative height was significantly greater in successful seine hauls for all three years ($P < 0.05$), averaging 10cm taller for 1991 and 1992 and over 32cm higher in 1993 (Table 5). Vegetation was significantly closer to the surface in 1991 and 1993 ($p = 0.005$ and 0.001 respectively) in hauls where muskellunge were taken. It was also closer to the surface in 1992, but not significantly so (Table 5). Percentage cover was greater in successful hauls in all three years, but was significantly greater ($p < 0.05$), only during 1992.

Aquatic Plant Relationships

Based on our earlier extensive sampling that suggested a negative correlation between muskgrass stem density and muskellunge use of a bay for a spawning and nursery site we separated plant species groups into muskgrass and non-muskgrass categories. A bay by bay comparison of the percentage of quadrats that were dominated (>50%) by muskgrass in muskellunge and non-muskellunge bays indicated no significant differences between bay categories (Table 6).

Seine haul determinations of muskellunge and non-muskellunge hauls were statistically correlated with muskgrass densities during 1992 and 1993 ($X^2 = 4.37$ and $X^2 = 4.01$ respectively). Specifically, hauls in which muskellunge were captured were characterized by having fewer quadrats dominated by muskgrass than unsuccessful seine hauls (Table 7). There were no signifi-

Variable (Year)	Muskellunge Taken	No Muskellunge Taken	P
% Cover ('91)	77.50 (10.22)	73.06 (4.80)	0.557
% Cover ('92)	81.95 (5.72)	71.25 (5.38)	*0.034
% Cover ('93)	97.78 (2.46)	74.97 (5.64)	0.051
Vegetative Height ('91)	40.83 (8.28)	30.94 (3.24)	*0.036
Vegetative Height ('92)	40.67 (5.92)	30.41 (4.70)	*0.027
Vegetative Height ('93)	66.11 (16.12)	33.99 (4.98)	*0.001
Distance from Surface ('91)	31.19 (9.48)	56.54 (5.20)	*0.005
Distance from Surface ('92)	58.33 (6.72)	65.34 (4.54)	0.164
Distance from Surface ('93)	29.72 (12.82)	67.64 (5.38)	*0.001

*Table 5. Comparison of values for vegetative parameters between successful and unsuccessful seine hauls in the St. Lawrence River (1991-1993) [Mean (2 standard error) (*= 0.05)].*

cant differences in 1991 (X^2 = 1.49). Overall, the total number of quadrats dominated by muskgrass were significantly lower in successful as opposed to unsuccessful seine hauls (X^2 = 16.67).

DISCUSSION

A sampling of bays in the 1000 Islands section of the St. Lawrence River has shown that bays used by muskellunge have lower densities of muskgrass than those not being used by muskellunge. They also appear to have somewhat less vegetative cover in the deeper part of the bays.

Within a bay, young-of-the-year muskellunge appear to distribute themselves in relation to aquatic vegetation. In addition to general habitat characteristics of the nursery area, muskellunge appear to be strongly oriented to physical aspects of the vegetation found in these habitats. The significance of percent cover, vegetative height, and proximity to the surface along successful seine hauls illustrate the utilization of specific habitat components within a bay. Vegetative height appears to be an extremely important habitat characteristic for muskellunge fingerlings. Vegetation within seine hauls capturing muskellunge was always taller than within unsuccessful seine hauls.

Our percentage cover data for the three years differed from other studies. Craig and Black (1986) reported values of percent coverage of the bottom by submergent vegetation between 19% and 50%. The average values of 78%, 82%, and 98%, reported for successful seine hauls in the current study are dramatically higher than those reported in Georgian Bay. These values are probably the result of muskgrass abundances which dominated in excess of 50% of all quadrats during all three field seasons. When encountered, muskgrass usually consisted of dense unbroken beds, with few gaps. Muskgrass was found to be one of the most abundant submergent species in the Georgian Bay study, but the relatively low cover values suggest that its density was much less than seen in the St. Lawrence River. Strand (1986) and Dombeck et al. (1984a) also reported spawning areas in Leech Lake, Min-

Table 6. Comparison of the number of quadrats dominated by muskgrass or non-muskgrass between designated muskellunge and non-muskellunge bays in the St. Lawrence River (1991-1993).

