

GREAT LAKES FISHERY COMMISSION
Research Completion Report *

**LAKE ONTARIO:
A GREAT LAKE IN TRANSITION
GREAT LAKES MONOGRAPH NO. 2**

by

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March, 1989

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PREFACE

Clean, usable water resources are an essential element in the quality of life which makes the Great Lakes region an attractive place to live and develop business opportunities. These Lakes have experienced a diversity of perturbations, however, including increased loadings of nutrients and toxic chemicals, as well as restructuring of the fish communities through both exotic invasions and purposeful stocking. Human intervention has been by default as well as deliberate. Although such a large natural system is resilient and adaptive, continued human intervention will need to be sensitive to the interconnectedness and complexity of system parts when dealing with water quality, the lakes' flora and fauna, and even climate.

Basic Great Lakes research still falls short of providing a sound scientific basis for effective management of this vital freshwater resource, especially with regard to issues that relate to pollution control and ecosystem rehabilitation. The need for an integrated, cross-disciplinary perspective for seeking scientific knowledge on the Great Lakes has never been greater. Instead of focusing upon a single chemical or a particular fish species, we must begin considering how the entire ecosystem works, how components are interconnected and affect one another, and then attempt to preserve the integrity of natural resources using this holistic knowledge.

The question here is, where do we proceed from this point forward? At what state is our scientific knowledge and what level of integration can we achieve with our present understanding, that will begin to address the complex problems related to ecosystem management? Better measurement of how ecological factors affect the outcome of alternative management decisions is needed. Quantitative information is required to assess the probability that proposed strategies will result in desired changes, and not surprise us in the future. Too often decisions are made on solving a problem at hand as quickly as possible, with little regard for the history that led up to it or the future global consequences of actions taken. One reason is that "piecemeal" or "quick-fix" approaches often work well in the short-term of economic and political worlds. We must develop a feeling for foresight, if for no other reason, because the escalating rate of change leaves less and less time to alter a threatening trend after recognizing its dangerous implications. It took hundreds of years to evolve a biologically productive, historic fish community; decades to destroy this community; and less than a decade to restore a new but variable fish community mix that may be as productive.

Lake Ontario

Management of Lake Ontario, therefore, is a challenging task. Management decisions will need to be based on a high level of analytical thinking supported by a vigorous, cross-disciplinary research program that provides the necessary expanded knowledge base. Only then will strategic points for intervention be possible to identify and implement. Thus, the need for an assessment of the state of our present knowledge on the Lake Ontario ecosystem. That is the intent of this monograph. The National Oceanic & Atmospheric Administration's National Marine Pollution Program Office and the Great Lakes Fisheries Commission recognized this need and provided the funding to accomplish this task.

In this monograph we do not simply present a re-analysis of the extensive data available on this Great Lake, but rather we want to offer strategic statements that address the issues and describe how we can progress in solving the problems at hand using the large data bases already available. An attempt is made in this monograph, through the assessment of long-term trends and synthesis of data for Lake Ontario, to demonstrate how the scientific knowledge base might be improved so that future management practices will be effective in protecting and rehabilitating this vital freshwater resource.

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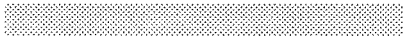
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CHAPTER 1

INTRODUCTION

The Great Lakes have changed and they have changed society. Pristine and untouched by human hand for thousands of years, the Great Lakes have felt the influence of man but for a moment in time. The hundreds of years that have passed since the first European explorers roamed these shores seem like an eternity to modern man. To the Great Lakes, however, these years have been a brief, but rude, awakening.

To the first North Americans, the Lakes and their resources were valuable sources of food and sustenance. Europeans who appeared nearly 400 years ago hoped the Lakes would serve as a water route to the Orient. Later they served as an avenue for exploration of the North American interior. The Lakes' value as a transportation lifeline to distant markets dominated thinking for the 300 years following the establishment of European commerce. First they served as a network to move valuable furs to Europe. Then they became a route for European settlers into the region and for shipping natural resources and farm products out of the region to the prosperous markets of the Eastern seaboard. The Lakes' trade then fed the basin's growing industrial centers with iron ore, lumber and coal. Foreign trade with Europe resumed with the completion of the St. Lawrence Seaway in 1954.

Attitudes toward the Lakes have changes as the Lakes and their resources changed. The fisheries provided food for growing American markets, until the fish populations declined. To the industrial centers of two nations, the Lakes provided

a convenient place to dispose of waste until the adverse effects became obvious. Municipalities and the growing population they serve have depended upon the Lakes as a source of water.

The Great Lakes also are becoming recreation mecca. This probably would have seemed unimaginable to the voyagers in 1700 and perhaps to homeowners along Lake Erie in 1965. Yet to the thousands who cruise the Lakes in luxury steamers at the turn of this century and the boaters of today, it was and is a pleasant reality. So, too for the thousands who flock on a summer day to a sandy lakeshore beach to swim or to a pebbled lakeshore to cast their fishing line.

It has been predicted that within ten years 75 percent of the entire population of the United States will live within 50 miles of the ocean and the Great Lakes (Federal Reserve Bank of Chicago 1985). These large numbers of people will make intensive, varied and often conflicting uses of coastal areas; uses which in many circumstances already have degraded environmental quality and have contributed to substantial reductions in natural resources, in contrast to historical records of pristine waters. Many of our major cities in the coastal zone were established by colonists as early settlements. Early accounts of these settlers, who had left crowded and polluted cities bordering European coastlines, praised the clarity of waters and the bounty of the living resources found by settlers.

Because of the enormous importance of the Great Lakes to society resulting from the numerous and varied purposes they serve, and because of the stresses imposed by these multiple and conflicting uses, it is not surprising that society has directed its attention to protecting and, when necessary, to rehabilitating these valuable natural resources. This attention has been directed at developing strategies to stop pollution and to enhance the aesthetic values and living resources of the Great Lakes.

The responses of governments at all levels to citizens' requests for action, and the multifaceted programs which have been generated by our federal, provincial and state agencies responsible for protecting and managing our Great Lakes waters, have been laudable. One can not argue with their intent. But there is a problem. The remediation programs have not always worked; at least not very well. The programs have been only marginally effective in improving the Lakes' quality and our ability to manage their resources. It is exceedingly difficult to effectively manage such a vast and complex ecosystem as the Great Lakes when you do not totally understand system dynamics.

Over the years, the environmental effects resulting from the transboundary movement of pollution have become more complex and difficult to define. For example, in the Trail Smelter case in the 1930s, a single sulphur dioxide source in Canada was readily identified as the cause of environmental damage on both sides of the border and the solution pointed to reduction of the pollution from that source. The problems encountered in recent decades are more complicated than that fairly simple early case.

Occurrences of eutrophication and toxic contamination of the Great Lakes and the transboundary flow of air pollutants stem from a multitude of causes and result in a wide variety of environmental effects. Consequently, environmentally effective and socially acceptable solutions to the

problems require greater scientific understanding of the issues than had been the case in the early part of this century.

The Great Lakes are presently experiencing perturbations through nutrient and contaminant fluxes as well as introductions of fish and zooplankton that represent potential large-scale changes in function of these systems. If the pathways and fate of nutrients and contaminants are not understood, and related to man's activities, it will be impossible to make learned judgments about consequences of particular actions or resource exploitations. Management of the Great Lakes, therefore, challenges our scientific endeavors. Those with whom we charge this enormous responsibility must be provided access to the best scientific and technical information available at any given time and must be assured of a constant flow of new knowledge. Management, like politics, is the art of the possible. With greater scientific knowledge and understanding, the limits on what is possible are extended and the likelihood of making appropriate and effective management decisions is increased.

Lake Ontario, one of the Laurentian Great Lakes, poses management issues of an economic and public health nature. For example, the food web supports important sport, commercial, and forage fisheries which provide economic vitality to regions surrounding the Lake. Managing Lake Ontario waters and maintaining productive fisheries in the face of multiple stresses requires thorough scientific understanding of the interconnected components of this ecosystem. It must be recognized that interactions among biota control the rates of biochemical and biogeochemical processes, including deposition of sediment, cycling of materials, and residence times of toxics.

The history of the Great Lakes is marked by change; change in attitudes and perspectives, change in use and abuse, change in the Lakes themselves and society.

This report offers a glimpse of this change to one of our Great Lakes, Lake Ontario. The long-term trends and patterns in the biophysical nature of Lake Ontario will be highlighted, and an attempt will be made to illuminate how these changes came about and what they have meant to society. While significant advances have been made in our understanding of how any one of our Great Lakes operate, the level of uncertainty remains unacceptably high when predicting the consequences of management strategies. Until scientific knowledge is increased

substantially, the ambiguity of which management strategies to apply and what results to expect from a given strategy, or combination of strategies, will likely remain high. To reduce uncertainty is to improve understanding, first through synthesis and integration of existing information and secondly through new research at the most creative level possible. Only then will strategic points for management intervention be possible to identify and implement.

Lake Ontario

CHAPTER 2

CULTURAL PATTERNS

Historical Development

Human influence on the Great Lakes began many hundreds of years ago when native American tribes discovered the Lakes on their journeys from the northwest. The lakes not only provided abundant fish, waterfowl, and edible wetland plants but also an easy transportation route across the region. The first value attached to the Lakes must have been for their productive food resources, since trade among Great Lakes tribes was slow to develop. In the spring, summer, and fall months, numerous native villages moved to the shores of the Great Lakes to take advantage of their abundant resources. When these villages were disbanded in late fall, individual family groups would move to winter hunting grounds generally in more inland areas.

The harvesting of fish, fowl, and water plants by the tribes of the Great Lakes region did little to affect the ecology of the Great Lakes. Their population numbers never reached a point where over-harvesting threatened the ecosystem. Their attitudes toward resources were also very different than those of the Europeans. They viewed resources as gifts that could be used to sustain and shelter. The resources were not owned by individuals, and it was against cultural norms to waste them. Because of these attitudes, native Americans have been labeled as the first conservationists.

In 1615 a new chapter in the history of the Great Lakes region began when Le Caron, a Recollect friar, and Samuel de Champlain, the founder and Governor of Canada, first paddled the Great Lakes.

These early European explorers traveled the Great Lakes for the sole purpose of finding a water route to the rich lands of the Far East. They hoped to avoid the long and dangerous journey around the tip of Africa, the only route to the Far East known to exist at the time. When it became clear that the Great Lakes were not a pathway to China, the focus shifted to exploration of the North American interior. Not until the 1650s, when traders returned to Quebec with rich cargos of furs nurtured by abundant marshes and wetlands, did the Great Lakes region begin to be eyed for the vast resources it contained. This marked the first real shift in European attitude about the Great Lakes and triggered a hundred-year struggle among the European powers to claim the region as their own.

During the initial European visits to the Great Lakes region in 1615, Lake Ontario was discovered by the French explorer Etienne Brule. A peace treaty was signed in 1667 between the French and Iroquois Indians, and shortly thereafter Lake Ontario witnessed the first large sailing ships on its waters. In 1749 the city of Toronto was founded. From 1689 to the culmination of the War of 1812 however, the Lake Ontario region was immersed in a series of wars which inhibited most settlement to this region.

During the last 150 years the development of the Ontario Basin has proceeded with haste. The battles for territory, so common during the era of empires and colonies, gave way to nation building, city building, and industrialization. The warriors of the previous era gave way to, or themselves

became the entrepreneurs, farmers, and laborers who ran the mills, tilled the soil, and provided the skills and services required for modern industrial economies.

The development of the Lake Ontario region proceeded along several lines which took advantage of the many resources within the basin. The lake became a major highway of trade and served as fertile fishing grounds. The first diversion of the Niagara River to the Erie Canal occurred in 1925, and in 1926 the Erie Canal was opened. In 1929 the Welland and St. Lawrence Canals were opened. The fertile land that had provided the original wealth of furs and food yielded lumber, then wheat, then other agricultural products. Bulk goods such as iron ore and coal were shipped through Lake Ontario ports and manufacturing grew. These events in history comprise a cultural heritage that is probably unparalleled anywhere.

Lake Ontario also supported a thriving commercial fishing industry during the first decades of the 20th century. Famous for whitefish, lake trout, and other species, the industry suffered setbacks from overfishing and the deadly effects of the sea lamprey, which was introduced through the man-made canal systems.

In 1985, Ontario harvested 1.7 million lbs. of fish while the New York State catch amounted to 200,000 lbs. This was the first time in the 20th century that the total commercial catch for Lake Ontario dropped below 2 million lbs. (Great Lakes Fishery Commission 1986). High value commercial fish production has continued declining to the present time. Control of the lamprey and introduction of new species of fish, however, are changing the pattern. More recently, contamination by metals (e.g., mercury) and pesticides have affected both the commercial and sport fishery in some areas of the Region.

Manufacturing Era

A description of development within the Great Lakes Basin in general, will give a good background for understanding the development and associated environmental problems in the Lake Ontario Basin. Industry in the Great Lakes Basin has been oriented towards manufacturing. In 1970, nearly four million Basin residents were employed in manufacturing, representing about 35 percent of the total number employed. The major manufacturing employers were producers of primary metals, food, and kindred products. Agriculture and mining employed 1.8 and 0.3 percent of the Basin's workers, respectively. Agricultural employment decreased 50 percent between 1940 and 1960, both nationwide and in the Great Lakes Basin. By 1970, it had decreased another 34 percent in the Basin to a total of 200,000 employed. The number employed in mining has decreased nationally but has remained fairly constant at about 39,000 in the Great Lakes Basin.

Industrial production is a key factor in the economic strength of the Great Lakes Region. The steel-producing districts in counties immediately adjacent to the Great Lakes, and served by lake ports, produced 50 million tons of steel in 1970, or 39 percent of the national total. The transportation of iron ore is considered to be confined to the St. Lawrence Seaway, since no economically viable alternative exists. This demonstrates the importance of the Lakes themselves and the commercial navigation which supports Basin industries.

The Great Lakes Basin also contains significant concentrations of petroleum refining industries and manufacturers of chemicals, paper and food products. These manufacturing industries account for about 80 percent of the Basin's industrial water requirements. This concentration of manufacturing has contributed to water quality problems in the Basin. Consequently, the heavy concentration of industrial activity in the region has played a major role in this region's

international standing as a manufacturing center, but has also added significantly to the pollutant load of waters within the Basin.

Lake Ontario, the lowermost of the Great Lakes, is the recipient of waters (and contaminants) from all upstream sources. These enter via the Niagara River and the Welland Canal. In addition, Lake

Ontario receives contaminants from municipal and industrial sources located in its basin, nonpoint land runoff via its tributaries, and direct atmospheric deposition. The drainage basin of Lake Ontario (Figure 1), including the international section of the St. Lawrence River, covers an area of approximately 3.3×10^6 hectares within the Province of Ontario and about 4.1×10^6 hectares in the

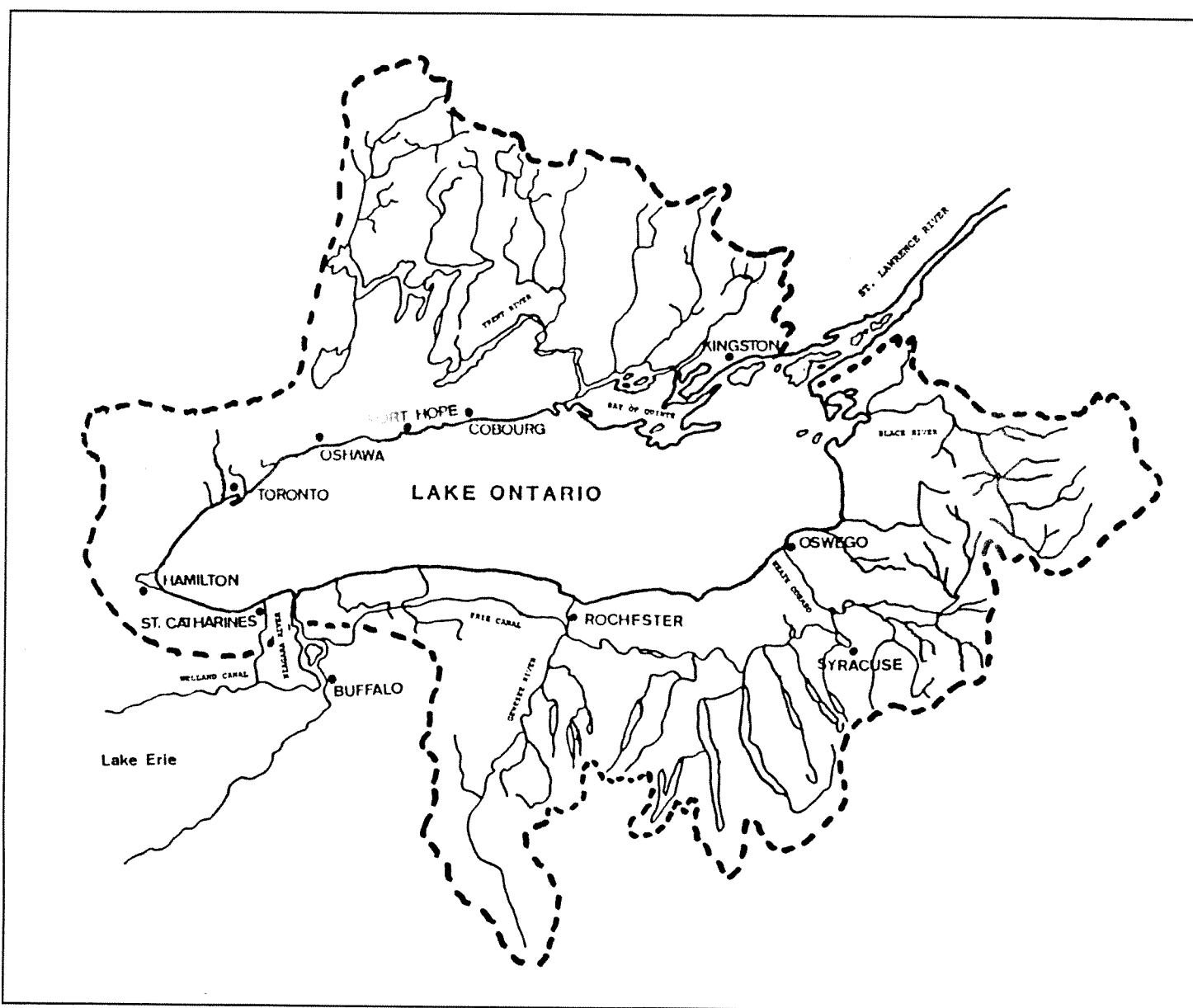


Figure 1 Map of the Lake Ontario Basin showing the watershed outline and the major tributaries of the watershed.

Lake Ontario

United States. Lands in the Lake Ontario drainage basin are used primarily for agricultural, industrial, recreational, municipal and forestry purposes. The Lake Ontario region has been a vital, growing, industrialized center of commerce since the 1940s.

The development of its plentiful water routes, including the New York Barge Canal, the St. Lawrence Seaway and the interconnection of the Great Lakes shipway, has placed this region in a strategic geographical location within the trade zones of the United States and Canada. The economy of the region is closely tied to these navigable routes which have provided an impetus for the development of a wide diversity of industries. One half of the \$150 billion per year U.S.-Canada trade, starts and ends in Great Lakes states and provinces. Fifty-four percent of all U.S. trade to Canada starts and ends in Great Lakes ports while 53 percent of the trade from Canada to the U.S. also begins and ends in Great Lakes ports (Federal Reserve Bank of Chicago 1985). Shipping on the Great Lakes is an irreplaceable means of transporting bulk goods cheaply. A ton of grain for example, moving by ship from the Port of Duluth to mills in Buffalo, costs the shipper approximately half as much as the same trip by rail (Thurow, et al. 1984).

Industry around Lake Ontario, in the Province of Ontario, is centered on the urban fringe skirting the western end of the Lake from Oshawa to St. Catharines, whereas in New York State the activity is based in specific urban centers. Principal among these are the Rochester and Syracuse areas, with Buffalo located on the Niagara River in the same metropolitan area as Niagara Falls. In Ontario, Niagara Falls, Hamilton and Toronto metropolitan areas are the dominant urban centers. Dominant industries include steel, automotive and transportation, paper, rubber products, chemical, photographic equipment and petroleum refining. Power production, including thermal-electric, hydroelectric

and nuclear, is also extensive. The ready availability of raw materials and energy sources, due largely to cheap transportation and hydropower development, and the central location relative to both Canadian and American markets, have been decisive factors in the industrial development of the Lake Ontario region.

The Province of Ontario and, in particular, the western Ontario region stretching from the Toronto conurbation to Hamilton-Niagara Falls, is the only truly mature industrial-economic region in Canada. Its industries, directly or indirectly, are highly dependent upon the water resources of the region. In Canada, the development of Hamilton both as a port and as an industrial center typifies this dependency. Iron, nickel and other ores are supplied from the Lake Superior mining areas. Coal is transported by ship and by railroad from nearby fields in the Appalachian Mountains and Indiana. As a result, Hamilton is the center of Canada's primary metals industry, producing steel, nickel and other metal products. Niagara Falls and Toronto, because of their proximity to this source of intermediate materials, have become major producers of farm and transportation equipment and machinery.

The conurbation of Toronto deserves special attention. Possessed with neither major hydropower sites nor coal or mineral resources nearby, Toronto developed from its historical basis as the region's capital to become the major transportation center of southern Ontario. Experiencing the influence of its proximity to both Montreal and New York, it became the nucleus of the region's roads and railroads as well as an active port. Wide diversity is a major asset of Toronto's industry. Besides the iron and steel products previously mentioned, other leading industries include food and beverage, electrical apparatus and supply and printing and publishing. Overall, 30 percent of the manufacturing employment of the Province is in Toronto. In addition,

Toronto is one of the nation's leading financial centers and an excellent educational center.

Just east of Toronto is the Oshawa area, Canada's leading automobile production center. Further east, industrial development in the Lake Ontario basin is limited. Peterborough is a center of the lumber trade and flour milling. Kingston and Cornwall are known for synthetic textiles and pulp and paper industries. Ottawa, the nation's capital, is noted for its printing and publishing and its basic research facilities mainly associated with government laboratories.

On the New York State side of Lake Ontario, there are three large industrial centers, Buffalo-Niagara Falls, Rochester, and Syracuse. The development of Buffalo-Niagara Falls, although preceding that of Hamilton, is similar in the manner in which it has been influenced by the Great Lakes water route. The primary metals industry predominated, with Buffalo being one of the nation's leaders in steel production through the early 1970s. Transportation equipment is the second leading industry of the area, with the aerospace industry also being a leader. Machinery, food processing, chemicals and paper are also prominent industries in the Buffalo-Niagara Falls area. Employment in manufacturing industries in the Buffalo economic area exceeded 200,000 in 1965.

Rochester is a manufacturing center largely oriented to the instruments industry, photographic equipment being the largest subgroup. The chemicals, electronics and food processing industries are also prevalent. Manufacturing employment for the Rochester area was nearly 140,000 in 1965. Syracuse is known for its chemicals industries and was the world's largest producer of soda ash and caustic soda chemicals near the shores of Onondaga Lake in the mid-1960s. Electronics and electrical equipment

industries are the largest employers in the area, while primary metals and related durable goods groups, papermaking and food processing are also prevalent industries. Aluminum producing and processing plants are found in Massena, while the papermaking industry is located in Watertown and St. Lawrence County areas.

Agriculture is of major significance to the economy of the Province of Ontario. About 70 percent of all commercial farms in Ontario are either dairy or cattle farms, with the eastern Ontario region north of Lake Ontario having the highest concentration. Principal crops are hay, oats, mixed grains and corn. Wheat has declined greatly in recent years, while corn production is rapidly increasing. Fruit and vegetable farming is extensive near the Lake Ontario shoreline, particularly in the Niagara Peninsula region. Here, the continuing competition for land use from urbanization has removed more than 40 percent of the original land in production, and within twenty years, the valuable Niagara region may be permanently lost for agriculture. In terms of profitability, tobacco is also an important crop grown in northshore counties bordering Lake Ontario.

In the southern portion of the Lake Ontario basin, in New York State, only about 2 percent of the population is engaged in agriculture. As in Canada, livestock and livestock products, mostly dairying, dominate U.S. agriculture in the basin. The fruit and vegetable belt extends from the Niagara peninsula eastward along most of the southern Lake Ontario shoreline. Off the north end of the Lake, in the snow belt region surrounding the Tug Hill upland area, farming is generally poor with dairying and lumbering predominating. The Adirondack Preserve is largely non-agricultural. Field crops for the basin as a whole are primarily dairy oriented, including hay, oats, mixed grain, corn, barley, and wheat.

Basin Demographic Trends

Since the mid-1800s changes in the Lake Ontario basin have occurred rapidly. For example, in the early nineteenth century less than 300,000 people inhabited the entire Great Lakes basin (MacNish and Lawhead 1968). By 1860 the population had increased to 1.4 million in the Ontario basin (Beeton 1969). In 1970 the Lake Ontario basin had the third largest population of the five basins of the Great Lakes. In the 1980s the population alone numbered in excess of 8 million people. Figure 2 indicates the population changes that have occurred in the Ontario basin overall, as well as in the major urban centers around Lake Ontario for more recent times. Today the Province of Ontario population represents more than twice that living in New York state. In addition, the Ontario population is growing at a faster rate. For example, during the decade 1970-1980 the Ontario population grew at an annual average rate of 1.7 percent (Statistics Canada 1986). In contrast, the New York population remained almost unchanged. Population included in

municipal and other non-agricultural areas is around 6.6 million, compared with the total basin population of around 8.0 million persons. About 62 percent of the total basin population is Canadian and this percentage is expected to increase to around 70 percent by 2020.

Municipalities encompass more than 2×10^5 hectares of land, principally for residential use, in the Lake Ontario region. As of the mid-1970s the active breakdown of land usage along the shore of Lake Ontario followed that outlined in Table 1. Urban/residential use was the greatest on the U.S. shoreline followed closely by forest and brushlands. On the Canadian side of the lake urban residential was only third in land usage, preceded by both forest and brushlands and agricultural use.

With changes in demography of the Lake Ontario basin, there have been changes in economic development as well in recent years. For example, as Figure 3 demonstrates, there has been a significant shift in the industrial structure of the basin between 1950 and 1980. This shift is most notable in a move away from the heavy manufacturing industries toward the more service-oriented industries. This shift is notable for all major metropolitan areas of the basin. The shift is related to changing tax structures and heavy manufacturing costs over the last three decades, as well as to new opportunities for service-oriented industries that are heavily tied to changing patterns in leisure time in the North American society.

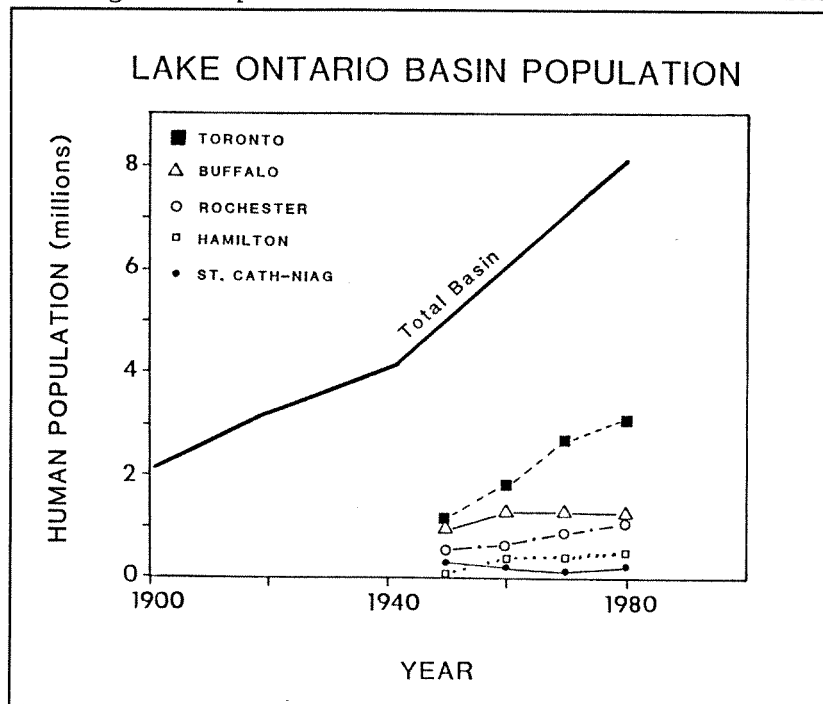


Figure 2. Lake Ontario Basin human population, 1900-1985, along with population trends over the last 30 years for the larger cities in the basin.

Water Usage

Lake Ontario is of considerable socio-economic value, providing water for human consumption, manufacturing, transportation, power, recreation and a variety of other uses. Withdrawals by municipalities for public water supplies constitute the major

State of a Great Lake

Table 1. Land use around the Lake Ontario Basin in percent total miles (Doworsky 1985).

LAND USE	CANADA	UNITED STATES
Urban Residential	25.5	42.4
Industrial - Commercial	3.2	0.4
Agricultural	30.9	13.1
Forest - Brushland	40.3	5.7

consumptive use. Approximately 2.6 million cubic meters are withdrawn daily, predominantly in Ontario where the population distribution is heavily oriented along the shoreline. Use for power generation, essentially for cooling purposes in thermally generated power, is also substantial (more than 36 million cubic meters withdrawn per day), but little of this water is actually consumed. For example, in 1983, 23.7 billion kilowatt hours of hydroelectric power were generated in the U.S. and another 20 billion in Ontario by waters flowing through the Great Lakes.

As indicated above, Lake Ontario serves as a major source of potable water. Withdrawals are made at many points around the lake. More than 75 percent of the population living in the basin (approximately six million people) relies on Lake Ontario water for drinking. The Lake also serves as a major receiving body for many of the waters that are withdrawn. The Niagara River serves as a source of $1.47 \times 10^3 \text{ m}^3/\text{day}$ of wastewater flow to Lake Ontario with 66 percent of this wastewater represented by municipal sewage treatment facilities and 34 percent by industrial discharge. Within the Lake Ontario basin a total of $6.3 \times 10^3 \text{ m}^3/\text{day}$ of wastewater flows occur from both tributaries (excluding the Niagara River) and direct lake discharge. Municipal discharge comprises 57 percent of this total while

industrial release makes up 43 percent. In addition, the Province of Ontario contributes 76 percent of this total municipal and industrial discharge of wastewater to Lake Ontario receiving waters. To date, there have been no human health problems identified which can be associated with drinking Lake Ontario water via public water supplies, that also receive all these discharges. Chemicals of concern or potential concern however, may be unmonitored by present methodologies.

In more recent times, Lake Ontario as well as the other Great Lakes, have been impacted from industrial, municipal, non point, atmospheric, and landfill leachate releases. DDT, mirex, dioxin, PCB, and other persistent contaminants have been identified in the tissues of Lake Ontario salmonids as well as forage fish (Reinke *et al.* 1972; Kaiser 1974; Haile 1975). High levels of mirex in fish prompted the New York State Department of Environmental Conservation to issue a ban on the possession of salmonids from 1977 to 1978 subsequently followed by public warning and guidelines.

In the western portion of Lake Ontario, the Niagara River has been identified as a prime source of contaminants (Niagara River Toxic Committee 1984). The Niagara River drains a highly industrialized section of Buffalo and western New York State. A significant portion of the dissolved contaminants it delivers to Lake Ontario originate from landfills and abandoned hazardous waste disposal sites. The Oswego River has been identified as a major source of mirex in eastern Lake Ontario (Martin 1982). There are also several large, uncontrolled abandoned hazardous waste sites on the southeastern shore of Lake Ontario. One of these, The Pollution Abatement Service Site, has been designated as the

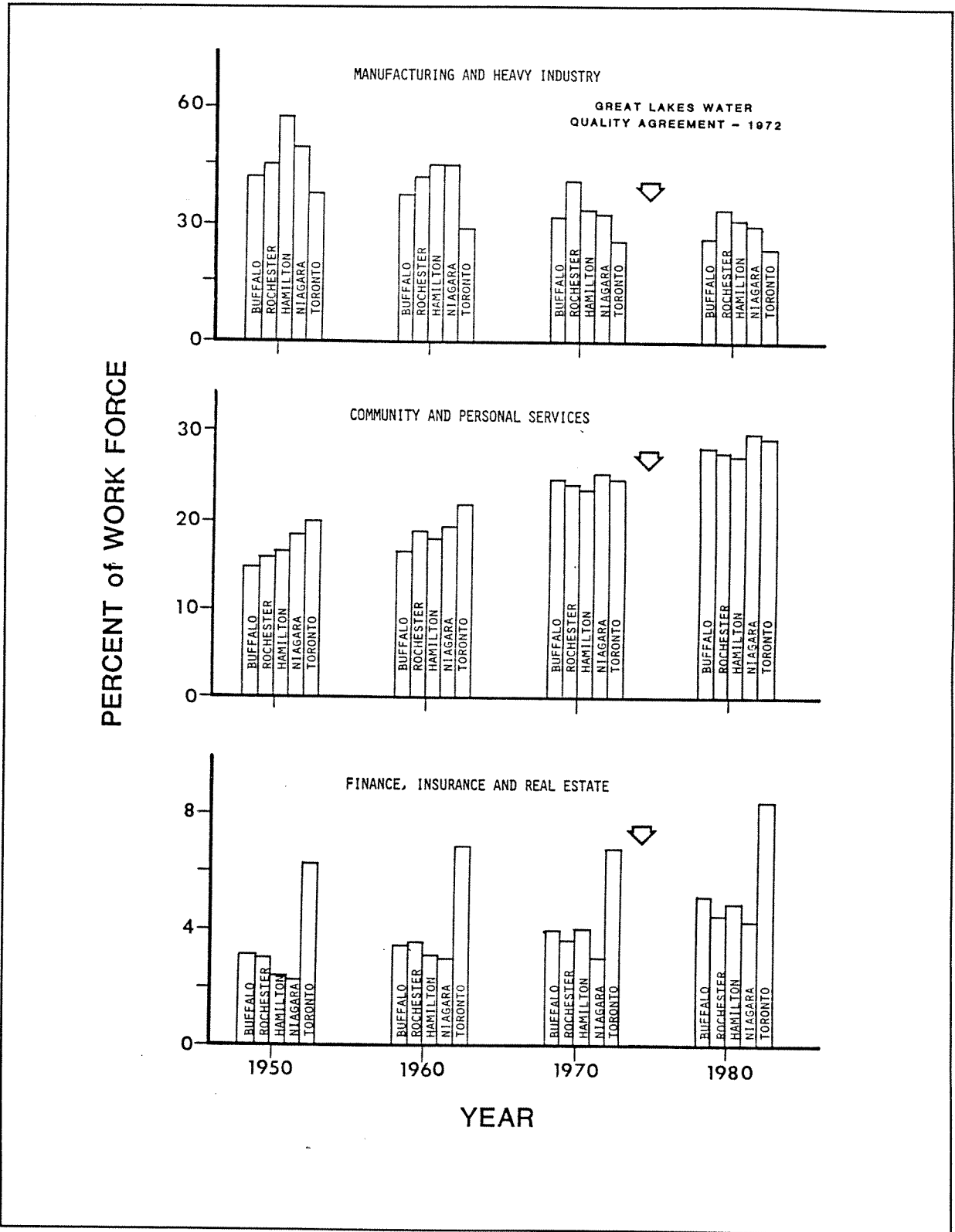


Figure 3. Industrial structure and work force distribution in major cities of the Lake Ontario Basin from 1950 to 1980. From Rogerson & Stack (1987).

number one Superfund site in New York State by the New York Department of Environmental Conservation. Soils from this site are contaminated with numerous toxic compounds to depths of up to 4.4 meters (Scrudato and Hinrichs 1982).

Clearly, hazardous waste disposal sites and other industrial locations around the shores and tributaries of Lake Ontario pose a serious threat to the quality of these surface waters. The fact that this Lake supplies significant sources drinking water translates into a potential threat to human health. The control and long-term assessment of the Lake Ontario environment is thus, crucial to the future of freshwater usage from this Lake.

The Great Lakes Water Quality Agreement

For centuries people from both the U.S. and Canada have used the Great Lakes for many purposes including drinking, boating, swimming, fishing, shipping, industrial processing, and disposing of wastes. Millions of persons now depend on these lakes for economic, recreational and aesthetic benefit. In recent decades it has been recognized that the water quality of the Great Lakes must be rehabilitated and/or maintained or all these benefits to society will be lost.

Both nations, Canada and the U.S., have recognized the importance of boundary waters, including the Great Lakes, for many years. In 1909 the two countries signed the Boundary Waters Treaty. It was written to help settle and prevent disputes regarding the use of boundary waters and to ensure open and free navigation. This treaty also included an article stating that waters flowing across the boundary shall not be polluted on either side to the injury of health or property on the other side. This treaty created the International Joint Commission (IJC) to assist the two countries in carrying out treaty obligations.

The Great Lakes Water Quality Agreement in 1972 was the outgrowth of an increasing binational recognition of shared water quality concerns which relate back to the 1909 treaty. Its origins can be traced largely to a 1964 reference to the International Joint Commission for a study of water quality conditions in the lower Great Lakes and the St. Lawrence River. The widely publicized plight of a "dying" Lake Erie played an important role in generating binational interest in such a study. The results of that study, released in 1970, suggested that the two governments: agree on the adoption of water quality objectives; develop programs for the reduction of phosphorus discharges; and agree on controls and/or compatible regulations on a number of pollutant sources.

Negotiations on an international agreement commenced soon thereafter, culminating in the signing of the Great Lakes Water Quality Agreement of 1972. The principal features of that Agreement included specification of water quality objectives, a schedule for reduction of phosphorus loadings, a stated policy of nondegradation, and two studies, one addressing water quality problems and needed remedial measures in the Upper Great Lakes and the other addressing pollution from land use activities. The signing of this agreement had a significant impact on water quality trends in Lake Ontario, as well as the other Great Lakes (Slater and Bangay 1980).

Consistent with provisions in the 1972 Agreement, a review and subsequent renegotiation took place several years later, culminating in the signing of the 1978 Agreement. The latter, while reaffirming the basic tenets put forth by its predecessor, established more comprehensive and stringent water quality objectives. The purpose of this new Agreement was to restore and maintain the chemical, physical and biological integrity of waters of the Great Lakes Basin Ecosystem. To achieve this purpose the parties agreed to develop programs,

practices and technology necessary for a better understanding of the Great Lakes Basin Ecosystem and to eliminate or reduce to the maximum extent practicable the discharge of pollutants into the Great Lakes System.

In 1987 several Amendments were added to the 1978 Water Quality Agreement and signed by both countries. In recognition of the exceptional quality of the 1978 Agreement, the purposes, policy and general objectives were not changed by the 1987 Amendments. Key Aspects of the 1987 Great Lakes Water Quality Agreement Amendments included the following.

1. Technical aspects of the Agreement have been brought up to date particularly with respect to: contaminated sediments, airborne toxic pollutants, contaminated groundwater and non-point sources of pollution. This will better support quantification of all sources of toxic substances entering the Lakes and determination of the reductions necessary to meet water quality objectives.

2. Accountability and management aspects of the Agreement clarify responsibilities and set dates for completion of various milestones and reports. Key features are the process for reviewing water quality objectives and preparation of remedial action plans for geographic areas of concern, and Lake management plans for critical pollutants.

The net effect of the amendments is to call for a clear definition of problems and a clearly defined series of actions that will result in solving them.



CHAPTER 3

LAKE ONTARIO BASIN DESCRIPTION

Lake Ontario is the last in the Great Lakes chain and the smallest in surface area (18,960 km²) and shoreline length (1,146 km). With a mean depth of 86.5 m, and a maximum depth of 244 m, however, it has the fourth largest volume of the Great Lakes, is second only to Lake Superior for greater percentage of cumulative area representing deeper water, and is among the fifteen largest lakes of the world (Herdendorf, 1982). In addition, of all the Great Lakes, Lake Ontario has the largest ratio of watershed land area to lake surface area (3.2) indicating a much larger relative drainage basin than the other lakes.

One state, New York, borders Lake Ontario to the south while one province, Ontario, borders it on the north (Figure 1). The industrial heartland of Canada and the northeastern U. S. extends from the shores of Lake Ontario westward forming an urban-industrial corridor including the cities of Toronto, Rochester, and Buffalo/Niagara Falls. Lake Ontario is also in close proximity to the Mohawk-Hudson Valley industrial and commerce corridor of New York State extending to New York City, a distance of about 250 miles. To the north, the unsettled Canadian wilderness is only 300 miles away. Lake Ontario, as a unique water resource, is strategically located to serve the multiple needs of the urban/rural region in which it is located.

Geologic Setting and Physiography

Major geologic events affecting the topography of the Lake Ontario basin have been the periods of preglacial erosion and glacial invasion following the close of the

Paleozoic Era. The Canadian Shield composed of precambrian rocks, occupies all but the southern lowlands of the Ontario Province, extending 100 miles north of Lake Ontario. Lake Ontario itself lies below the Niagara Escarpment and is bracketed between Lower Silurian shales on its southern shore and Ordovician limestones on its northern shore. Ordovician shales underlie the deeper parts of the lake.

Following the marine sedimentation of the Paleozoic Era, 300 million years of continuous subaerial erosion carved out the lowlands surrounding the southern border of the Canadian Shield and sculptured local physiographic features, many of which survived the glacial periods. During the last two million years the Lake Ontario region experienced four major glacial periods separated by long interglacial periods. The present Lake Ontario basin results from glacial scour during ice advance along the axes of a preglacial river valley. Areas near the Lake were covered with water following the glaciation period, resulting in beaches, wave-cut cliffs and deltas. At the higher elevations, the relief reflects the action of the ice, and the land forms are typical of a glaciated area with moraines, drumlins, eskers and till plains.

In the northeast portion of the basin the area is interlaced with lakes and frequent outcrops of the Precambrian Shield. This extends eastward to the Adirondack Plateau as an outlier of the Precambrian Shield. South from the Lake lowlands occur near the shore rising to the glaciated upland with its moraines and drumlins. Behind this is the Allegheny Plateau which forms the northern edge of the Appalachian formation. The plateau is

deeply indented by the Finger Lakes of New York State. The bedrock of Lake Ontario is composed principally of marine sedimentary rock deposited during the period between the Cambrian and Silurian ages. It is largely composed of shale and limestone. The present distribution on the Lake bed is the result of erosion of the previous glacial overburden and sedimentation during the past 10,000 years.

Drainage of the Lake Ontario basin is characterized by small streams draining the lowland areas which have their sources in the steeper slopes of the moraines. These lowlands are the most important areas around the Lake since the principal cities and agricultural areas are located on them near the Lake. The soils are generally sands, silts and clays near the Lake with loams and coarse materials in the moraines. Most of the land has been cleared for agriculture. The drainage of the uplands is by the larger river systems. Typical are the Trent River on the north shore and the Genesee and Oswego Rivers on the south shore. Tributary flow, excluding the

Niagara River, is divided almost equally between Ontario (434 m³/sec) and New York State (429 m³/sec) (International Joint Commission 1969). The Niagara River, the dominant inflow to the Lake, in contrast exhibits a flow rate of 5,700 m³/sec.