Year	Variable	Muskrass	Non-muskgrass	%
1991	Muskellunge Bays	81	58	58.3
	Non-muskellunge Bays	35	37	48.6
1992	Muskellunge Bays	72	51	58.5
	Non-muskellunge Bays	50	36	58.1
1993	Muskellunge Bays	48	46	51.1
	Non-muskellunge Bays	28	37	43.1
Total	Muskellunge Bays	201	155	56.5
	Non-muskellunge Bays	113	110	50.7

Year	Designation	Muskgrass Dominated	Non-muskgrass Dominated
1991	Muskellunge Taken	50	50
	No Muskellunge Taken	61	39
1992	Muskellunge Taken	49	51
	No Muskellunge Taken	65	35
1993	Muskellunge Taken	28	72
	No Muskellunge Taken	51	49
Total	Muskellunge Taken	127	173
	No Muskellunge Taken	177	123

Table 7. Comparison of the number of quadrats dominated by muskgrass or non-muskgrass vegetation between successful and unsuccessful seine hauls in the St. Lawrence River (1991-1993).

nesota to be located over dense beds of muskgrass. A X^2 analysis showed that muskellunge fingerlings were disproportionately located in areas where muskgrass was not dominant. Since it is more of a bottom cover plant and is usually shorter than many of the other common species such as *Potamogeton spp., Vallisneria, and Myriophyllum*, and since our results indicate that taller vegetation is generally preferred by juvenile muskellunge, stands of muskgrass do not appear to be a suitable habitat.

Depressed levels of dissolved oxygen at the water-substrate interface within beds of muskgrass offer the simplest explanation for the high mortality rates of muskellunge yolk sac larvae. Dombeck (1984a) has suggested that low dissolved oxygen levels near the sediment water interface lead to high mortality rates for the relatively immobile egg and yolk sac larval stages of muskellunge. Since dense beds of muskgrass hinder water movement this could lead to severe oxygen depletion and ultimately increased mortality. This hypothesis has been confirmed by other studies (Clapsadl 1993).

We can identify several reasons for juvenile muskellunge distribution within or near vegetation in close proximity to the surface. The coloration of muskellunge fry gives the appearance of a small dead stick (MacKay and Werner 1934), or a piece of aquatic vegetation, and helps to conceal it among the vegetation (LaPan 1985). This characteristic may allow muskellunge to remain relatively safe from aerial predators. Water temperature may be another factor influencing distribution. The historical distribution of muskellunge indicate that their range does not extend as far north as the northern pike (Scott and Crossman 1973). Bevelhimer et al. (1985) reported ideal muskellunge growth at 26°C, as opposed to 24°C for northern pike. Since the surface zone normally provides the warmest temperature, these areas would probably provide the best conditions for muskellunge growth. Furthermore, this surface habitat may be important because it allows separation from northern pike which are normally found deeper in the water column (Wahl and Stein 1989, Engstrom-Heg et al. 1986).

It is clear from this work that the nature and distribution of aquatic vegetation in muskellunge spawning and nursery areas are important habitat components affecting survival of muskellunge during the first few months of their life. Vegetation such as muskgrass can enhance mortality by reducing water flow and thus oxygen concentrations leading to high mortality on the immobile egg and yolk sac larval stages. Other types of vegetation that approach the surface of the water in the nursery area can enhance survival later in the summer by providing cover for young fish. The dynamics of the changing vegetation and the changing needs of the young muskellunge during this critical period in their life will provide a challenge for fishery managers trying to establish self-sustaining populations of muskellunge.

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Detection of Contaminant Exposure in Fish by Activation of Detoxication Enzymes

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ABSTRACT

Dredging at the General Motors (GM) site on the St. Lawrence River at Massena, N.Y. provided an opportunity to study the effects of elevated contaminant levels on phase I (cytochrome P450; ethoxyresorufin O-deethylase, EROD) and phase II (conjugation; glutathione S-transferase, GST) enzyme activation in rainbow trout and black bullhead caged at the GM plant and a control site. Trout (1994, 1995) liver enzymes were not activated after 35 or 42 days, even though fillet PCB levels were elevated 37 times above control site values. Bullhead (1994) liver EROD activation occurred to 3.5-times the control values; GST levels were slightly elevated, but not significantly. These studies indicate that fish caged in contaminated areas may respond differently to suspended PCBs, and question the appropriateness of these enzymes as indicators of contaminant exposure for all species.

INTRODUCTION

The St. Lawrence River is a polluted ecosystem that has been the subject of remedial action plans by both the United States and Canada. This aquatic resource has been abused for decades by both the massive industrial sector and human populations lining its banks. A huge variety and quantity of toxicants have been dumped into the River and watershed. Although discharges have been reduced or curtailed and levels of toxicants are progressively declining or at least stabilized (Government of Canada 1991), the negative impact of these chemicals will exist for some time (e.g., Sloan and Jock 1990, Fitzgerald et al. 1992, Skinner 1992).

One long-term consequence of pollution in the St. Lawrence River has been the accumulation and concentration of contaminants in wild fishes to levels that are unsafe for human consumption (Sloan and Jock 1990). Located adjacent to the GM plant and near other industrial plants (Reynolds Aluminum, Alcoa Aluminum), Mohawks of Akwesasne have been particularly affected (Figure 1). Not only has a food source been lost, but an important cultural tradition has been jeopardized (Buttner and Puglisi 1993). In response, aquaculture has been pursued on a limited basis as a socially acceptable alternative. Community members grow "clean fish" as a substitute for harvesting wild fish (Mascari 1994; Buttner et al. 1996). It has been demonstrated that fish with greatly reduced or non-detectable levels of contaminants can be grown in waters of the St. Lawrence River by the selection of appropriate sites and following standard aquacultural practices (Kadlec 1994; Buttner et al. 1995). However, a rapid and sensitive assessment method is needed to confirm that each batch of fish is in fact clean.

Activation of specific enzyme systems has been correlated with the presence of chemical contaminants. For instance, both laboratory (e.g., Otto and Moon 1995) and environmental (e.g., Otto et al. 1994) exposure of fish to organic contaminants like polychlorobiphenyls (PCBs) have demonstrated the activation of cytochrome P450 (Phase I, monoxygenases) and conjugation (Phase II) enzymes. The use of these enzymes as a tool to establish the presence of contaminants has potential as a practical monitor for fish grown in the St. Lawrence River and other less pristine sites (Otto et al. 1994; George et al. 1995).

A unique opportunity presented itself with the removal of contaminated sediments at the GM Plant (Massena, NY). The dredging operation, which initiated in 1994, facilitated the examination of enzyme activation in fish maintained at the GM site (disturbed, contaminant area) and at a nearby undisturbed (contaminant-free) control site (Figure 1).

	METHOD	14 October, 1994	18 Nov., 1994	22 June, 1995	2 August, 1995
pH	meter	6.4/6.4	7.8/8.0	8.0[2]/---	8.6[2,3]/---
Alkalinity (mg/L)	titrimetric	88/94	90/96	---	---
Temperature (°C)	thermistor	15/15	9.8/10.1	17.9[2]/16.7	23.5[2,3]/22.6
Conductivity (umhos/cm)	meter	180/210	250/250	240[2]/---	250[2]/---
D.O. (mg/L)	polarographic meter	9.4/9.5	11.5/10.6	9.8[2]/---	8.2[2,3]/---
Current (m/sec)[1]	meter		0.58/0.09-1.2	---	---
Water Column PCB (µg/L)	mass spectro-photometer	---	---	ND[4]/0.34	ND/1.32[5]

1/current increased with distance from shore.
2/water samples were not taken at the Control site, but at the water intake for the Mohawk Community at the mouth of the Racquette River.
3/water samples taken 14 August, 1995.
4/ND - 0.062 µg/L is detection limit
5/sample collected 29 July, 1995.