Basin Morphometry

The general bathymetry of Lake Ontario is depicted in Figure 4. The Ontario basin is an erosional feature, situated along an outcrop of erodible Ordovician shale bedrock. The north slope of the basin is quite shallow and featureless due to the proximity of an underlying layer of durable middle Ordovician limestone. The southern slope, not sharing this feature, is considerably steeper. This gives the Lake a relatively deep bottom contour, and as a result, a significant fraction of the bottom (47%) is classified as non-depositional. Unlike the Upper Great Lakes, the bottom topography of Lake Ontario is relatively smooth and undistinctive.

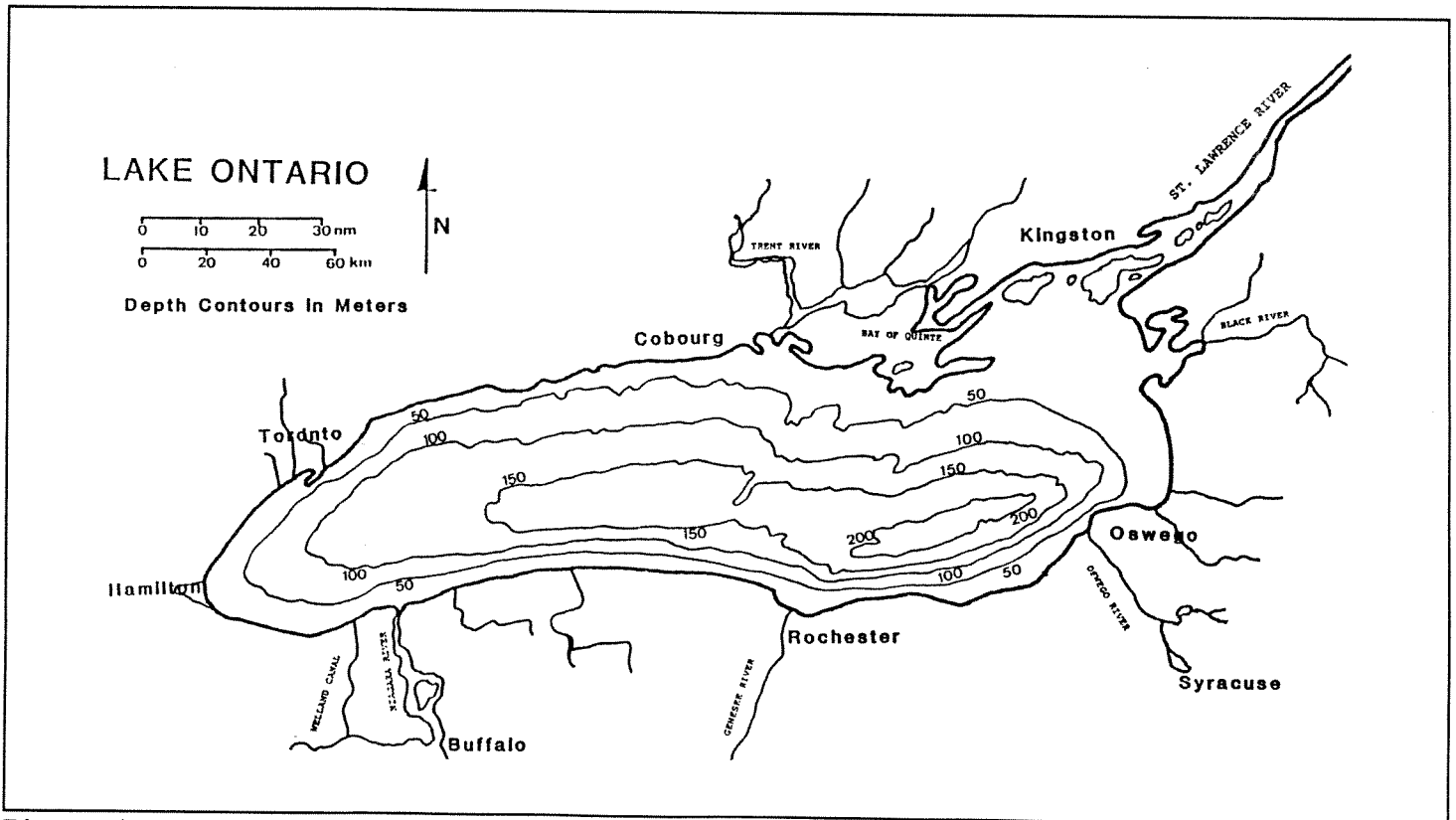


Figure 4. Lake Ontario bathymetry. Depth contours in meters.

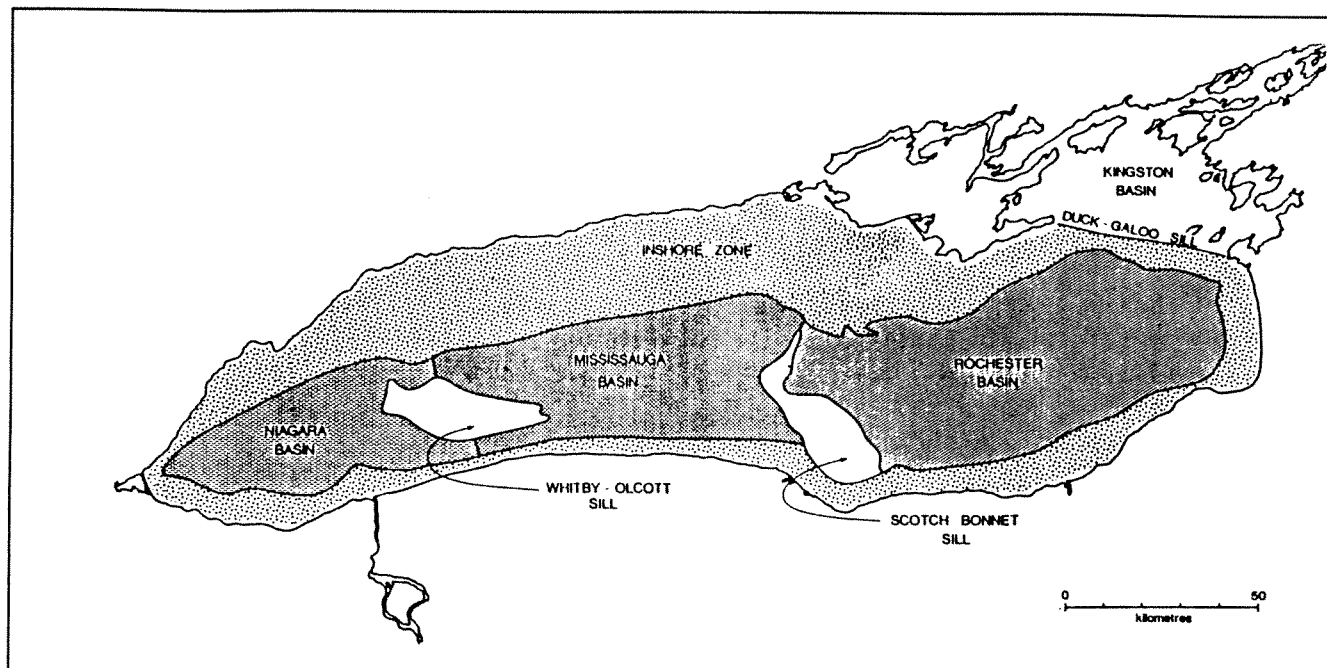


Figure 5. Basins of Sedimentation in Lake Ontario. From Thomas (1983).

Four relatively distinct sedimentation basins exist within the Lake which are, ranging from west to northeast, the Niagara, Mississauga, Rochester and Kingston basins (Figure 5; Thomas 1983). The differentiation between the first three basins is indeterminate, as illustrated by the bathymetric contours of Figure 4, and there is likely a considerable amount of interchange of resuspended material between them (see Charlton 1983; Sandilands and Mudroch 1983). The separation afforded by the Duck-Galoo Sill between the Rochester and Kingston basins is much more pronounced and, consequently, contributes to the Kingston basin, exhibiting water quality characteristics quite distinct from that of the main Lake.

Hydrography

Lake Ontario is a physically dynamic system, strongly influenced by meteorological events. Water circulation patterns are highly variable, being influenced by wind stress on surface waters, hydraulic flows from discharging tributaries, water stratification and

mixing and upwelling phenomena. Circulation patterns for Lake Ontario are described in greater detail in Simons and Schertzer (1985) and Simons, *et al.* (1985). Day-to-day changes in the thermal and current distribution result from variation in surface water either toward or away from coastal regions. Surface wind stress is generally weaker in the spring and increases to a maximum from October to February.

The generalized circulation pattern shows the flow from the Niagara River moving predominantly eastward along the south shore of the Lake in the summer. This is balanced by a westward flow on the north shore, thus setting up a lake-wide counterclockwise circulation pattern. In the winter, the strong eastward flow on the south shore is balanced by a return westward flow in the mid-lake region. Net flow in winter along the north shore is often negligible with both eastward and westward components usually counterbalancing each other. The south shore eastward flow in the Lake is usually stronger than the flow from the Niagara River.

Lake Ontario

After normal thermocline development in July, the volume of water transported in the coastal flow is greater, the current is quasi-geostrophic, and northwest shore upwelling events are common (Aubert 1978). Currents along the south shore prevail more to the east than do currents along the north shore, again influenced strongly by upwelling (Figure 6).

Simons, *et al.* (1985) computed net water transport in Lake Ontario to show that the eastward flow along the south shore was $70,000 \text{ m}^3/\text{second}$. Comparing this to the outflow to the St. Lawrence and observations of periodic westward flows from the Niagara, they concluded that more

than 90 percent of the inflowing water must be recirculated. With a mean speed of 5 km/day in the belt of the eastward flow and the length of the lake being approximately 300 km, it was suggested that the time scale for recirculation is a few months. This is indicative of a relatively short mixing time in the Lake which could ensure the distribution of an introduced persistent substance throughout the lake within a time frame of 1-2 years. Thomas (1983) and Simons, *et al.* (1985) have also correlated the distribution of sediment contaminants in the depositional basins with the water circulation patterns and demonstrated a consistency between circulation and materials transport.

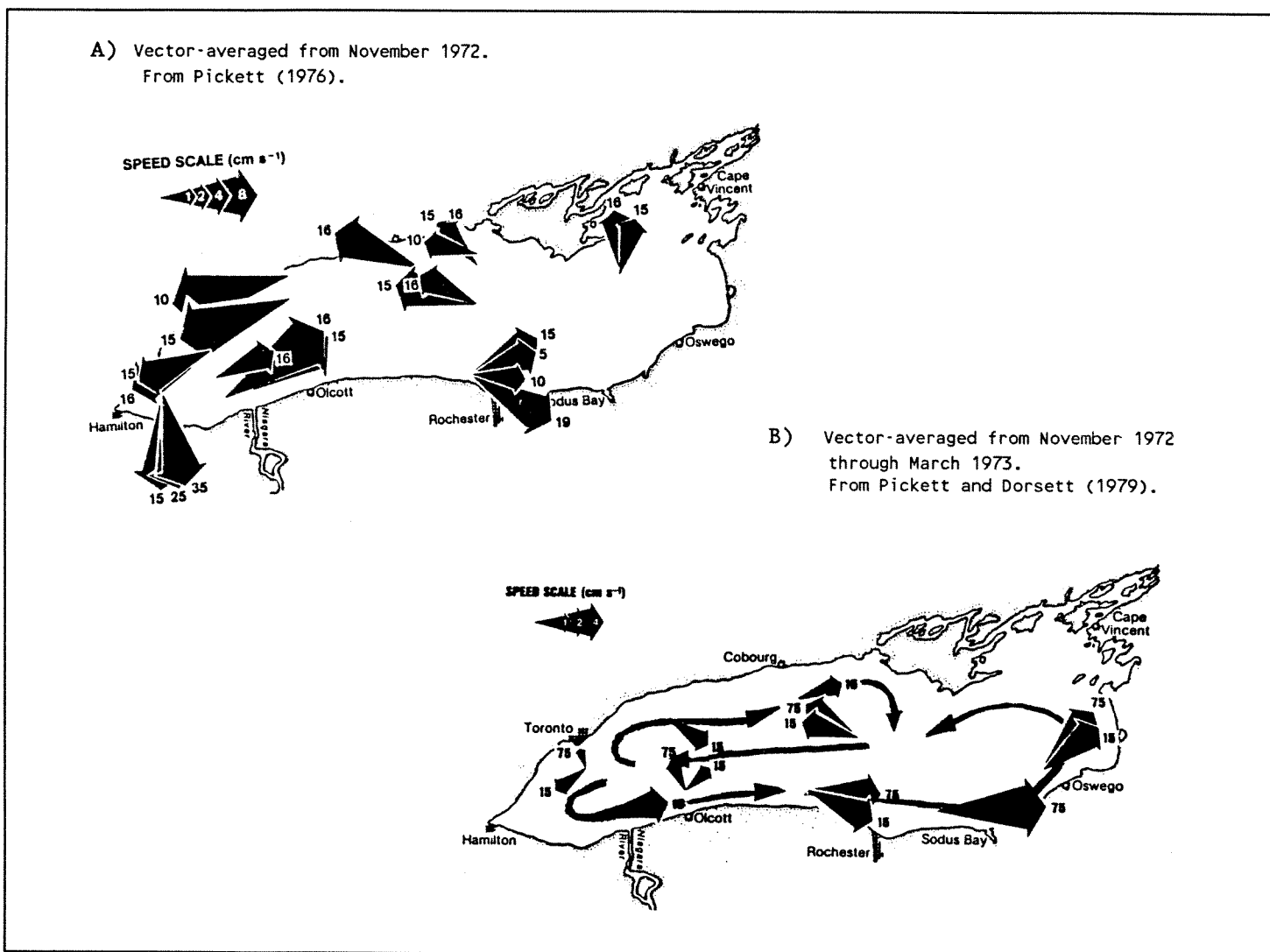


Figure 6. Lake Ontario observed currents (depths in meters at arrow tips).

The dominant inflow of water to Lake Ontario is from the Niagara River. The average flow out of Lake Erie into the Niagara River for the period 1900 to 1983 was 5,700 m³/second which is 85 percent of the average Lake Ontario outflow to the St. Lawrence of 6800 m³/second (Yee and Lloyd 1985). The five major tributaries to the Lake, the Trent, Oswego, Twelve Mile Creek, Black and Genesee Rivers with flows of 198, 189, 179, 117 and 79 m³/second, contribute an additional 11 percent (762 m³/second) of the outflow. Direct precipitation to the lake surface accounts for an additional 500 m³/second (50 year average), while evaporation represents an average annual loss of approximately 530 m³/second (Bruce and Rodgers 1962). Assuming a lake volume of 1640 km³ and an outflow of 6800 m³/second, the residence time of water in Lake Ontario, defined as the time required to displace all the water in the lake, is approximately 7.6 years.

Because of the physically dynamic nature of Lake Ontario, retention times of water masses and chemical gradients are strongly influenced by inshore-offshore exchanges, as regulated by thermal structure and currents.

In the nearshore region a thermal bar generally develops in the spring. This bar slowly migrates in an offshore direction and ultimately intersects with a mid-depth thermocline in early summer. Major physical perturbations of the thermal structure in the nearshore zone of Lake

Ontario are usually associated with upwelling events of variable duration and intensity. These events and the overall thermal structure of the lake strongly influence dynamics of lake biota. For example, thermal bar formation tends to structure the nearshore habitat so that potentially competing species can avoid excessive interaction through niche separation. In addition, the nearshore zone is potentially influenced by more prominent upwelling events in the late summer period when available nutrients have usually been depleted, increasing rates of primary productivity (Stadelmann and Frase 1974).

In light of these suggestions, the upwelling events of Lake Ontario were monitored (Haffner, *et al.* 1984a). It was concluded that such events did not result in sufficient mixing to recharge nutrient of coastal waters but rather served to displace nearshore waters to the middle of the Lake, which then returned to the coastal area at the end of each event. Rather than nutrient enrichment, these upwelling events were observed to cause species composition changes in the plankton and served as a mechanism to inoculate these species into the epilimnion of the offshore region. The above scenario demonstrates the need to understand the physical dynamics of an environment such as Lake Ontario if one wishes to assess the distribution and characteristics of flora and faunal communities and the dynamics of material transport.

Lake Ontario

CHAPTER 4

BIOLOGICAL CHARACTERISTICS OF LAKE ONTARIO

As a result of increasing nutrient enrichment of Lake Ontario, a doubling of the mean annual biomass of algae at the Toronto water intake was observed over the period 1923 to 1954 with a shift in dominant genera similar to that of Lake Erie (Shenk and Thompson 1965). More recent data on phytoplankton indicator species suggest that the status of the lake is changing from meso-eutrophic to meso-oligotrophic, which would be compatible with the decreases in phosphorus loadings (International Joint Commission 1987). The present zooplankton community structure is thought to be indicative of mesotrophic to oligotrophic conditions (Makarewicz 1985). Benthic invertebrate community structures have also changed through time.

A substantial transition has been observed in the fish communities of the lake over the last 60-80 years (Beeton 1969; Loftus and Regier 1972; Christie 1974). The native forage and top predator species (e.g. Atlantic salmon, lake trout and blue pike) have been eliminated or dramatically reduced through a complex interaction of habitat alteration, pollution, overfishing and the introduction of exotic species, most notably the sea lamprey and alewife. Today, populations of top predator fish (lake trout and exotic Pacific salmon) are being maintained and increased yearly through an extensive hatchery rearing and stocking program.

In order to adequately relate changes in environmental quality of the Ontario Basin with changes in the biologic communities, as briefly summarized above, it is important to characterize the long-term trends in these communities with whatever data may be available.

Phytoplankton

The most extensive evaluation of Lake Ontario phytoplankton occurred during the International Field Year of the Great Lakes (IFYGL) investigations. During these studies it was observed that there were large variations in both temporal and spatial scales with respect to the distribution of phytoplankton (Christie and Thomas 1981: p. 328). The western and northwestern end of the Lake showed the highest number of species occurrences for all lake areas and the greatest abundances were observed in waters adjacent to Toronto. The mid-lake region exhibited an east-west differentiation in phytoplankton species composition, moving away from the more urbanized regions of the basin. The southeastern and northeastern nearshore areas exhibited high phytoplankton abundances and lower diversity within the species assemblages.

The Lake Ontario flora was found to be most similar to more eutrophic regions of the other Great Lakes (Thomas, et al. 1978). Species characteristic of meso-trophic to eutrophic waters were among the plankton dominants. Diatom groups such as the *Cyclotella* association were rarely encountered while eutrophic indicators such as members of the genus *Stephanodiscus* were quite abundant (Lorefice and Munawar 1974).

Seasonal succession of phytoplankton species groups was identified during the IFYGL investigations (Stoermer, et al. 1975). In the spring, species abundance was greatest in the warmer nearshore waters. The phytoplankton group Bacillanophyta dominated during this period, while microflagellates also exhibited high abundances. As the Lake

stratified during mid-summer, abundances in the mid-lake region increased. Species from the Chlorophyta group dominated during this period. The phytoplankton group Cyanophyta reached its maximum abundance in Lake Ontario in the fall and dominated species assemblages during this period. Bacillanophyta species again became abundant in mid-winter, leading to their dominance of species assemblages in the spring.

Munawar, et al. (1974) and Stoermer, et al. (1975) observed that highest phytoplankton abundances usually occurred between depths of 1-10 m, with peak abundances at 10 m, reflecting both nutrient supply rate and water transparency conditions. Genera such as *Dinobryon*, *Oscillatoria*, and *Gymnodinium* tended to be found at discrete depths while other genera such as *Cryptomonas* and *Rhodomonas* tended to be more uniformly distributed throughout the water column (Haffner, et al. 1984b). The discrete depth distribution of some of these genera, compared with weak thermal stratification, suggested depth selection and habitat preference within the algal community.

In the lake-wide IFYGL survey there were differences observed in patterns between nearshore and offshore phytoplankton communities (Stoermer, et al. 1975). Similar assemblages were noted in the early spring (Figure 7). Differences began appearing in community structure during the late spring and continued into the early fall. It was suggested that the development and excursion of the spring thermal bar was the major factor controlling seasonal succession and distribution patterns for the phytoplankton in the Lake. Most of the species characteristic of the inshore waters followed the excursion of the thermal bar offshore (Thomas, et al. 1978). At the mid-lake station 25 percent of the species were diatoms as compared to 18 percent at the nearshore site. In contrast, more Cryptomonadinae were found inshore than offshore.

There were also differences in phytoplankton biomass and chlorophyll pigments (Figure 7). At the nearshore station biomass ranged from 0.2 - 2.3 g/m³ and chlorophyll varied from 1.9 - 8.0 mg/m³. At the mid-lake site biomass ranged from 0.2 - 1.6 g/m³ and chlorophyll from 1.9 - 7.7 mg/m³. Annual primary production rates were observed to be 50 percent greater at the nearshore site than at the mid-lake station. In general the peaks in biomass, chlorophyll pigments, and production rates always occurred earlier in the nearshore region (May-June) than at the offshore site (August-September). These spatial differences were probably related to higher average temperatures first occurring in the shallower waters and eventually moving offshore with the migration of the thermal bar (Thomas, et al. 1978).