Figure 1. Activation of selected enzyme systems was determined in rainbow trout and black bullhead maintained in cages near Massena, N.Y. (insert). The dredge (A) and control (B) sites in the St. Lawrence River are separated by approximately 10 km.

	METHOD	October 14, 1994	Nov. 18, 1994	June 22, 1995	August 2, 1995
pH	meter	6.4/6.4	7.8/8.0	8.0[2]/---	8.6[2,3]/---
Alkalinity (mg/L)	titrimetric	88/94	90/96	---	---
Temperature (°C)	thermistor	15/15	9.8/10.1	17.9[2]/16.7	23.5[2,3]/22.6
Conductivity (umhos/cm)	meter	180/210	250/250	240[2]/---	250[2]/---
D.O. (mg/L)	polarographic meter	9.4/9.5	11.5/10.6	9.8[2]/---	8.2[2,3]/---
Current (m/sec)[1]	meter		0.58/0.09-1.2	---	---
Water Column PCB (ug/L)	mass spectro-photometer	---	---	ND[4]/0.34	ND/1.32[5]

1/current increased with distance from shore.
2/water samples were not taken at the Control site, but at the water intake for the Mohawk Community at the mouth of the Racquette River.
3/water samples taken August 14, 1995.
4/ND - 0.062 (g/L is detection limit.
5/sample collected July 29, 1995.

Table 1. Water quality parameters and analytical methods used to describe the GM and control site at Akwasasne (Figure 1). Values observed at both sites are reported (control/GM).

MATERIALS AND METHODS

The study was conducted in two phases, fall 1994 and summer 1995. Rainbow trout (*Oncorhynchus mykiss*) and black bullhead (*Ameiurus melas*) were maintained in cages located at the GM and control sites for 35 and 42 days (Figure 1; Buttner 1992; Otto et al. in press). Water quality was monitored to describe test conditions (Table 1). Fish were stocked (14 October 1994, 22 June 1995), fed a commercial ration (40% protein), harvested (18 November 1994, 2 August 1995), processed and analyzed for enzyme activities (Otto and Moon 1995). In 1995, skinless fillets were retained, frozen at -80°C and analyzed for PCB content (Bush et al. 1989). Water contaminant data were obtained from the Environment Division, Mohawks of Akwesasne, in 1995 (Table 1).

RESULTS AND DISCUSSION

Except for soluble PCBs, water quality at both control and test sites was similar and varied with time (Table 1). Although many enzymes were assayed, activities of two hepatic enzymes routinely used to assess exposure to organic contaminants, ethoxyresorufin 0-deethylase (EROD; cytochrome P450 phase I enzyme) and glutathione-S-transferase (GST; phase II, conjugating enzyme) are reported. These enzymes are activated as part of the *Ah* gene battery by polyaromatic hydrocarbons (PAHs) and other halogenated compounds, including a variety of PCBs (e.g., Xiao et al. 1995). Phase I and II enzymes catalyze the conversion of toxicants to more water soluble metabolites for excretion.

Despite vandalism, theft, cage loss, and limited numbers of appropriate-sized fish (≥ 140 g) in 1994, enzyme activity in 10 fish from each site (average = 336 g, control; 270 g, GM) was determined (Table 2). In 1995, trout survival was 100% and enzyme activation for 10 fish per site (average = 166 g control; 141 g, GM) was determined. Results indicated no activation of either EROD (Table 2) or GST (Table 3) in trout liver between either stock and harvest times or control and GM sites. In fact, activities generally decreased, reaching significant differences in two cases (see Tables 2 and 3). The absence of enzyme activation occurred even though total fillet PCB content increased. At stock, total PCB content for five trout was 0.049 ±0.01 μg/g; and, for six fish from each of the sites at harvest averaged 0.053 ± 0.01 and 1.8 ±0.4 μg/g for the control and GM sites, respectively.