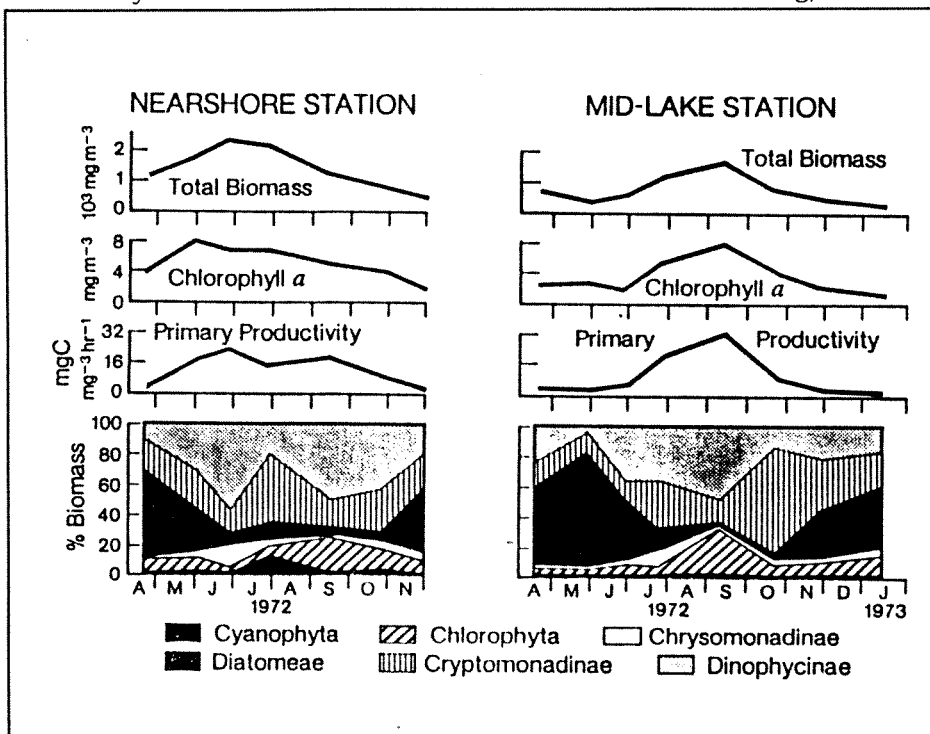


Figure 7. Seasonal changes in the composition of phytoplankton samples at two Lake Ontario stations in 1972. From Munawar (1980).

Primary production values on a temporal basis in Lake Ontario were low in the winter, reached a maximum in spring, declined during the summer, and increased slightly in the fall (Glooschenko, et al. 1974). Rate of increase in spring production for nearshore waters (< 20 m depth) was greater than in offshore, deeper waters (Figure 7). Over an annual cycle highest mean primary production usually occurred in the western end of the Lake, the southern nearshore waters, and in the region of the Bay of Quinte. During most periods of the year the depth of maximum photosynthesis was usually between 3-5 m, probably due to photoinhibition above that depth and light limitation below. Maximum mean in situ production was measured in the spring at a rate of 26 mg C/m³/hr. In contrast, winter rates ranged around 2 mg C/m²/hr for the total water column.

Based upon all measurements between January and December 1972 (Glooschenko, et al. 1974), it was estimated that annual phytoplankton production equaled 190 g C/m². In addition, Stadelmann and Munawar (1974) calculated annual phytoplankton production for a nearshore and mid-lake station and estimated values of 270 g C/m²/yr and 185 g C/m²/yr, respectively for the two separate areas. They concluded that these values were indicative of mesotrophic to secondary eutrophic conditions, as proposed by Vollenweider, et al. (1974).

More recently Lean, et al. (1987) made measurements of phytoplankton photosynthesis at a mid-lake station in Lake Ontario and concluded that primary production has declined to the point that zooplankton grazing rates appear to exceed carbon produced through photosynthesis in the Lake. For a period of 6 months in 1982, measurements of total water column photosynthesis (Lean, et al. 1987) indicated a total carbon production rate of 69.6 g/m² (June-Nov). If one assumed an additional 15 g C/m² for May (Lean, et al.

1987) and approximately 4.44 g C/m² for the winter months of December through early April (Munawar and Munawar 1986), then total Lake Ontario production equalled 90 g C/m² yr in 1982.

Since the early 1970s there have been some notable changes in the Lake Ontario phytoplankton community (Table 2). The offshore region of the Lake has shown a great deal of variability over this period with minor decreases in the Bacillariophyceae and Chrysophyceae. Similar small increases were seen for Cryptophyceae and Dinophyceae. In the nearshore region a significant increase was observed for Dinophyceae and a corresponding significant decrease seen for the Cyanophyceae. These changes, especially for the shallow-water communities, were probably the result of decreasing phosphorous loading to the system between 1970 and 1982.

In many regions of Lake Ontario, decreases in total algal biomass were also observed between 1970 and 1982 (Table 3). These changes were most obvious for the summer means at all locations. Annual means, however, with the exception of the Kingston basin, also showed general decreasing trends. In addition, chlorophyll a concentrations between 1967 and 1981 (Dobson 1985) served to indicate further long-term changes in the Ontario basin. Peak concentrations during a high year (1970) were greater than 10 ug/L (Figure 8a). In contrast, from 1977 forward, peak concentrations for any year never exceeded 7 ug/L. Most years the annual average was less than 4 ug/L chlorophyll a. Secchi depth transparency correlated well with the patterns exhibited for chlorophyll (Figure 8b). Mean summer transparency at mid-lake sites was lowest during the early 1970s and progressively increased through 1980. Maximum secchi depth occurred in 1977 corresponding to some of the lowest chlorophyll measures of the 1970s.

Lake Ontario

Table 2. Average annual percentage composition of algal classes in nearshore and mid-lake areas of Lake Ontario (from Stevens 1987).

ALGAL GROUP	YEAR			
	1970	1972	1981	1982
<u>NEARSHORE</u>				
Bacillariophyceae	36.9	24.4	26.8	32.1
Cryptophyceae	21.2	22.8	26.7	26.7
Dinophyceae	2.6	29.7	24.0	19.2
Chrysophyceae	7.8	5.8	6.1	5.5
Chlorophyceae	15.0	12.5	13.5	14.0
Cyanophyceae	16.2	4.2	3.3	2.4
<u>MID-LAKE</u>				
Bacillariophyceae	35.7	35.4	23.9	25.6
Cryptophyceae	16.9	25.5	36.8	32.5
Dinophyceae	7.5	23.4	24.0	17.0
Chrysophyceae	12.6	4.2	4.2	6.9
Chlorophyceae	18.7	9.8	8.7	15.6
Cyanophyceae	9.2	1.6	2.1	3.2

Table 3. Changes in Lake Ontario phytoplankton biomass between 1970 and 1982. Data from Stevens (1987).

LOCATION	YEAR	PHYTOPLANKTON BIOMASS (g/m ³)	
		ANNUAL MEAN	SUMMER MEAN
MIDLAKE STATION	1970	1.80	1.80
	1977	0.88	1.01
	1981	1.15	0.55
	1982 ^a	0.84	0.58
	1982 ^b	0.93	0.74
WESTERN BASIN STATION	1970	1.57	3.21
	1975	1.14	1.42
	1977	0.97	1.17
	1978	-	1.07
	1981	1.37	1.17
	1982	0.97	0.99
KINGSTON BASIN	1970	1.40	2.98
	1975	1.32	1.39
	1978	1.65	0.96
	1981	1.67	0.97
	1982	1.73	0.97

^a Data from LONAS study of Lake Ontario (Gray 1987)

^b Data from Fisheries & Oceans Canada Bioindex station sampling.

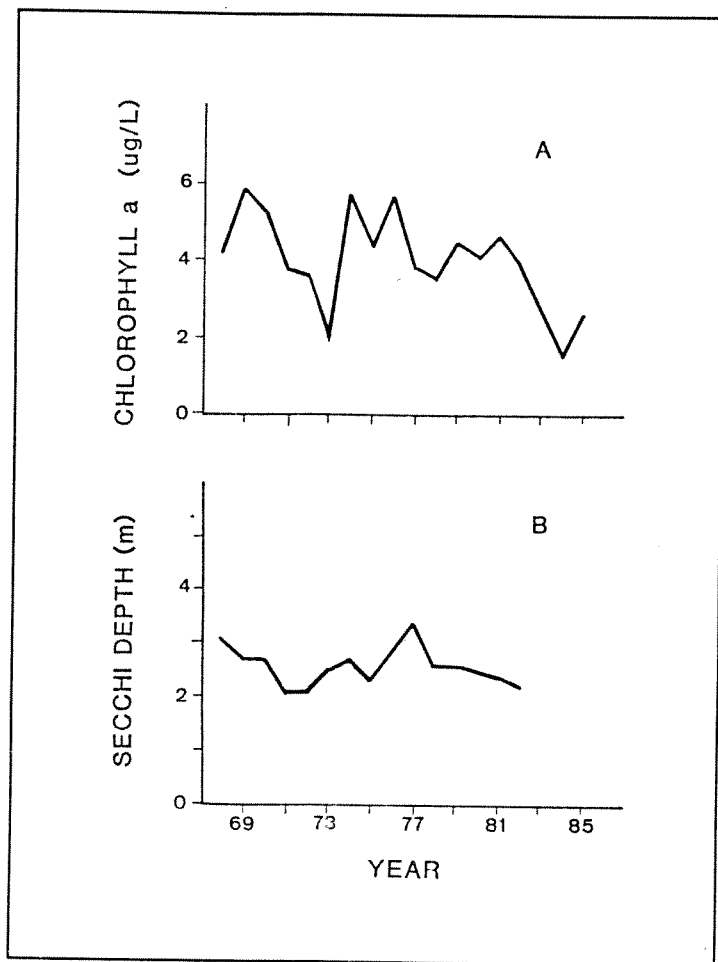


Figure 8. Chlorophyll *a* patterns (A) and secchi disc depth trends (B) in offshore, near-surface waters of Lake Ontario. From Dobson (1985).

From a detailed study of a northern nearshore station and a mid-lake station (Gray 1987), differences between nearshore and offshore phytoplankton biomass and composition were evident in Lake Ontario in 1982. Phytoplankton biomass was characterized by multiple peaks which ranged over three orders of magnitude. At the northshore station phytoplankton biomass ranged from 0.09 to 9.00 g/m³. In contrast, biomass ranged from 0.10 to 2.40 g/m³ at the mid-lake station. Bacillariophyceae was the dominant group at the northshore station until September when Cyanophyta contributed most to the biomass (83%). Although Bacillariophyceae was the dominant component of the spring phytoplankton community at the mid-lake station, phytoflagellates (49%) and

Chlorophyceae (25%) were responsible for summer biomass, with the Chlorophyceae expanding to 80 percent in the fall. The seasonal pattern of epilimnetic chlorophyll *a* correlated with temperature.

Phytoplankton differences were also observed in comparing the 1982 data with the earlier 1970 and 1972 data. While chlorophyll *a* concentrations for 1982 fell within the range of values from 1970 and 1972, algal biomass declined from a mean of 1.36 and 0.72 g/m³, respectively for nearshore and mid-lake stations in 1972 (Munawar, *et al.* 1974), to means of 0.57 and 0.41 g/m³, respectively in 1982 (Gray 1987). In addition, a number of eutrophic species (e.g., *Melosira binderana*, *Stephanodiscus tenuis*, *S. hantzschii* var. *pusilla*, and *S. alpinus*) previously found in the early 1970s were absent in 1982.

Bacterial relationships with algal biomass were investigated in Lake Ontario by Rao, *et al.* (1979). During the spring and summer low bacterial biomass and high chlorophyll pigment levels were observed. The dominant algal group during this period was diatoms. It was suggested that the silica cell wall of diatoms was resistant to bacterial attack, thus limiting the substrates available for bacterial colonization. From May through September a high bacteria, low chlorophyll pigment relationship was noted in Lake Ontario nearshore waters. In contrast to the spring, this period was characterized by large amounts of non-diatomous algae which were available for bacterial colonization after the summer blooms of *Chlorophyta* genera. Besides the apparent relationship between bacteria and the species composition of the algal community, Rao, *et al.* (1979) observed that bacterial biomass was also significantly correlated with particulate organic carbon and nitrogen of the water column, and water temperature.

It should be reemphasized that physical processes cannot be ignored in determining factors controlling the phytoplankton dynamics described above. The importance of physical, environmental

factors is often overshadowed by considerations of nutrient limitation and its influence on phytoplankton communities (Haffner, et al. 1984a). Physical processes, however, probably play a major role in the timing of pulses and spatial distribution patterns observed for Lake Ontario plankton. The role of physical processes and their effects on algal production in Lake Ontario are thought to be significant. For example, Gray (1987) noted that physical events promote nearshore and offshore differences. Influential physical processes included boundary currents that persist throughout the growing season, intensified by a few summer storms which generally result in upwellings along the north shore of Lake Ontario.

Cladophora

The sessile attached macroalga, *Cladophora glomerata*, is abundant throughout 4 of the 5 Great Lakes. The most comprehensive information on this alga's distribution is available for Lake Ontario from IFYGL investigations. As early as the 1930s *Cladophora* growth was noted as causing problems in the western end of Lake Ontario near the Toronto area. By 1958 virtually all suitable substrate in most of Lake Ontario contained growths of this macroalga. Between 1960 and 1972 additional changes were noted in distribution and density. In 1960 heavy growths of *Cladophora* populated all suitable substrates in the western end of the Lake, with minimal growth in other littoral areas. By 1972 the eastern end of the Lake exhibited as heavy or heavier accumulation of this alga on hard substrates. This green alga forms extensive mats on hard bottoms in shallow waters which periodically break off and litter beaches and other public areas. This problem is of major significance around the shorelines of Lake Ontario and represents extensive economic losses to recreation related businesses.

Cladophora overwinters as a prostrate thallus or as short basal portions of the upright filaments in rock crevices. Growth is renewed each spring at approximately 5° C water temperature and reaches greatest production rates at 18° C (Bellis and McLarty 1967). The maximum rate of photosynthesis occurs between 15 - 20 °C with medium light intensity, but rates also increase during the summer period with medium to full sunlight (Herbst 1969).

Cladophora filaments serve as a substrate in shallow waters for various species of benthic diatoms. Protozoa such as *Verticella* and *Campanella* also colonize the filaments. The macroalgal filaments provide shelter for such invertebrates as gammarid amphipods, cladocerans, rotifers, molluscs, and young crayfish. This community provides an important food assemblage for organisms that feed in the shallow sublittoral zone of Lake Ontario (Herbst, 1969). Mallards and black ducks are the most well known feeders upon the *Cladophora* filaments directly.

Spatial and temporal distributions of *Cladophora* in the Lake Ontario environment are considerable and constantly changing as influenced by water quality (Shear and Konasewich 1975). Average biomass from samples collected at specific depths in Lake Ontario generally indicated that production decreased with depth. Standing stocks ranged from 224 g (d.w.)/m² at 1.5 m water depth to 100 g (d.w.)/m² at 5 m water depth. Temporally, average biomass for a number of study sites ranged from 158 g (d.w.)/m² in June, 224 g (d.w.)/m² in July and 67 g (d.w.)/m² in August. Along the U.S. shoreline of Lake Ontario, for a 350 m wide area between the Niagara River and Rochester, 66 percent of the nearshore zone was covered by *Cladophora* mats at a standing stock of 1.6 x 10⁴ kg (d.w.)/km during the mid-1970s (Thomas, et al. 1978). East of Rochester 79 percent of the nearshore region supported *Cladophora* mats and standing stock was 33 x 10³ metric tons (d.w.). These estimates were determined for the peak period of biomass for this macroalga (Shear and Konasewich 1975).

Phosphorus and nitrogen content of *Cladophora glomerata* appear to increase with depth of the algal growth. This macroalga also increases its concentration of phosphorus and nitrogen as the growth season progresses. On the average, *Cladophora* dry weight biomass during peak growth represents approximately 45 percent carbon, 0.4 percent phosphorus, and 2 percent nitrogen (Kinney, et al. 1976).

In the nearshore region of Lake Ontario, *Cladophora* represents a good indicator of nutrient changes that have occurred in this system. The best gauge of this response comes from studies conducted on the Canadian shoreline. Painter and Kamaitis (1987) reported that substantial reductions in standing crop (as dry weight) between 1972 and 1982/83 were observed at all their study sites, except Oakville, with the largest decline being at Eurenic Pt., on the south shore of Amherst Island in the Kingston Basin (Figure 9). Standing

dry weight, averaged on a lakewide basis, was 205.8, 80.0 and 85.9 g/m² for 1972, 1982 and 1983, respectively, or, in other words, a 58 percent decrease. The 1982 and 1983 data compared well even though the 1982 data was based on only one sampling date at the seven sites. Comparison of the standing crop between stations indicated that no definite geographical trend for *Cladophora* biomass along the Lake Ontario shoreline was apparent.

A more reliable measurement with which to evaluate yearly and station-to-station differences is tissue phosphorus levels because it exhibits less weekly variance than does standing crop. Tissue phosphorus levels have been substantially reduced between 1972 and 1983 (Painter and Kamaitis 1987). The 1972 lakewide average tissue phosphorus was 0.49 percent (AFDW) and the 1983 lakewide average was 0.20 percent (AFDW), a 29 percent reduction over the eleven year interval.

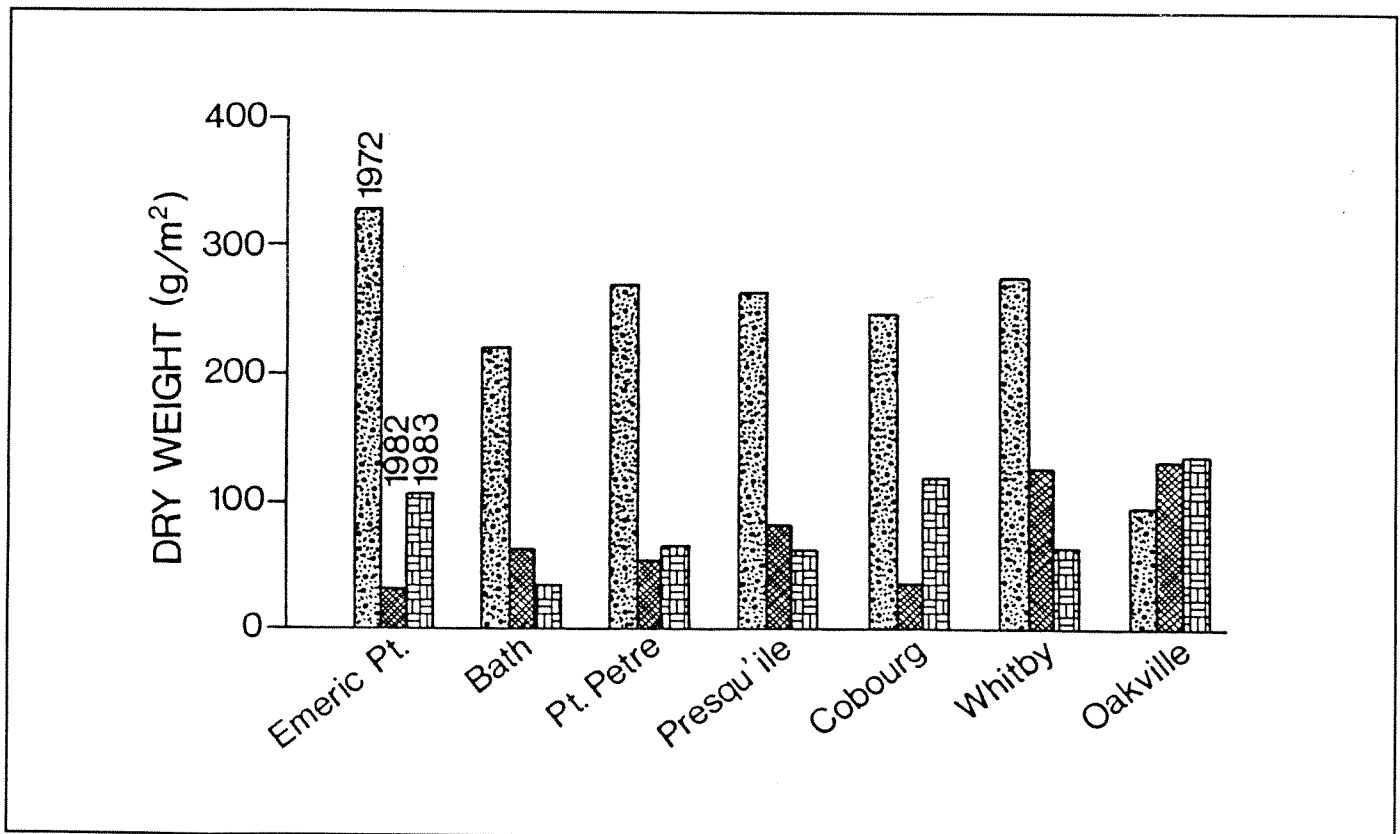


Figure 9. *Cladophora* biomass in 1972, 1982, and 1983 at each sampling station in Lake Ontario indicated. From Painter and Kamaitis (1987).

Zooplankton

The most extensive lakewide account for zooplankton distributions were presented from the IFYGL cruises of 1972-73 (Christie and Thomas 1981). Eutrophic indicator species were found to be most abundant in the vicinity of urbanized areas of the Lake such as Toronto, off the Oswego River, and in the Bay of Quinte. Fewer species of crustaceans were present in lake waters adjacent to urbanized areas of the basin than in Lake waters nearer rural basin areas. The cladocerans were more abundant in the waters not impacted by urban areas and included such genera as *Daphnia*, *Ceriodaphnia*, and *Chydorus*. In contrast, *Bosmina longirostris*, a species often associated with highly eutrophic conditions, was found in densities as high as 300,000/m³ off urban areas of Lake Ontario. With respect to long-term cultural eutrophication impacts on zooplankton communities in Lake Ontario, McNaught (1975) concluded from IFYGL sampling that a shift in species dominance had occurred over the last 30 years. There appeared to be a decrease in abundances of *Diaptomous* and *Daphnia* and a significant increase in *Bosmina* and *Cyclops* genera.