Water current was viewed as excessive for bullhead and required special treatment (Table 1). In 1994, bullhead (and cages) were located in areas of reduced flow or with tubing placed inside the cage in 1994. Enzyme activity in eight fish from the control site (average = 72 g) and 10 fish from the GM site (average = 80 g) was determined (Table 2); no data on the enzymes were obtained in 1995. EROD activities were increased at both sites, although the activation was greater at the GM site (3-fold) than the control site (2-fold) compared to stocking values (Table 2). GST activation was not noted, although the very large variance in activities at the GM site precluded significant differences (Table 3).

	EROD as mol/min/mg protein	
	STOCK	**HARVEST**
Rainbow trout - 1994		
Control	19.7 ± 3.1	25.3 ± 3.6
GM site		11.6 ± 2.1[1]
Rainbow trout - 1995		
Control	11.9 ± 2.4	5.8 ± 0.8
GM site		7.4 ± 1.0
Bullhead - 1994		
Control	3.0 ± 0.3	6.8 ± 0.9[1]
GM site		10.5 ± 1.0[1]

1/Significantly different from value at stocking (Anova, P<0.05).

Table 2. Mean (± SEM) hepatic ethoxyresorufin O-deethylase (EROD; pmol/min/mg protein) in five to ten rainbow trout and black bullhead exposed for 35 (1994 study) and 42 (1995 study) days to St. Lawrence River water at the GM and Syne Channel (control) sites (see Figure 1).

Our objective was to show that detoxication enzyme activation could be used to assess the presence of contaminants. Trout accumulated and bioconcentrated PCB contaminants present in the water column at low concentrations at the GM site. The levels obtained in the skinned fish were similar to those found by Kadlec (1994) at a nearby site. One must assume that the bullheads also accumulated PCBs, but none of these fish were analyzed for contaminants. These studies showed, however, that detoxication enzyme systems in bullheads responded quickly, but trout did not respond even though they were caged in the same area.

The reason for this inconsistency between species must be assessed. EROD activity is a widely used and reliable indicator to assess exposure to organic contaminants (Otto et al. 1994; George et al. 1995), but no activation occurred even though trout accumulated PCBs. Laboratory experiments where a single high dose of a PCB congener (3,3'4,4'-tetrachlorobiphenyl; 5 mg/kg) was injected into trout did show significant induction of phase I and II enzyme activities (Otto and Moon 1995). The trout caged at the GM site were constantly exposed to high PCB content over the experimental period. High PCB concentrations are found to inhibit EROD activities in vertebrates including fish (e.g., Hahn et al. 1993) which may explain our results with trout but questions the reliability of this enzyme as an index of contaminant loading in an environmental context. The inability of the trout to detoxicate the accumulated PCBs either by phase I (EROD) or II (GST) enzymes could ultimately impact the survival of this species in a highly contaminated area and their suitability for human consumption. The bullhead, however, may not be so affected.

In conclusion, the present data indicate that fish caged in highly contaminated areas respond differently to the contaminants. The use of enzyme activation as an index of contaminants appears inappropriate for the trout, but may be suitable for the bullhead. The different responses by these two species may ultimately impact upon the survival of the species and its usefulness in aquaculture. Further experimentation appear warranted to validate our findings and to assess their implications.

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Table 3. Mean (± SEM) hepatic glutathione-S-transferase (GST; nmol/min/mg protein) in five to ten rainbow trout and black bullhead exposed for 35 (1994 study) and 42 (1995 study) days to St. Lawrence River water at the GM and Syne Channel (control) sites (see Fig. 1). Significantly different at stocking([1]Anova, P<0.05) or from control site at harvest ([2]; Student t-test, P<0.05) are indicated.

	GST as nmol/min/mg protein	
	STOCK	**HARVEST**
Rainbow trout - 1994		
Control	437 ± 33.8	390 ± 40.7
GM site		307 ± 21.4
Rainbow trout - 1995		
Control	725 ± 59	538 ± 15.0
GM site		455 ± 25.0[1,2]
Bullhead - 1994		
Control	718 ± 23.4	700 ± 34.8
GM site		931 ± 139

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