Watson and Carpenter (1974) evaluated the overall distribution of zooplankton in Lake Ontario by sampling 32 stations lakewide. The results of these studies showed that the peak abundance of crustacean zooplankton occurred in late August at concentrations of 60,000 - 70,000 individuals/m³. From January to May zooplankton numbers rarely exceeded 3,000/m³. A strong relationship was observed between crustacean zooplankton abundance and average water temperature for the top 40 m of the water column. Cycloids, copepods, and cladocerans were the most abundant crustaceans observed during the 32 station survey (Christie and Thomas 1981).

A north-south gradient in zooplankton abundance was often observed and thought to again be related to spatial variations in water temperature across the Lake (Taylor, et al. 1987). The horizontal differences

were thought to be caused by occasional upwelling that was influenced by wind stress and thermal stratification. Consequently, greater abundances of zooplankton were associated with warmer waters. Whether this was caused by displacement from upwelling (Aubert 1978) or simple zooplankton selection of a warmer environment, was not addressed by Watson and Carpenter (1974).

In 1982 a north-south transect was monitored for zooplankton dynamics and reported by Talor, et al. (1987). From this work it was observed that the seasonal pattern of zooplankton succession in Lake Ontario was similar at three stations along a north-south transect, with differences attributable to physical factors. Because of warming within the thermal bar zone, the spring succession at the northshore station was early relative to the mid-lake station, but after stratification the northshore station lagged behind due to frequent upwelling events. Thermal bar formation along the south shore was followed by downwelling throughout the summer, giving a deeper epilimnion. These events may have contributed to not only the greater biomass at this location, but also succession which preceded that at the other stations during summer and fall.

In 1970 Watson and Carpenter (1974) also evaluated seasonal changes in crustacean zooplankton biomass. They observed that mean biomass ranged from 10 mg/m³ to 28 mg/m³ between January and June. During July increases in community mean biomass were noted and peaks occurred at greater than 150 mg/m³ in August. During September and October zooplankton mean biomass remained above 100 mg/m³ and then declined to winter values of less than 20 mg/m³ in December. On a spatial dimension zooplankton biomass increased much earlier in waters shallower than 30 m depth. Greatest abundances for these nearshore waters occurred between March and May with a secondary peak in August and September. The mid-lake area, in contrast, only exhibited the August-September peak in biomass. The earlier peaks in biomass in the shallower waters were linked to

increases in chlorophyll pigments and phytoplankton blooms during the spring in Lake Ontario.

Borgmann, *et al.* (1984) observed that there was much greater seasonal variation in biomass of cladocerans than in biomass of cyclopoid copepods. Cladocerans were recorded to be low in biomass through May and then showed steady increases in standing stock, to their maximum August peak. Cyclopoids rarely varied from their mean standing stock by more than 15 percent throughout the year. In general biomass was dominated by cladocerans throughout much of the growing season while copepods were more important early in the year (Culver, *et al.* 1985), when the cladoceran standing stocks were low.

According to the comparison of numerous study results, zooplankton community structure from 1981 to 1985 appeared similar to that observed in the late 1960s and early 1970s (Johannsson 1987), the timing of summer succession was similar between 1970 and 1982 (Taylor, *et al.* 1987) and the community in 1982 was

characteristic of populations undergoing intense size-selective fish predation (Taylor, *et al.* 1987). No shift has occurred in the relative dominance of the common species since 1967, with the cladocerans *Bosmina longirostris*, *Daphnia retrocurva*, *Ceriodaphnia lacustris*, and *Eubosmina coregoni* and the cyclopoid: *Diacyclopops thomasi* and *Trophocyclops prasinus mexicanus* all contributing greater than one percent of the seasonally-weighted mean biomass (Johannsson 1987). Average summer zooplankton abundance, on the other hand, appeared to vary greatly between 1967 and 1985. Minimum levels were observed in 1970 and 1972 while maximum levels of greater than $3 \times 10^6/m^2$ were found in 1967 and 1982 (Figure 10). Abundance in 1981, 1983, and 1984 were intermediate, between $10-20 \times 10^5$ individuals/ m^2 . Although significant differences in total abundance occurred between years, all earlier studies fell somewhere within the range of zooplankton abundances observed between 1981 and 1984. Therefore, one must conclude that there has been no significant change in zooplankton abundance between 1967 and 1984 in response to other system

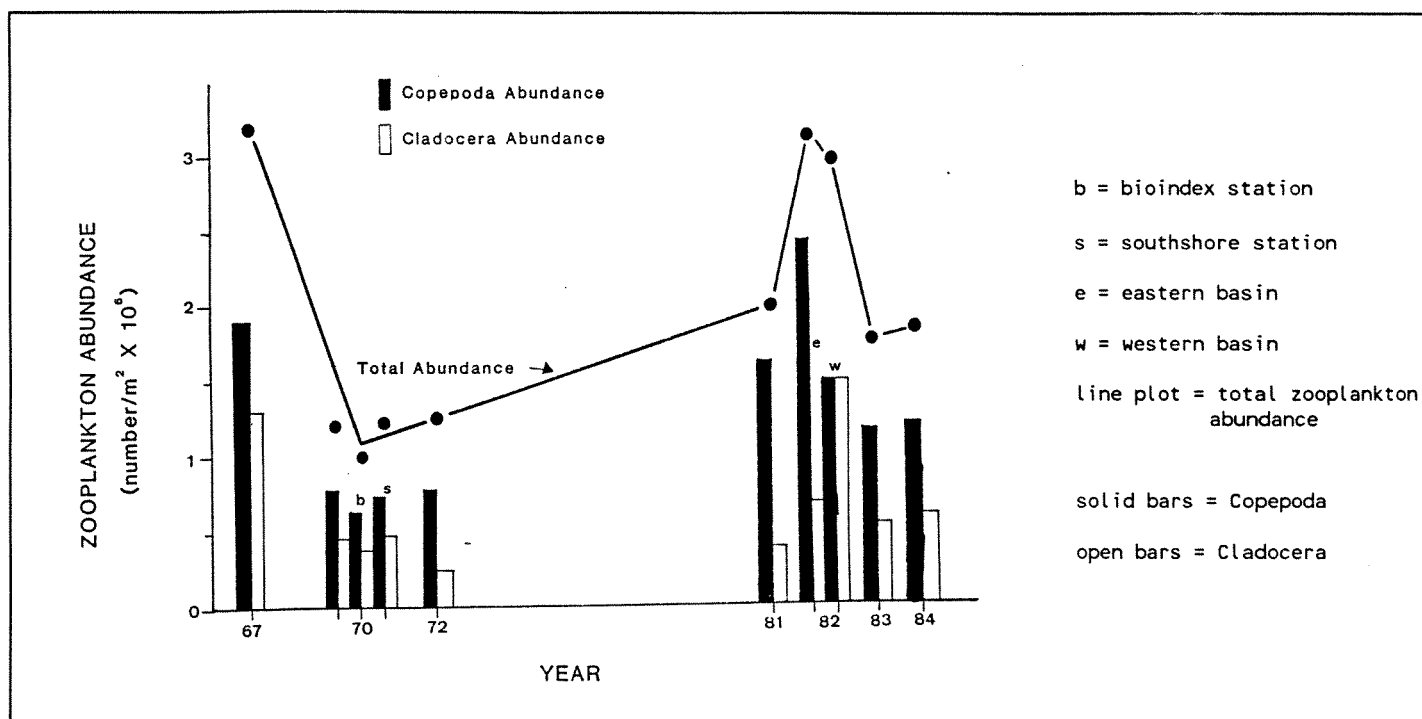


Figure 10. Average June to October Lake Ontario zooplankton abundances from 1967 to 1984. From Johannsson (1987).

changes such as nutrient loading and stocking of salmonid predators (Taylor, et al. 1987). In addition, a comparison of the structure of the cladoceran and copepod communities between 1967 and 1985 (Johannsson 1987), indicated that there has been a continuum of community structure with few exceptions between these 18 years.

Production estimates for the most abundant zooplankton species observed in Lake Ontario were calculated by Borgmann, et al. (1984). The species used to calculate production each represented more than one percent of total zooplankton standing stock and included *Bosmina longirostris*, *Ceriodaphnia lacustris*, *Daphnia retrocurva*, *Eubosmina coregoni*, *Diacyclops thomasi*, and *Tropocyclops prasinus mexicanus*. The greatest individual production was observed for *D. thomasi* at 5.5 g (d.w.)/m²/yr, while outside of the calanoid copepods, the least production was calculated for *C. lacustris* at 0.96 g (d.w.)/m²/yr. For all dominant species in Lake Ontario, Borgmann, et al. (1984) estimated that annual zooplankton production equaled 17.4 g (d.w.)/m². These estimates were for epilimnion production only and did not include production of coldwater zooplankton such as *Limnocalanus macrurus* or the larger decapod crustacean *Mysis relicta*, which spend much of their time deeper in the water column.

The vertical movements within the water column of the freshwater opossum shrimp, *Mysis relicta*, the dominant biomass producer within the zooplankton community, are well known (Beeton and Bowers 1982). Horizontal migration of these populations are also extensively documented (Morgan and Threlkeld 1982). Carpenter, et al. (1974) recorded that mysids in Lake Ontario were rarely found in nearshore waters less than 25 m deep, with maximum population numbers observed at offshore stations, greater than 100 m deep. They also observed that offshore peak abundances occurred in the summer with dispersement of populations to shallower waters in the spring and fall.

During IFYGL cruises in Lake Ontario, epibenthic and plankton sled tow collections indicated that mysids were most dense between the water depths of 70 - 150 m (R. O'Gorman, U.S. Fish and Wildlife Serv., Oswego, NY, written communication, January 1985). Peak population densities were recorded near the bottom sediments during daylight sampling. Vertical Bongo net tows at 10 m depth intervals during darkness indicated peak mysid densities within the water column and only small quantities of the shrimp near the sediments, where they had been dense during daylight. Decreases in light intensity to between 10 - 150 lx were thought to stimulate the vertical migration of mysids at dusk (Teraguchi, et al. 1975), causing the opossum shrimp to swim well into the water column, often to the depth of the thermocline (Hakala 1978). Temperatures above 14°C were thought to inhibit mysids from spending much of their nighttime migration period in epilimnion waters.

During the IFYGL program, mysid populations in the deep waters of Lake Ontario were observed to represent the greatest biomass component of any lower trophic level fauna taken in net or sled tows, either in the water column or near the bottom sediments (R. O'Gorman, U.S. FWS, Oswego, NY, written communication, January 1985). More recently Johannsson (Canada Center for Inland Waters (CCIW), Burlington, Ontario, written communication, July 1985) observed an average density of 420/m² for mysids at a 125 m deep station in Lake Ontario.

It is thought that the annual production of mysid biomass in Lake Ontario is the result of two overlapping populations. These populations produce two hatches a year, one in April and one in August (O. Johannsson, CCIW, Burlington, Ontario, written communication, July 1985). The April hatch represents offspring from adults hatched 19 months earlier in August and adults hatched 24 months earlier in April (second brood). The August hatch of the same year represents offspring from adults hatched in April, 16 months earlier,

and adults hatched 24 months earlier in August (second brood). This pattern agrees with Carpenter, et al. (1974) for observed growth rates of mysids in Lake Ontario. Based upon two reproduction periods during the year, Johannsson estimated production of mysids in Lake Ontario to be 2.2 g (d.w.)/m²/yr, which included an estimate for reproductive material. The 95 percent confidence limits around this mean production estimate were 7.2 g (d.w.)/m²/yr. For comparison, Sell (1982) summarized mysid study results from several Great Lakes and found that mean population biomass for all lakes examined ranged from 0.1 to 1.1 g (d.w.)/m² and population production varied from 0.3 to 3.2 g (d.w.)/m²/yr.

Benthos

A series of shallow (7-35 m), intermediate (40-90 m), and deep (96-223 m) water stations were sampled for benthic fauna and sediment parameters in Lake Ontario by Nalepa and Thomas (1976) in November 1972, during IFYGL. Sand with particle sizes of less than 4.0 Phi units dominated most of the shallow water sites in the Lake. At intermediate depths some stations on the northern shore were dominated by sandy sediments while most other stations showed a mixture of sand and silt (4-6 Phi units) particle sizes. At the deep water stations silt dominated the substrate at all sites except a few on the northern side of the Lake, which exhibited greater proportions of sand. Mean particle size distributions in Phi units for the shallow, intermediate, and deep water station clusters were 3.0, 4.7, and 6.2, respectively.

Sediment chemistry for these same station groupings generally exhibited increasing patterns with depth (Nalepa and Thomas 1976). Mean total carbon at the shallow, intermediate, and deep water stations were 1.2, 1.5, and 3.0 percent, respectively. Greatest carbon content, at 10 percent of total sediment weight, was observed for a 190 m deep station. Total

nitrogen content of the sediments exhibited similar increasing trends with depth. Mean total Kjeldahl nitrogen for shallow, intermediate, and deep water sediments was 0.07, 0.14, and 0.20 percent, respectively. Similar to sediment carbon trends, the highest nitrogen content of sediments, 0.36 percent, was observed for a 190 m deep station in the Lake Ontario basin. Depth patterns for sediment phosphorus content were not evident from the 1972 study. The mean content at the deeper sites was slightly higher and showed less variability than at the intermediate and shallow stations. Sediment phosphorus content ranged from 0.02 to 0.72 percent of total sediment weight throughout the basin.

Nalepa and Thomas (1976) reported on the macrobenthic community from the same stations described above for sediment characteristics. They noted that a total of 48 benthic taxa were identified during the study and that oligochaetes and the amphipod *Pontoporeia hoyi* accounted for 91 percent of all fauna collected. The oligochaetes dominated faunal assemblage: in the shallow water zone while the amphipod was the dominant organism observed in the intermediate and deep water benthos. The oligochaetes, *Stylodrilus heringianus* and *Limnodrilus hoffmeisteri*, were the most widely distributed macrobenthic species occurring at 51 of the 55 stations sampled during this study. These species along with *Tubifex tubifex* were also identified as indicator species where *L. hoffmeisteri* and *T. tubifex* were associated with more polluted sediments and *S. heringianus* was found in cleaner benthic habitats. For this reason Figure 11 was used to illustrate the distribution of ubiquitous oligochaete fauna around Lake Ontario. *Stylodrilus heringianus* appeared in the majority of samples, especially all the deep water samples. In contrast, the more limited occurrence of *Tubifex tubifex* in samples was indicative of this species tolerance to polluted conditions. It occurred and usually dominated at eutrophic sites around the basin (Figure 11).

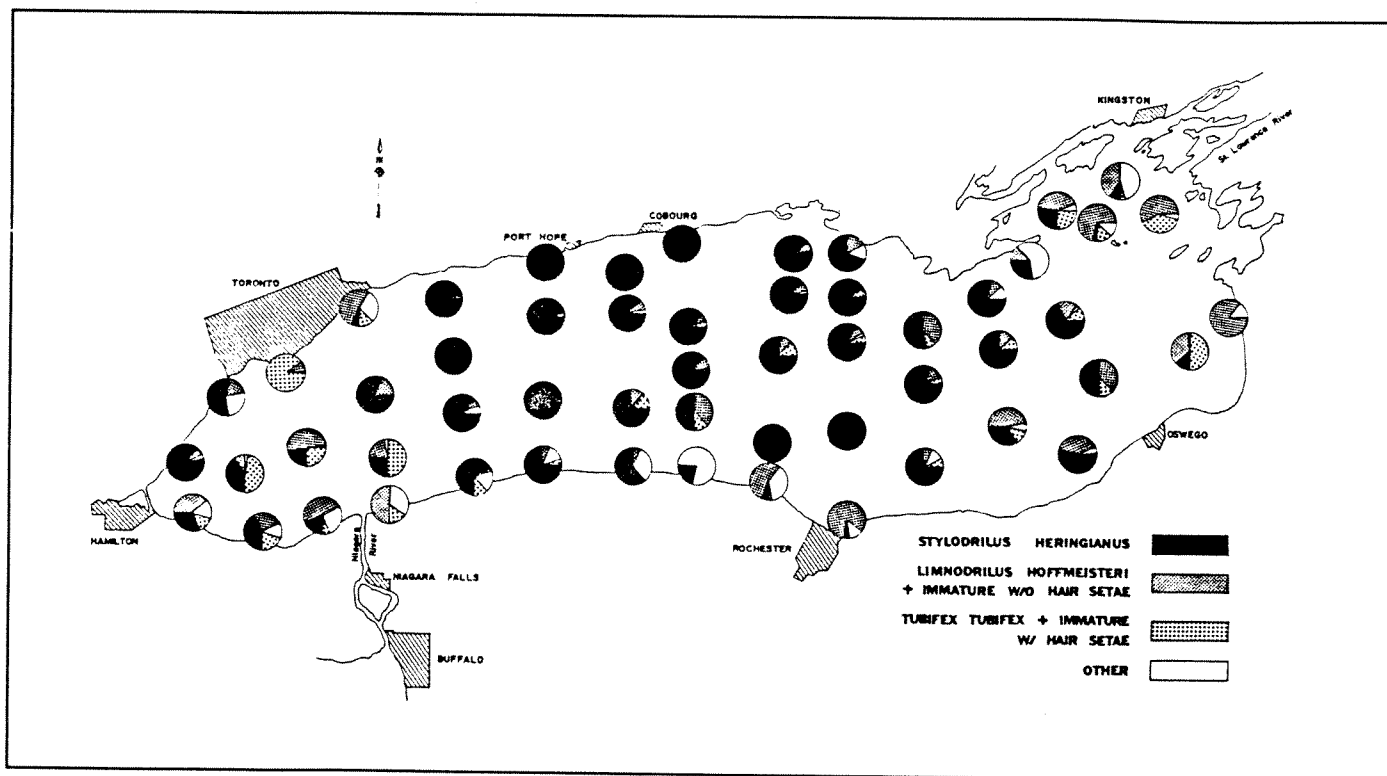


Figure 11. Percent composition of major oligochaete species and species groups at the various IFYGL sampling stations. From Nalepa and Thomas (1976).

Mean estimates for invertebrate total densities from the three depth zones were 11,000/m² for the shallow stations (<40 m), 7,000/m² for the intermediate depths (40-90 m), and 1,200/m² for the deep stations (>90 m). Nalepa and Thomas (1976) distinguished the shallow water zone as characterized by the dominance of *L. hoffmeisteri*. The intermediate and deep water areas, although dominated by *P. hoyi*, were also found to support *S. heringianus* as the dominant oligochaete.

In another study of Lake Ontario benthic macroinvertebrates, Kinney (1972) investigated species assemblages at a number of stations covering the same depth ranges as Nalepa and Thomas (1976). In Kinney's study, the deep water zone was characterized by a sparse macrobenthos with a total of 14 taxa representing a mean density of 1,131 organisms/m². The burrowing amphipod, *Pontoporeia hoyi*, was the most abundant fauna, accounting for 58 percent of the total benthos.

Oligochaetes, dominated by the lumbriculid *Stylodrilus heringianus*, were the second most abundant fauna at the deep water sites, constituting 32 percent of the benthos community. Spheriidae, especially genera representing the fingernail clams, Chironomidae, such as *Heterotrissocladius subpilosus*, the opossum shrimp *Mysis relicta*, and Hirudinea were the only other taxa observed from deep water collections.

The intermediate depth stations supported a more varied and abundant macrobenthic fauna than did the deep water sites (Kinney 1972). This zone supported 28 taxa with mean total densities of 4,368 organisms/m². Again *Pontoporeia hoyi* and oligochaetes were the most abundant taxa, representing 43 and 40 percent respectively, of the total fauna. Although *Stylodrilus heringianus* (33%) still dominated the oligochaete fauna in the intermediate depth zone, *Limnodrilus hoffmeisteri* increased in number to become the second most dominant worm population

(30%) and *Tubifex tubifex* increased its numbers over the deep water zone to represent 22 percent of total oligochaete density.

Kinney (1972) noted pronounced changes in the benthos in the transition from the intermediate depths to the shallow water, nearshore zone of Lake Ontario. A total of 38 taxa were observed at shallow water stations, representing a mean faunal density of 8,279 organisms/m². Oligochaetes replaced the amphipods as the most dominant fauna, comprising 67 percent of the total benthos. *Stylodrilus heringianus* populations in the shallow zone decreased to 9 percent of all oligochaetes and more pollution tolerant oligochaetes such as *Limnodrilus hoffmeisteri*, *Tubifex tubifex*, and *Potamothrix vej dovskyi* represented more than 70 percent of all oligochaete fauna. *Pontoporeia hoyi*, the deeper water amphipod, had its populations replaced in shallower waters by the amphipod *Gammarus fasciatus*. Amphipods accounted for approximately 16 percent of all the shallow water benthos. Sphaeriids, such as the fingernail clams, increased in overall mean densities, as compared with deeper zones of the Lake, and represented 13 percent of the total shallow water benthos. Leeches, several species of snails and chironomids, *Asellus sp.* and a few opossum shrimp, *Mysis relicta*, accounted for the remainder of benthos collected at shallow sites.

From a study in the Bay of Quinte and waters in adjacent eastern Lake Ontario, Johnson and Brinkhurst (1971a) described benthic species assemblages along a pollution gradient. Several genera, including *Limnodrilus*, *Chironomus plumosus*, *Tubifex tubifex*, *Procladius spp.* and *Coelotanytus sp.* were abundant in the inner, highly polluted portions of the Bay of Quinte. In the outer Bay of Quinte this association of benthic species was replaced by oligochaetes such as *Pelosclex ferox*, *Potamothrix spp.* and *Aulodrilus spp.* The benthos of the cleaner, colder offshore Lake Ontario waters, outside the Bay of Quinte, were characterized by species such

as *Pontoporeia hoyi*, *Stylodrilus heringianus*, *Tubifex tubifex*, *Chironomus anthracinus*, and *Heterotrissocladia subpilosus*.

Study of benthic fauna of a lake provides the most stable and consistent record of the lake's biological condition over time. Because most are sedentary benthic animals serve as useful long-term indices of the disturbance effects from pollution to which their populations have been exposed (Christie and Thomas 1981). From the studies described above it was apparent by examination of the benthos that there were several areas of Lake Ontario that experienced effects of pollution. Some of these areas included nearshore waters off of metropolitan Toronto, at the mouth of the Niagara River, and in the Bay of Quinte. In these areas, less pollution tolerant fauna such as *Pontoporeia hoyi* and *Stylodrilus heringianus* were depressed while oligochaetes that were pollution tolerant, such as *Limnodrilus hoffmeisteri* and *Tubifex tubifex*, dominated the benthic communities. One of the greatest densities of benthic invertebrates (68,000/m²) was observed in sediments at the mouth of the Niagara River (Nalepa and Thomas 1976) suggesting highly polluted (enriched) conditions.

The most abundant single species of benthic macroinvertebrate in Lake Ontario as a whole is undoubtedly *Pontoporeia hoyi* a glacial relict species of amphipod which can only survive in Lake Ontario because of a large, continually cold and well oxygenated hypolimnion. This species decreases and almost disappears from nearshore waters with heavy sedimentation and occasional low oxygen concentrations in the bottom waters. These populations are considered the most important benthic taxon in Lake Ontario because of their dominant role in production and transfer of energy to fish populations (Morsell and Norde 1968; Wells 1980). This species accounts for 80-90 percent of total benthic secondary production in eastern Lake Ontario and 68 percent in the main basin of

the lake (Johnson and Brinkhurst 1971b). According to Lake Michigan estimates for production of *P. hoyi* (Sell 1982), biomass ranged from 1.9 to 5.2 g (d.w.)/m² and production of this species varied from 0.8 to 6.0 g (d.w.)/m²/yr. Johnson and Brinkhurst (1971b) estimated that *P. hoyi* standing stocks in Lake Ontario equaled 1.8 g (d.w.)/m² and 4.2 g (d.w.)/m² at a deep water site and a 30 m site, respectively. For Lake Ontario deep waters overall, these investigators calculated production of the amphipod to range from 8 to 10 g (d.w.)/m²/yr.

On a transect from the inner Bay of Quinte to the open waters of Lake Ontario, Johnson and Brinkhurst (1971c) measured sediment metabolism in order to determine use of energy imported from the water column by the benthos. By examining a trophic gradient (Johnson and Brinkhurst 1971a) they felt one could understand the dynamics of energy flow in the benthos of Lake Ontario. Sediment metabolic rates, moving from the inner Bay of Quinte to Lake Ontario, averaged 0.35, 0.25, 0.22, and 0.15 g O₂/m²/day, respectively. Approximately 90 percent of the imported energy from the water column to the benthos was used by benthic communities at the less eutrophic lakeward stations of the transect examined by Johnson and Brinkhurst. At the more heavily polluted inner-bay station, only 12 percent of energy imported to the benthos was estimated to be used by these communities. Low utilization of energy in the inner-bay was substantiated by greater sediment organic content (28%) as compared to the more lakeward stations (3-4%). In contrast to actual usage of imported energy to the benthos, the assimilation of this energy also showed a decrease along the trophic gradient. Whereas 30 percent of the energy was assimilated at lakeward sites, again a much lower rate of assimilation was observed for the benthos of the inner-bay station (2%).

Summarizing overall benthic dynamics on this same transect in the Bay of Quinte, Johnson and Brinkhurst (1971b) observed that average standing stocks from inner-bay

to the deep waters of Lake Ontario were 1.29, 8.12, 6.08, 6.94, and 2.40 g (d.w.)/m², respectively. Production estimates for the benthos of three Bay of Quinte sites and an open lake site of 30 m depth were 12.3, 38.8, 10.9, and 8.4 g (d.w.)/m²/yr, respectively. At all sites along this transect production of the benthos reached peaks in the summer, related to greater water temperatures. Minimum production rates were observed in the winter, during January and February.

From the previous discussions of material imported to the benthos and that portion of material used and assimilated by benthic animals, it is quite apparent that dynamics of the water column directly influence the communities living in the lakebed sediments. Benthic-pelagic coupling is a mechanism that cannot be overlooked in evaluating the benthos of Lake Ontario. For example, in conjunction with a study on chironomid energetics in the Bay of Quinte, Johannsson and Beaver (1983) evaluated the importance of various algal species to the diet of chironomids. In terms of energy contribution, diatoms, especially *Stephanodiscus* and *Melosira*, were most important in May, August, and October with respect to consumption in chironomid diets. Blue green and small green phytoplankton species only sporadically appeared important in the diets of these benthic species. Overall, Johannsson and Beaver (1983) found that the contribution of pelagic algae as a food source to chironomid energetics was small (15-34%) and that bacteria appeared to play a more important role than any other food source in the energetics of chironomid populations.

Fisheries

The only constant in long-term trends regarding the fish communities of Lake Ontario, since European settlers first began to record their observations of them, has been change (Christie 1974). There have been variations in species composition which lasted for periods of a few years to

a few decades. These can be considered oscillations around the norm expected for any natural system. On the other hand, there have been many changes of longer duration including extinctions of native species and colonizations by exotic species. Some of these may have been natural shifts associated with long-term climatic trends, or other factors. But by far the majority have, in one way or another, been induced by man's activities on, and around the Lake. Some of the major changes, especially in fish species important to man, can be seen from a summation of commercial catch for these species (Figure 12; Beeton 1969). In the case of all species illustrated there were significant decreases noted at various times between 1900 and 1960.

With the exception of annual evaluations of target species stocks by federal, state and provincial agencies, probably the most comprehensive assessment of fisheries in Lake Ontario occurred during the International Field Year of the Great Lakes (IFYGL) in the early 1970s. There has been no significant commercial fishery for lake whitefish (*Coregonus clupeaformis*), lake trout (*Salvelinus namaycush*), several deepwater ciscoes (*Coregonus spp.*), blue pike (*Stizostedion vitreum glaucum*), Walleye (*Stizostedion v. vitreum*), or sturgeon (*Acipenser fulvescens*) in Lake Ontario for many years (Christie 1973), and the IFYGL surveys confirmed that there were no major reserves of these species remaining.

The most conspicuous feature of the IFYGL sampling program was the paucity of fish in the abyss of the main lake in the early 1970s. The largest number of fish species and the highest diversity value were obtained for the far eastern part of the Lake during IFYGL. While the total number of species was lower, the Bay of Quinte catches yielded the next highest diversity value. This is not surprising since both areas are part of the outlet basin complex which traditionally produces the highest fish yield (Christie 1973).

Seventy-nine percent of all fish caught by gill nets were taken in 27 m of water or less and 91 percent were caught in depths shallower than 55 m (Christie and Thomas 1981). The general finding was that below 55 m, the fish biomass was small and dominated by three species: alewife (*Alosa pseudoharengus*), the most abundant; rainbow smelt (*Osmerus mordax*); and slimy sculpin (*Cottus cognatus*). After extirpation of the native lake trout (*Salvelinus namaycush*) and virtual elimination of burbot (*Lota lota*) in the 1950s (Christie 1973), offshore waters were almost devoid of large piscivores until the late 1970s when massive annual stockings of hatchery-reared salmonines began. The principal species stocked were lake trout, chinook salmon (*Oncorhynchus tshawytscha*), coho salmon (*O. kisutch*), brown trout (*Salmo trutta*), and rainbow trout (*S. gairdneri*). From an initial planting of 76,000 fish in 1968, the number stocked annually rose irregularly to about 2.8 million in 1974. Releases did not exceed 3 million until 1980-81, when plantings increased sharply to about 4.5 million and then increased each year thereafter, exceeding 8 million in 1984 (O'Gorman, et al. 1987).

Over time, most of the commercial fish catch for Lake Ontario has been taken in the eastern waters of this basin. In a large measure this is because shallow waters are more productive than deeper waters. The main trench of Lake Ontario has its western component beyond the Scotch Bonnet Sill, and its only appreciable area of shallow water is near Hamilton. The eastern basin contains the greatest depth (to 244 m) and it is separated from the eastern shallow Kingston (outlet) basin by the line of islands and shoals called the Main Duck Sill. The shoreline eastward from Brighton to Stony Point on the U.S. side, including the islands and the Bay of Quinte, makes up 72 percent of the whole length of the Lake Ontario shoreline. This shallow area provides a great variety of habitat for many kinds of fish, and accounts for the largest single area of water shallower than 55 m, in the lake.

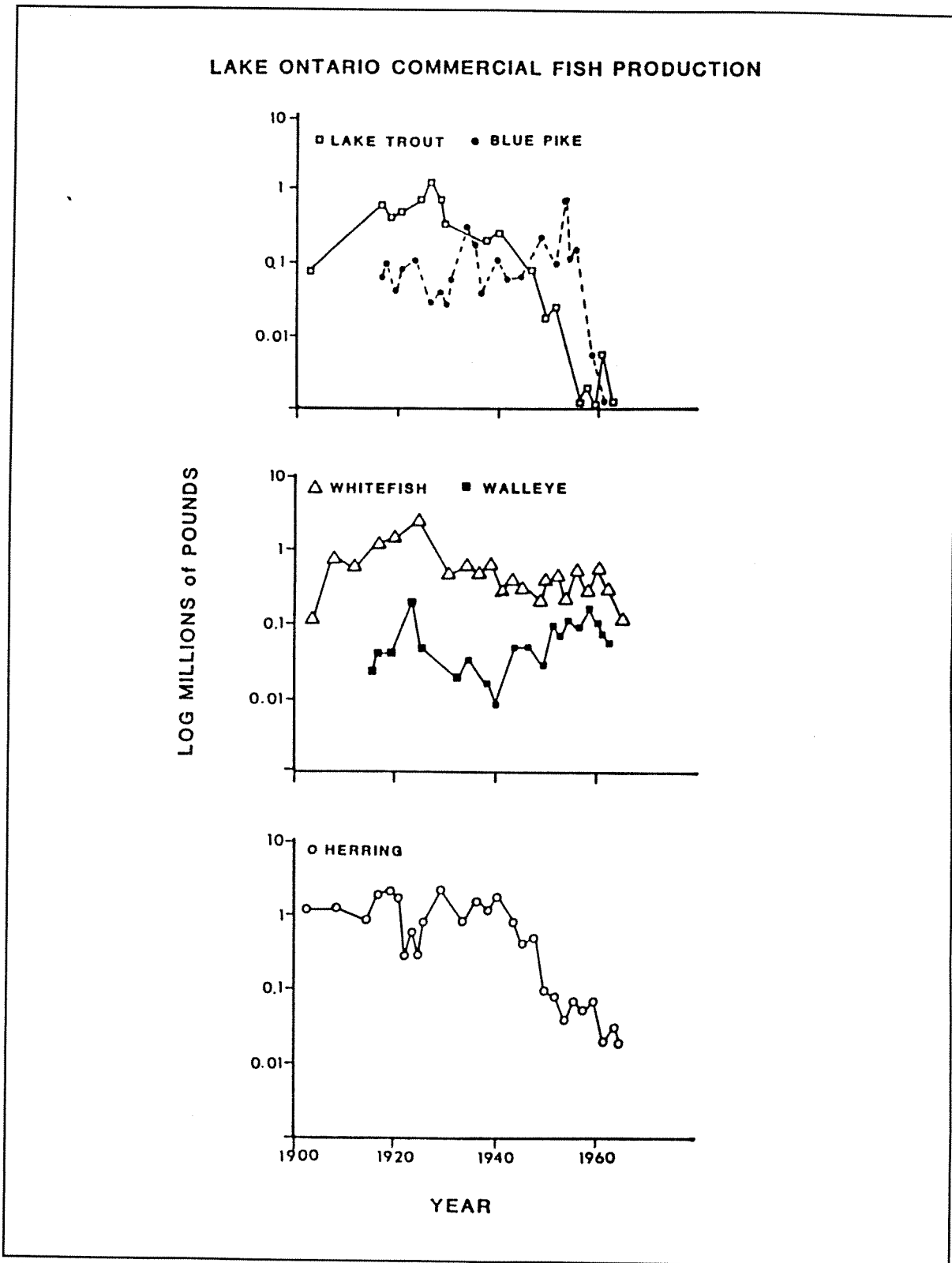
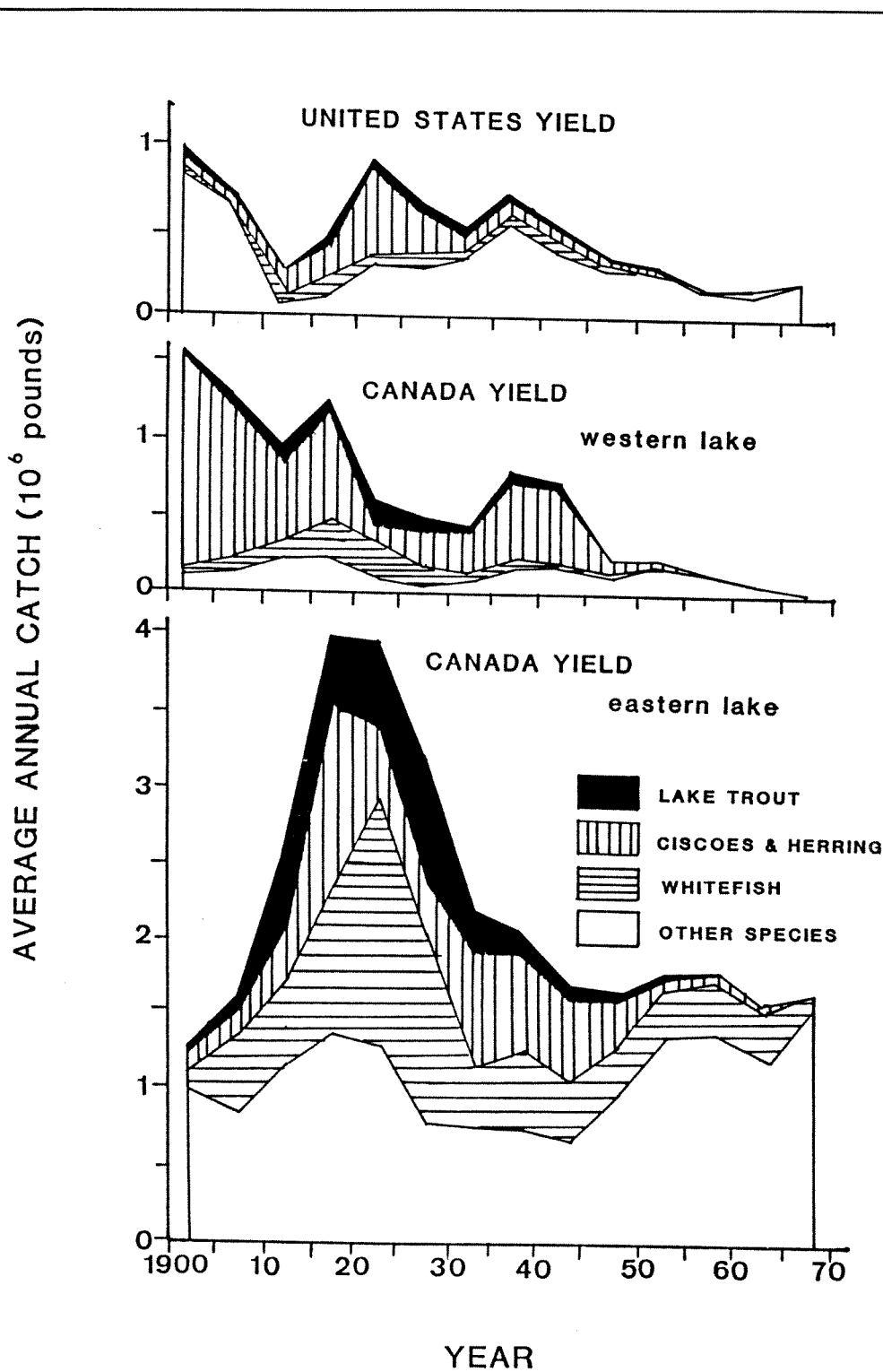


Figure 12. Commercial production of blue pike, lake trout, whitefish, walleye, and lake herring in Lake Ontario from 1900-1968. From Beeton (1969).



(Eastern lake includes the Bay of Quinte and the western and eastern yields are from Canadian waters. The U.S. yield is from U.S. waters only)

Figure 13. Trends in the major components of the Lake Ontario commercial fishery catch since 1900. From Christie (OMNR, Picton, Ontario, personal communication, August 1986).

An indication of how productive this eastern region has been over time is illustrated in Figure 13. These patterns for all parts of the lake further show the overall decline in fisheries. The Canadian western and U.S. fisheries had collapsed or were greatly reduced by the mid-1940s, whereas the eastern fisheries were not so drastically affected (Christie 1973). The western and central fisheries in the Canadian waters lost their stocks of ciscoes, whitefish, and lake trout, and the fisheries collapsed because there were no alternative species available. In the eastern waters, the lake whitefish persisted through the 1950s and increased yields of warm water fishes partly compensated for the disappearance of the important offshore species. Whereas the average yield fell 64 percent in the western region, it decreased only 40 percent in the eastern region and 27 percent in the Bay of Quinte (Christie 1973).

The declines of the premium species were virtually complete by the mid- 1960s (Figure 13), and in the eastern waters they were completely replaced in the catches by nearshore species of which the white perch, yellow perch and American eel were most important. It is important to note that the total catches averaged 2.5 million pounds per year from 1945 to 1973, regardless of the species mix.

Another interesting pattern has been the changes in fish that serve as the primary forage base for the more important predatory fish (piscivores) in the Lake. Brandt (SUNY Oswego, personal communication, 1986) performed an assessment of long-term fishery patterns in Lake Ontario based upon the fish species feeding habits (e.g. benthivore, planktivore, piscivore). His grouping of fish species on this basis and plots of their population trends through time (Figure 14), showed that most species had good population numbers between 1920-1940. After 1940 there was a general decline with a slight recovery for the planktivores and piscivores in the early 1960s. Another interesting aspect of Figure 14 was that the piscivore and planktivore abundances mirrored each other throughout time, while in most instances the benthivores in the Lake showed no direct relationship with either of these predominantly pelagic groups. These patterns suggested a fairly close link between the important predatory fish in the Lake and the pelagic forage base, represented by the planktivores. The patterns also illustrated the decoupling that may occur in a deep lake like Ontario with regard to benthic and pelagic components, since the benthivores did not always mirror the pelagic fish densities. From this plot it appeared as though the benthic zone was much more stable (buffered) through time from environmental changes than was the pelagic zone.

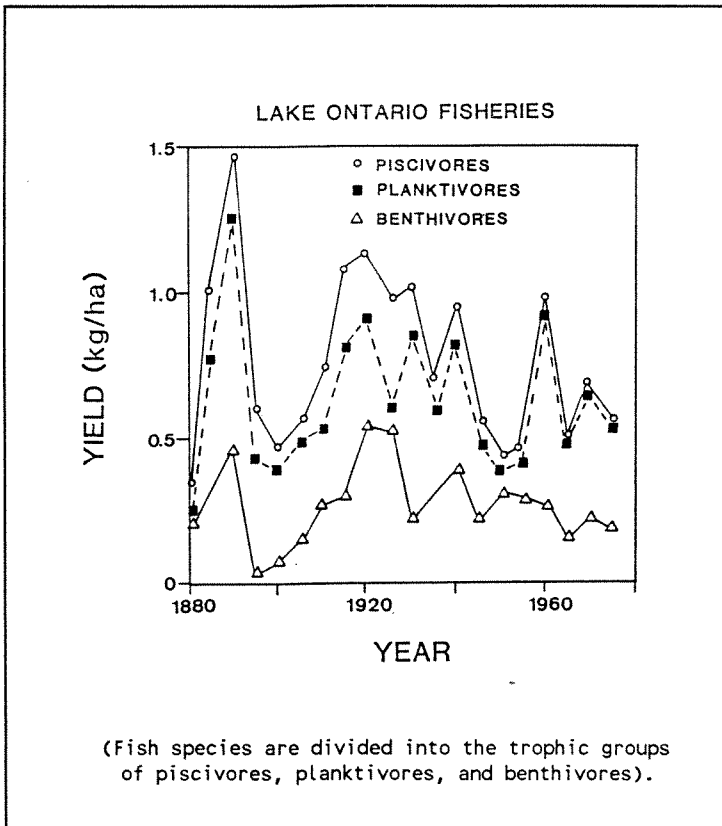


Figure 14. Plot of commercial fisheries yield from Lake Ontario between 1880 and 1980. From Brandt (Univ. Maryland, Solomon, MD, personal communication, January, 1987).

It is widely recognized that the continued support of predatory fish of importance to man that still survive in Lake Ontario or are annually stocked for population maintenance, relies heavily on the dynamics of the forage fish in the Lake, primarily the alewife. Size of the alewife population has fluctuated widely during the 1976-85 time period (O'Gorman, *et al.* 1987). The stock collapsed during the winter of 1976-77 when harsh weather triggered a severe die-off. After this period the stocks rose in abundance to a high in 1983 (Figure 15) before another, although less severe, die-off in the winter of 1983-84. Although not demonstrated by the data available to date, it is believed that these significant changes in alewife stocks will impact the larger predators of this stock as these predators became more abundant through stocking programs.

On a lakewide basis, smelt tend to be relatively more numerous in the waters of the outlet basin, while alewife dominate the main-lake trench. Within the range of depths occupied and where bottom conditions were sufficiently smooth to permit trawling along the main-lake transects, sculpin were often as numerous as alewife along the same transects. This was not the case in the Bay of Quinte, where sculpins did not enter the catch, or in the outlet basin, where sculpin were less available to the trawls (Christie, *et al.* 1987).

In the mid-water IFYGL trawling program, both smelt and alewife were found living pelagically from near the shoreline to the center of the Lake in the late summer, early autumn, and spring periods. The IFYGL evidence suggests that pelagic and vertically migrating invertebrates are harvested in the main trench of the Lake by smelt and alewife. These species contribute most to the nearshore communities by their annual inshore-offshore migrations when they experience highest mortality through predation by the piscivore fishes, or in the case of the alewife, through periodically severe shoreline mortalities.

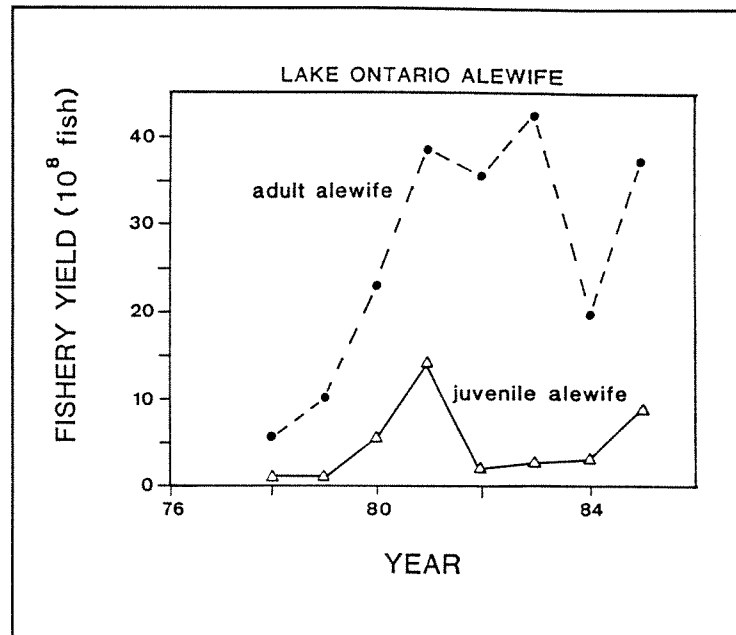


Figure 15. Mean estimates of number of adult and juvenile alewife near bottom in U.S. waters of Lake Ontario during late April to early May. From O'Gorman, *et al.* (1987).

A plot of biomass for forage base species from 1960 to 1985, along with one of the major predators (J. Christie, personal communication, 1986), illustrates the complexity involved in assessing fish stock relationships for such a large ecosystem that also possesses major pollution problems (Figure 16). Figure 16 illustrates abundances of alewife, smelt, slimy sculpin and lake trout from 1961 to 1985. Although the numbers for any particular time period may not be comparable because of different sampling gear used, the long-term relative patterns are important to contrast. Generally, the 25 year trends for alewife and smelt were extremely variable. Since the late 1970s however, smelt have shown a distinct decrease in population numbers (Figure 16). Lake trout, primarily because of stocking programs initiated in the late 1960s, have shown significant increases in the last 8 years (Figure 16). These increases in a top predator, which reflect the general increases in all salmonid populations in the Lake through stocking strategies, may in part be responsible for declines in the other species shown in Figure 16, which make up a significant part of the Lake Ontario forage base.

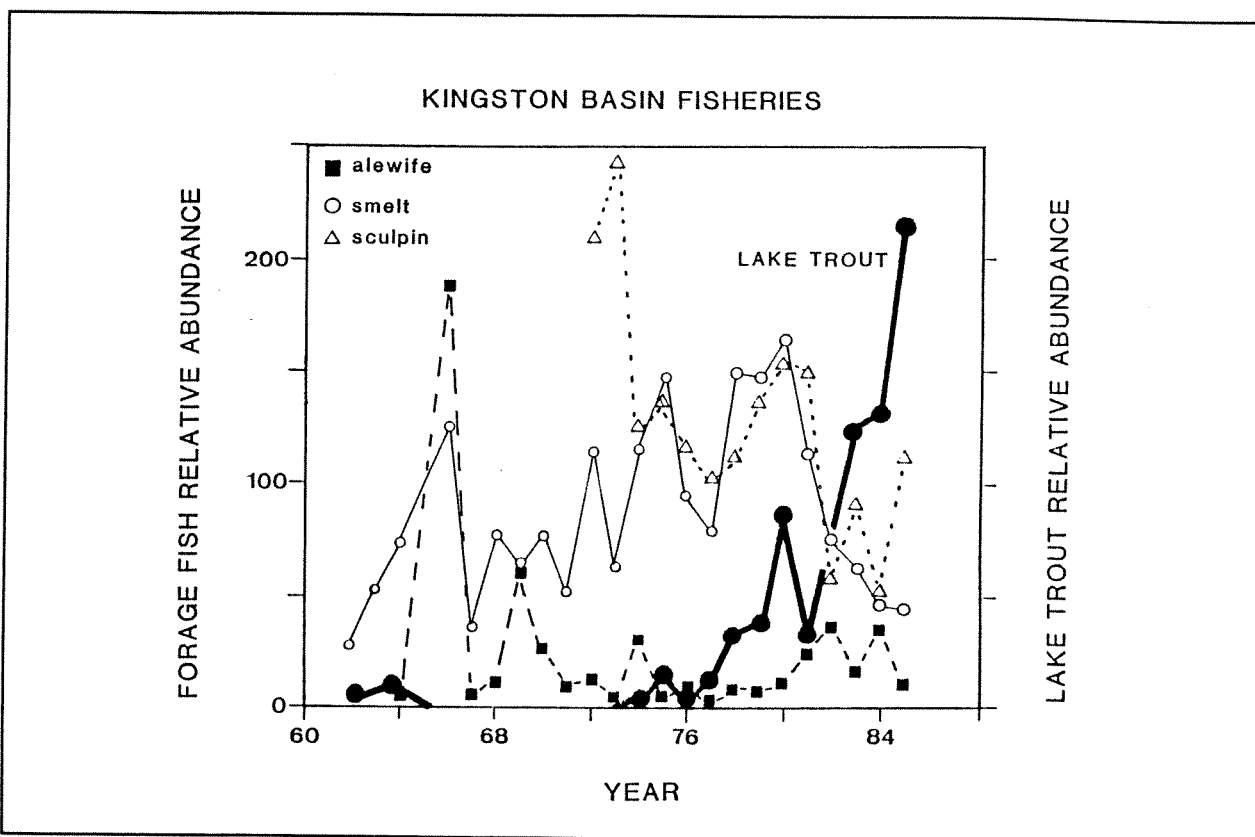


Figure 16. Mid-summer relative abundance estimates for fish species sampled by gill-net and trawls in the Kingston Basin of Lake Ontario. From Christie (OMNR, Picton, Ontario, personal communication, August 1986).

particular time period may not be comparable because of different sampling gear used, the long-term relative patterns are important to contrast. Generally, the 25 year trends for alewife and smelt were extremely variable. Since the late 1970s however, smelt have shown a distinct decrease in population numbers (Figure 16). Lake trout, primarily because of stocking programs initiated in the late 1960s, have shown significant increases in the last 8 years (Figure 16). These increases in a top predator, which reflect the general increases in all salmonid populations in the Lake through stocking strategies, may in part be responsible for declines in the other species shown in Figure 16, which make up a significant part of the Lake Ontario forage base.

An assessment of lake trout food habits was conducted by Christie (OMNR, personal communication, 1986) which illustrated some significant changes over time (Table 4). In the western portion of Lake Ontario, the dominant food item of lake trout in 1927 appeared to be alewife, followed by cisco and deepwater sculpin, two species that have disappeared from Lake Ontario today. During the 1978-1984 assessment period, the food habits of lake trout in the western basin shifted to slimy sculpins, followed by smelt as preferred food items. In the Kingston basin, at the eastern end of Lake Ontario, slimy sculpin was by far the dominant food item of lake trout during a 1959-1966 survey period (Table 4). In the 1974-1984 survey period, food preferences of the lake trout had shifted to a more equitable taking of slimy sculpin, alewife, and smelt in similar proportions.

State of a Great Lake

Table 4. Summary of Lake Ontario lake trout food habit observations from three different periods between 1927 and 1984. Observations expressed in percentages. From Jack Christie, Ontario Ministry of Natural Resources, Glenora Fish. Sta., Picton ON, personal communication, August, 1986.

SPECIES	WESTERN BASIN		KINGSTON BASIN	
	1927 ^b	1978-84	1957-66	1974-84
Alewife	66.4	7.2	11.1	28.4
Smelt		37.6	11.7	35.5
Cisco	31.2			
Deepwater Sculpin	12.5			
Slimy Sculpin	3.9	55.1	69.4	35.3
Trout-perch			1.0	0.4
Johnny darter		0.1	6.7	0.3
TOTAL STOMACHS EXAMINED	272	1882	925	5273

^a Percentages total to more than 100 because of multiple occurrences.

^b From Dymond (1928) for period of June 7 to August 23, 1927.

These more isolated changes in food habits that have occurred for one of the major predators in Lake Ontario over time hint at the significant changes that have occurred in the food web supporting the fish community during the last century. Brandt (SUNY Oswego, personal communication, 1986) developed a schematic demonstrating differences in the Lake Ontario food web between 1920 and 1960 (Figure 17). The most notable changes are the fact that many of the native species of fish serving as prey for predators such as lake trout have disappeared from the Lake Ontario community and have been replaced by exotic species such as the alewife and smelt. The diversity of food web linkages has also decreased significantly in more recent years. The factors that have

influenced these changes most certainly interact in a complex fashion but probably include overfishing, invasion of exotic species, fish stocking and degradation of the habitat through eutrophication and the occurrence of toxic substances in the Lake.

Lake Ontario Food Web

One of the problems facing many of the Great Lakes is the potential for lack of adequate nutrition, in the form of biological productivity, to support the significant increases in piscivore biomass reached through management stocking programs in recent years. This problem stems from a combination of nutrient

Lake Ontario

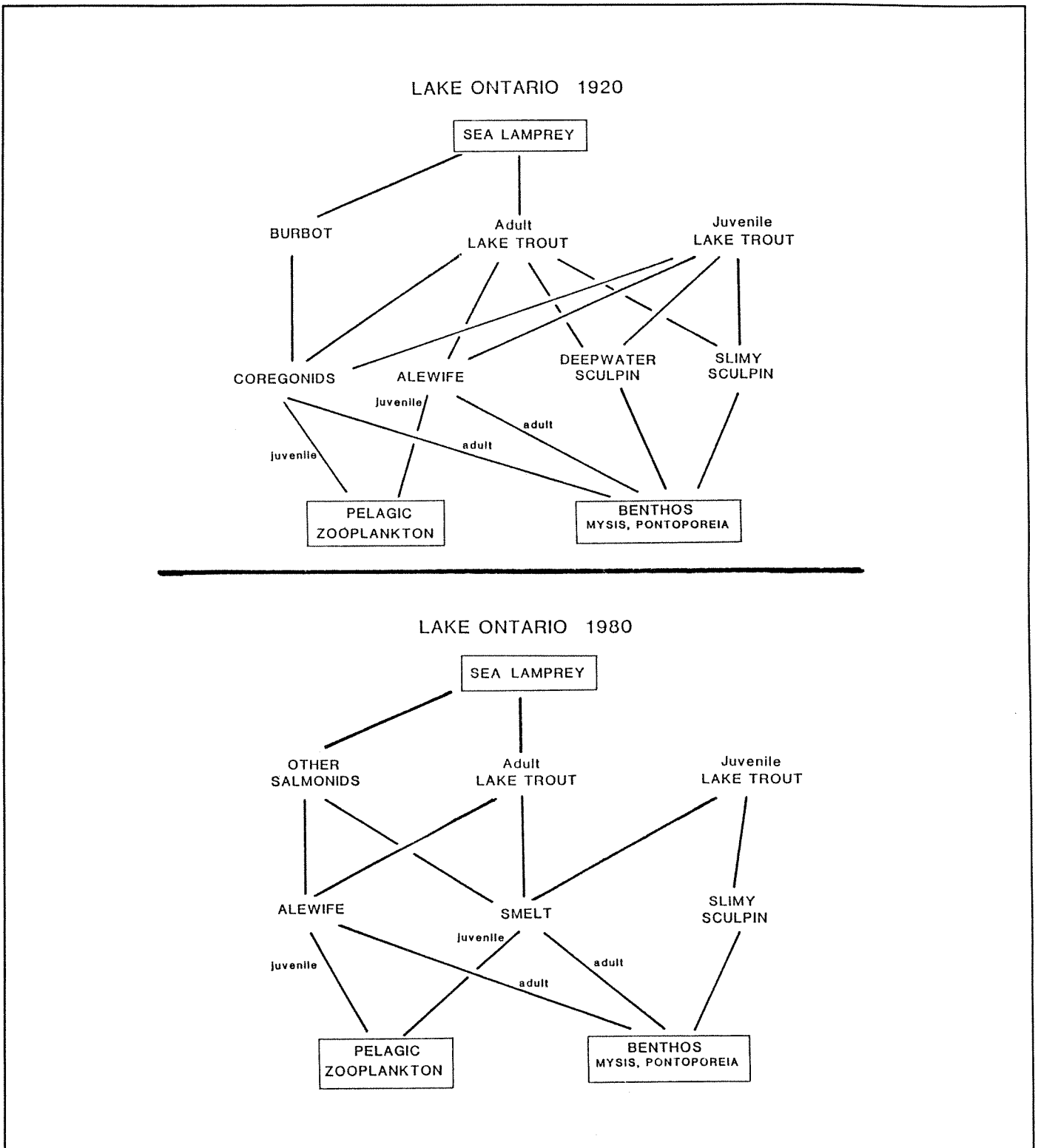


Figure 17. Schematic illustrating the changes in the fish community of Lake Ontario between 1920 and 1980. From Brandt (Univ. Maryland, Solomon, MD, personal communication, February, 1987).

abatement strategies and the stocking of millions of salmonine juveniles to the Lake annually. Eadie and Robertson (1976), Scavia (1979), Borgmann, *et al.* (1984) and Flint (1986) have addressed portions of this problem for Lake Ontario through the development of carbon budgets for some components of the Lake, the description of flux for various nutrients through the ecosystem, and the estimated support of salmonine biomass from zooplankton production measures.

Several accounts in recent years have attempted to conceptualize and simulate food web links in valuable fishery systems, producing useful heuristic tools which could help shape future research and management policy. Below, a mass balance conceptual model of the Lake Ontario deep water food web is described, based upon a report by Flint (1986), to evaluate pathways of energy flux and determine available nutrition for top level predatory salmonines. This conceptualization of carbon flow through the Lake Ontario deep water food web can contribute to our understanding of the dynamics underlying support of stocked salmonine populations in the ecosystem and present a clearer understanding, ultimately, of the impact of changing nutrient dynamics.

The development of this conceptual model was intended to provide inferences about: (1) the specific components of the Lake Ontario food web, (2) the relative importance of these components with respect to trophic rearrangements, and (3) the general nature of energy flow in the ecosystem.

The actual trophic links that were identified to develop the conceptual model are described in Flint (1986). These links unfortunately had to rely upon various literature reports that spanned a period from 1970 to 1982. Therefore, the initial conceptual scheme represents a composite of food web dynamics that cover a decade of information on Lake Ontario. This composite is illustrated in Figure 18 and will be briefly summarized below.

Estimates of annual phytoplankton production for Lake Ontario in the early 1970s ranged from 150 to 300 g C/m²/yr (Haffner, *et al.* 1984b). Assuming the inshore area equaled 10 percent of the total lake surface area, average primary production was calculated to be approximately 178 g C/m²/yr (Eadie and Robertson 1976; Stadelmann, *et al.* 1974). This production rate accounted for a total of 3.4 x 10⁹ kg C fixed during the year through photosynthesis by phytoplankton.

Zooplankton production for Lake Ontario has been estimated to be 15.02 g (dry wgt.)/m²/yr (Borgmann, *et al.* 1984). Using the assumptions of dry weight equalling 10 percent wet weight and carbon content equalling 6 percent of wet weight (Flint 1986), carbon production for zooplankton in Lake Ontario equaled 6 g C/m²/yr. From more recent data collections on Lake Ontario (O. Johannsson, Canada Center for Inland Waters, Burlington, personal communication, August 1985), mysid production was estimated at 2.2 g (dry wgt.)/m²/yr. Assuming the same conversions as above, this production rate equaled 0.9 g C/m²/yr. Thus, total water column crustacean production in Lake Ontario potentially equaled 6.9 g C/m²/yr, including both smaller zooplankton species and mysid populations.

Benthic production estimates were obtained from several sources. Johnson and Brinkhurst (1971c) measured metabolic rates of Lake Ontario benthos (oxygen consumption). Their estimate of shallow lake production (10% of lake surface area) was calculated to be 87.1 mg C/m²/day, while deep water production (90% of lake surface area) for the total benthic community was estimated at 47.9 mg C/m²/day. Combining these measurements indicated that the Lake Ontario benthic community produced 9.85 x 10⁸ g C/day. Assuming 182 days per year of maximum production because of temperature limiting factors (Johnson and Brinkhurst 1971a), annual macrofaunal benthic production (20% of total benthic production, which included bacteria and meiofauna; Walsh 1981) was calculated to equal 1.4 g C/m²/yr.

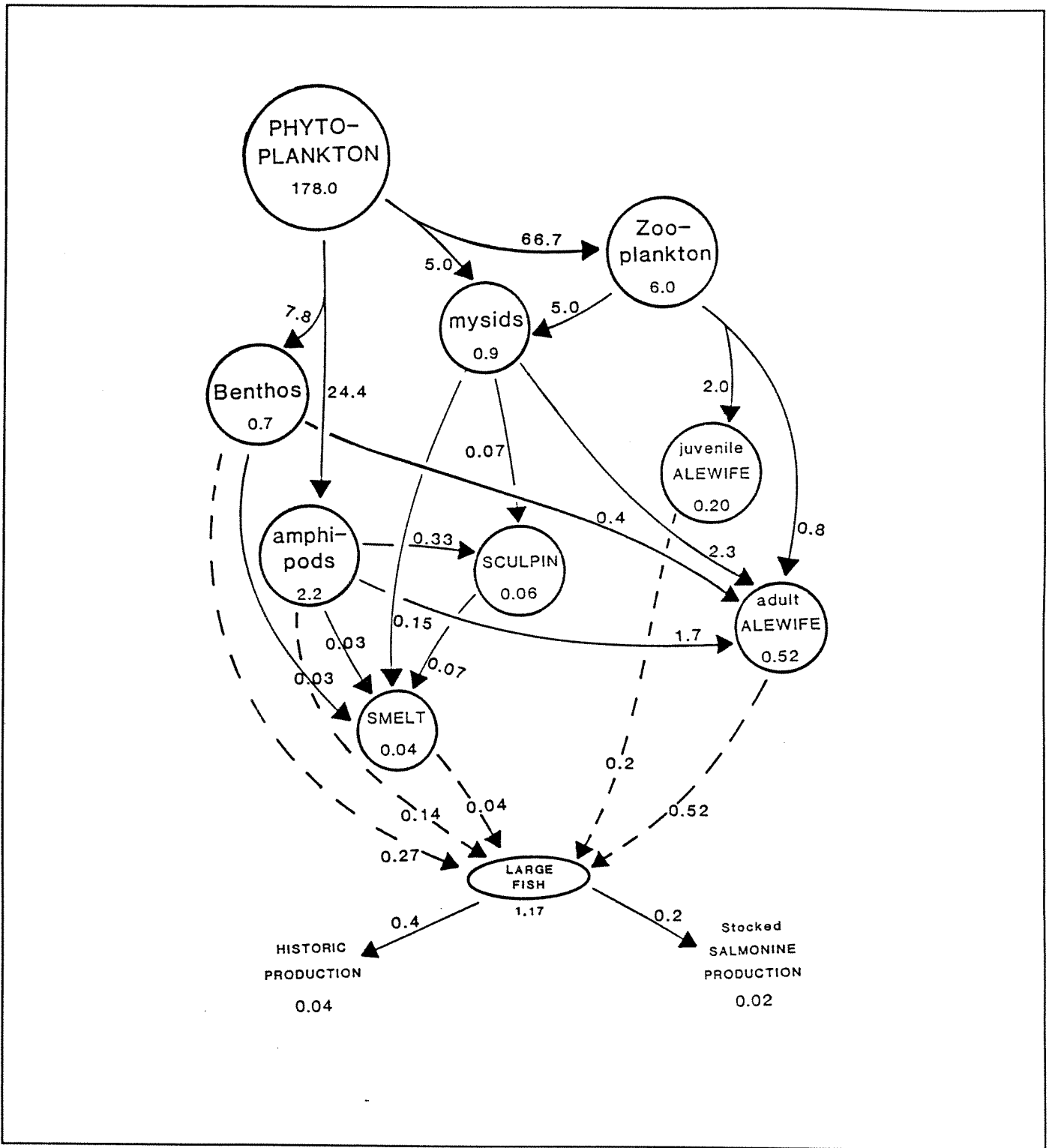


Figure 18. Hypothesized Lake Ontario food web. From Flint (1986).

A second means of estimating benthic production came from data of Johnson and Brinkhurst (1971a) that measured production of calories in the Bay of Quinte and deep waters of the lake. Assuming the caloric conversion to dry weight and carbon content described in Flint (1986), benthic production of the Lake Ontario shallow and deep water communities equaled 8.72 g C/m²/yr and 3.94 g C/m²/yr, respectively. Using the proportion of deep and shallow water habitat for the lake described above, total annual benthic production in Lake Ontario from Johnson and Brinkhurst (1971a) caloric measurements equaled 4.41 g C/m²/yr for the macrofaunal community component. For purposes of the conceptual model of Figure 18, an average of the two benthic production estimates from above was assumed representative (2.9 g C/m²/yr) and it was further assumed that amphipods comprised 75 percent of this production.

Forage fish production in Lake Ontario, can be characterized by a few dominant species such as alewife, rainbow smelt, and slimy sculpin. Average biomass of alewives near bottom in U.S. waters during spring over the last eight years in Lake Ontario (O'Gorman, *et al.* 1987) was 8.09×10^{10} g for adults and 2.36×10^{10} g for juveniles, for a lake surface area of 9.1×10^9 m² (New York waters). Assuming a conversion of wet weight to carbon content of 6 percent and a production to biomass ratio of 1.0 (Stewart, *et al.* 1981), alewife production for Lake Ontario equaled 0.2 g C/m²/yr for young of the year fish and 0.5 g C/m²/yr for adult fish (Flint 1986). Mean standing stock estimates for rainbow smelt in the U.S. waters of Lake Ontario equaled 6.5×10^9 g (O'Gorman, *et al.* 1987) while estimates of mean biomass for slimy sculpin between the Niagara River and Nine Mile Point in U.S. waters equaled 7.89×10^8 g (O'Gorman, U.S. Fish and Wildlife Service, Oswego, New York, personal communication, December 1985). Using assumptions stated in Flint (1986), annual smelt production was estimated to be 0.04 g C/m²/yr while slimy sculpin production equaled 0.06 g C/m²/yr.

Data on harvest of the top predatory fish species in Lake Ontario can be obtained from the 70 year average commercial fish statistics of this system (Christie 1973). The historic commercial catch of large predatory fish averaged 1.632×10^9 g/yr between 1900 - 1969. Employing a carbon equivalent conversion for wet weight of 10 percent and assuming a 10 percent catch efficiency for man, the production of large predatory fish in Lake Ontario was calculated to be 0.05 g C/m²/yr. In contrast, Loftus, *et al.* (1987) estimated that long-term top predator biomass yield in lake Ontario between 1920-1950 averaged 0.86 kg/ha or 0.009 g C/m²/yr. If we again assume a 10 percent catch efficiency for man, actual production of top predatory fish during the 1920-1950 period (from Loftus, *et al.* 1987) would be 0.09 g C/m²/yr. Borgmann, *et al.* (1984) estimated production of commercial fish species such as lake trout, herring, ciscoes, and whitefish through the last 70 years and, based upon assumptions stated above, their estimate equaled a carbon amount of 0.03 g C/m²/yr. Taking an average of Borgmann's and Christie's estimates and assuming a decrease from Loftus, *et al.* (1987) 1920-1950 estimates to the 1970's, average annual piscivore biomass production in the early 1970s equalled 0.04 g C/m²/yr.

Based upon the preceding estimates for carbon production of various Lake Ontario trophic levels, a conceptual food web exhibiting transfer of carbon for the early 1970s in Lake Ontario was constructed (Figure 18). Evidence supporting calculations for the flow of carbon among the various components of Figure 18 are detailed in Flint (1986). Trophic support for more recent salmonid production in Lake Ontario of 0.02 g C/m²/yr (Flint 1986), coupled with trophic support for long-term predatory fish production (0.04 g C/m²/yr), and assuming a 10 percent transfer efficiency from lower trophic levels, suggested that a total of 0.6 g C/m²/yr was required as nutrition for larger Lake Ontario planktivores, benthivores, and

Lake Ontario

Table 5. Minimum and maximum food consumption required by the different trophic components in the Lake Ontario food web to maintain assumed biomass production based upon reported variations in growth and assimilation efficiencies from the literature. From Flint (1986).

FOOD WEB COMPONENT	MINIMUM FOOD CONSUMPTION	MAXIMUM FOOD CONSUMPTION
ZOOPLANKTON ^a	50.0	100.0
MYSIDS ^b	5.3	15.0
BENTHOS ^a	5.8	11.7
AMPHIPODS ^a	18.3	36.7
SCULPIN ^c	0.3	0.6
SMELT ^c	0.2	0.4
JUVENILE ALEWIFE ^d	1.0	4.0
ADULT ALEWIFE ^d	2.6	10.4
TOP PREDATORS ^c (salmonines & others)	0.3	0.6

^a Energy conversion efficiency minimum of 6% and maximum of 12%.

^b Energy conversion efficiency minimum of 6% and maximum of 17%.

^c Energy conversion efficiency minimum of 10% and maximum of 20%.

^d Energy conversion efficiency minimum of 5% and maximum of 20%.

piscivores (Figure 18). This required support represents slightly over half of the total nutrition of 1.17 g C/m²/yr assumed available in the Lake Ontario conceptual scheme (Figure 18), based upon a composite, static model of carbon flow using data spanning 12 years on mean production and conversion efficiency rates (Flint 1986).

In evaluating the hypothesized flow of carbon through Lake Ontario, however, areas of significant predatory pressure became obvious. For example, the amount of zooplankton and mysid biomass required to support predators exceeded the amount of estimated carbon production by these trophic components (Figure 18). In addition, all slimy sculpin biomass was

apparently consumed and there appeared to be a significant amount of pressure exerted upon the amphipod populations in the Lake. Any alteration in trophic structure, with a shift toward further pressure being placed on these populations, could certainly overburden the capacity for biomass production. This is further evidenced by Table 5 which illustrated that using the minimum conversion efficiencies as reported in the literature for many of these species placed even more of a burden on each of the trophic components supplying food to a higher level, which would significantly alter the flow of carbon as demonstrated in Figure 18. In contrast, applying maximum energy conversion efficiency lessened feeding pressure on lower trophic levels (Table 5).

Development of this conceptual model has made two points clear. Two-to-three-fold changes in calculated biomass production within the different trophic levels will alter the model parameters and affect predictive ability. Similarly, the model is sensitive to small changes in energy transport, and conversion efficiencies have a major impact on the transfer of materials in the conceptual model, as demonstrated by Table 5. With this in mind, it is extremely important that research questions be directed toward addressing the actual biomass and energy

value of different prey and qualifying more precise information on energy conversion efficiencies for the different species involved. Various research projects in the Great Lakes are now focusing on these issues, and will be very important in future attempts to modify the conceptual scheme presented here.

This exercise has demonstrated the development of a conceptual model can further our understanding of dynamics in the Great Lake ecosystem. The confidence with which the model may be applied, however, remains uncertain. There may be errors in some of the components due to lack of sufficient data and values used for transfer efficiencies may be underestimated. These concerns point to the need for additional process-oriented research in these areas. The development and application of models such as the food web, carbon transport scheme described above, however, will hopefully suggest a way to evaluate decisions and predict future consequences that both depend on and affect Great Lake environmental characteristics. This conceptual construction of a food web model for Lake Ontario also represents an integration of biotic assemblage dynamics and energy transfer processes that can serve as a model for comparison through time relative to changing environmental conditions.

Lake Ontario

CHAPTER 5

LAKE ONTARIO WATER QUALITY

Lake Ontario possesses an extensive historical data base that represents an invaluable backdrop against which to evaluate long-term changes in water quality. The International Field Year of the Great Lakes (IFYGL), although not process oriented, provided most of our earlier records on water quality, especially nutrients, of this Lake. This extensive survey has been updated by several large Canadian studies and routine lake-wide monitoring by the U.S. Environmental Protection Agency, to develop an extensive long-term record of chemical dynamics in the Ontario system.

Because of its downstream location in the Great Lakes Basin, Lake Ontario represents a consolidation of most problems facing any of the other Great Lakes, including nutrient loading and toxic contaminant influx. In addition, Ontario also exhibits some water quality problems that are unique because of anthropogenic events that have only occurred in this Lake's watershed.

Reduction of anthropogenic loadings is a major thrust in current management of the Lake Ontario ecosystem, and understanding the relationship between all identifiable forms of input and the fate and distribution of nutrients and contaminants in the ecosystem is a key component. Therefore, ultimately a mass balance approach is needed to adequately manage the various inputs and move toward system rehabilitation. The mass balance approach requires information about the quantity of substances entering the system,

the quantity stored, transferred, or degraded within the system, and the quantity leaving the system. The difference between inputs and outputs is the quantity that remains to be recycled or lost to the sediments and the atmosphere.

Sources and pathways include tributary and connecting channels loading municipal and industrial discharges, urban stormwater discharges and combined sewer overflows, rural land runoff, atmospheric deposition, groundwater contamination (including pollutants coming from waste disposal sites and landfills), and release from bottom sediments. Amount of loading to the Lake Ontario system from these different sources will be dependent upon the particular chemical being considered. For example, a mass balance model developed for PCBs suggested that atmospheric loading to the Lake only represented approximately 7 percent (International Joint Commission 1987b). In contrast, connecting channel loading equalled approximately 81 percent and tributary loading another 10 percent. For lead input to Lake Ontario, on the other hand, atmospheric transport was calculated to be approximately 73 percent while connecting channels flow only contributed 23 percent.

If the mass balance approach to quantifying chemical dynamics in Lake Ontario is to be employed effectively, then the assessment of long-term trends in the various compartments that characterize the cycling of these chemicals in the system is a first step.

Major Ions

The ionic composition of Lake Ontario is governed largely by the quality of Lake Erie inflow water via the Niagara River, modified to some extent by inputs from both the atmosphere and the Ontario drainage basin. Annual mean ion concentration and loading estimates from the Niagara River for 1977 to 1983 indicated that there were no significant inputs of calcium, potassium or magnesium within the Lake Ontario basin, suggesting that Lake Ontario ion levels were similar to Lake Erie (Stevens 1987). More long-term patterns for these ions (Dobson 1985), however, indicate that since 1900 there has been an increase in total dissolved solids, chloride, sodium, potassium, sulfate and calcium within Ontario waters (Figure 19).

Stevens (1987) calculated average increases in load of 35% and 38%, respectively for sodium and chloride, between the Niagara River inflow and the St. Lawrence River outflow from 1927 to 1983. These trends are also suggested from Figure 19 where chloride in particular has exhibited major in-lake concentration increases since the early 1900s. The single most significant source of chloride within the Ontario basin is the Oswego River, the largest tributary within the basin, excluding the Niagara River. This river system has contributed approximately 65% of the total tributary load of chloride to the basin through the years. Effler, *et al.* (1985) determined that between 1970-1981 the mean annual release from the Oswego River drainage basin was 7.65×10^5 t, of which most was attributable to a chloralkali plant on Onondaga Lake near Syracuse, New York.

In general, Dobson (1985) indicated that chloride, sodium, sulphate and calcium were exhibiting increases of 0.49, 0.26, 0.17, and 0.11 ug/L/yr, respectively (Figure 19). Beeton (1969) cited an increase in major ions between 1910 and 1965 as indicative of eutrophication, although no causative relationship was identified. These influences, however, do

parallel man's increased activities on Lake Ontario. The interesting pattern to note with Figure 19 is the decreases in chloride and sulfate and leveling off of concentrations for calcium and sodium during the period since the mid-1970s. These changes parallel the beginnings of nutrient abatement programs in the Lake Ontario basin.

Nutrients

Spatial and temporal heterogeneity have been recognized as important in the resistance of lakes to eutrophication (Imboden 1974; Imboden, *et al.* 1983). Considering the level of effort being expended to reduce the extent of eutrophication of Lake Ontario, it becomes of considerable importance, then, to have an understanding of the distribution of chemical constituents within the Lake, as well as the underlying mechanisms responsible.

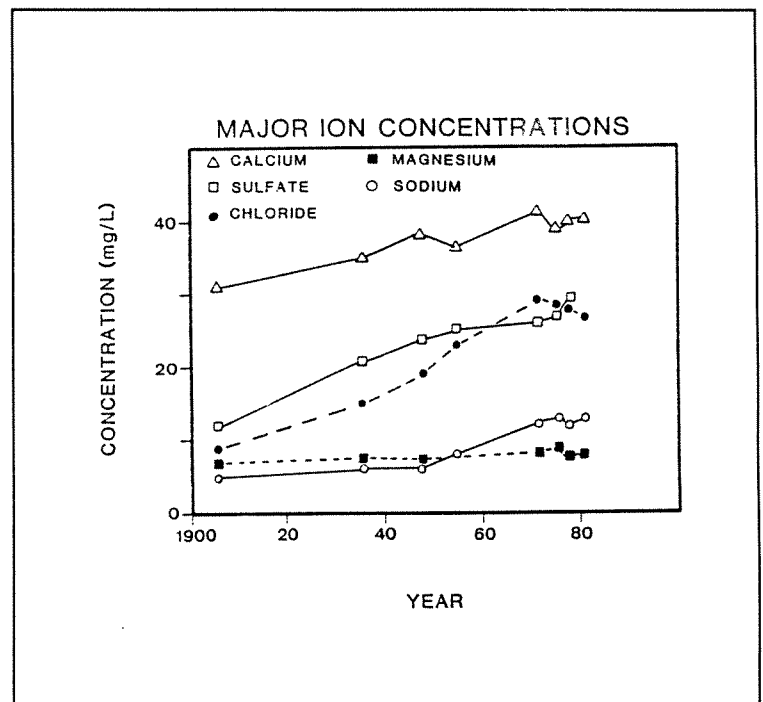


Figure 19. Trends in concentrations of five major ions in Lake Ontario. From Dobson (1985).

The distribution of nutrients in the offshore waters of Lake Ontario is the net result of anthropogenic inputs, thermal structure, circulation processes, assimilation by phytoplankton, and regeneration within the water column and from the bottom sediments (Gachter, *et al.* 1974). Maximum concentrations are generally observed during the spring in the nearshore zones, associated with increased loadings due to tributary runoff, and thermal bar formation which restricts nearshore-offshore water mass exchange. With the onset of thermal stratification, the epilimnion and hypolimnion are effectively isolated from one another. Subsequent losses of colloidal and particulate fractions from the trophogenic zone typically result in a summer minimum in the surface waters, while decomposition of this material in the tropholytic zone enriches solubilized fractions in the hypolimnion. Epilimnetic nutrient concentrations, in general, increase in the fall as the deepening thermocline entrains waters from the nutrient-rich hypolimnion. This bimodal distribution, with maxima in spring and fall, is characteristic of moderately productive, dimictic lakes.

Loading to Lake Ontario of wastewater flows, either directly into the Lake or from major tributaries, are the major source of nutrients to the system. A synthesis of all wastewater flows (Lake Ontario Toxics Committee 1988) indicates that tributary flow to the Lake contributes 3.75×10^6 m³/day while direct Lake discharge represents an additional 2.56×10^6 m³/day. In both cases the province of Ontario contributes greater than 75 percent of this total discharge while New York state averages less than 25 percent. Assuming a 1 mg/L concentration of phosphorus as a target load for all discharges to Lake Ontario (International Joint Commission 1987a) this flow rate of wastewater to the Lake would equal 6.31×10^6 g P/day of discharge.

Total phosphorus values averaged over all cruises for mid-lake stations between 1968 and 1985 showed considerable variability within each year (Dobson 1985). This variability could be attributed to the seasonal cycles described above. The annual mean total phosphorus values for mid-lake stations, however, showed a very definite decreasing trend after 1973 (Figure 20).

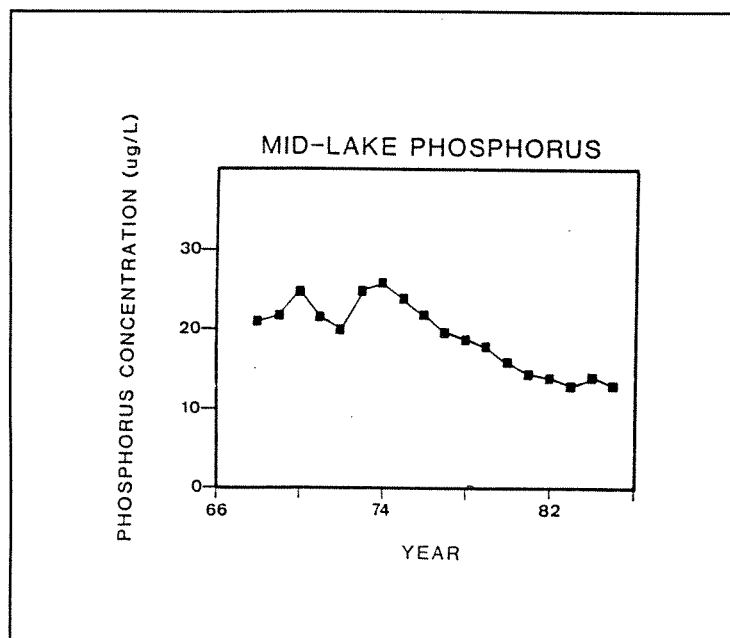


Figure 20. Mean total phosphorous in offshore, near-surface waters of Lake Ontario. From Dobson (1985).

The annual cycle of soluble reactive phosphorus in offshore near-surface waters indicated nearly complete depletion during the summer period, related to plankton growth (Figure 21). Again this seasonal pattern was similar as described above for a typical dimictic lake and followed patterns also observed for total phosphorus.

Cruise mean values of soluble reactive phosphorus from 1969 through 1982 in the mid-lake region are shown in Figure 22 for near-surface values and bottom water concentrations. It is believed that in Lake Ontario, long-term trends of soluble reactive phosphorus can best be studied by comparing late winter and early spring concentrations over years (Dobson 1985). March/April surface values peaked in 1973 (Figure 22) followed by declining values

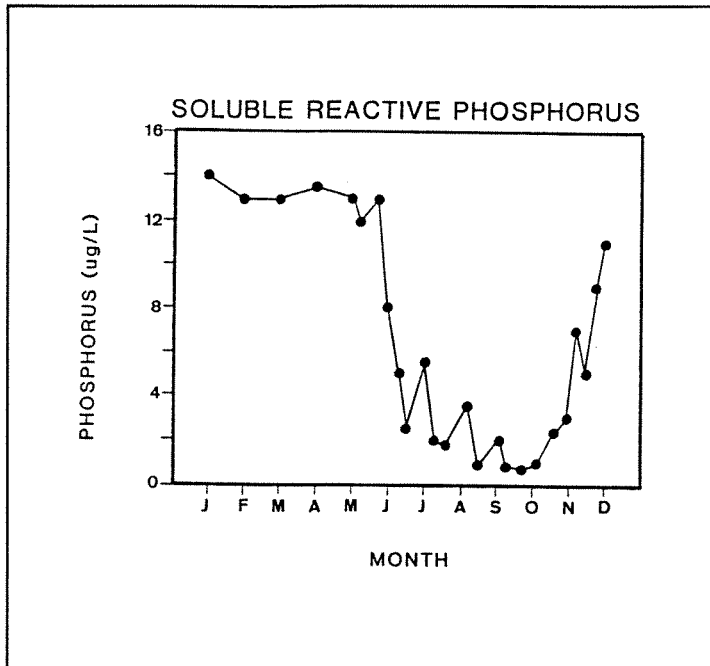


Figure 21. Seasonal cycle for cruise means of soluble reactive phosphorous in offshore, surface waters of Lake Ontario. From Dobson (1985).

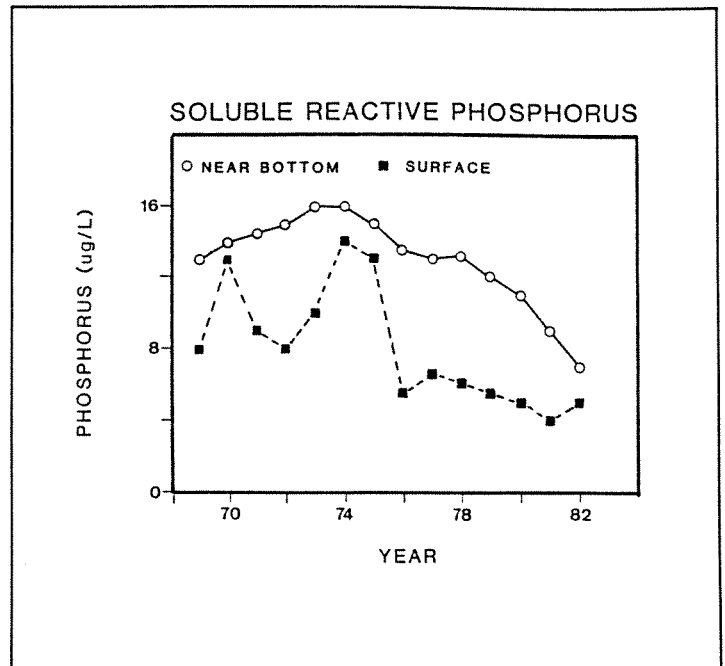


Figure 22. Annual mean soluble reactive phosphorous in the mid-lake, near-surface and bottom waters of Lake Ontario. From Dobson (1985).

through 1982. In contrast, bottom water concentrations did not show a similar annual decline until 1979 (Figure 22). In most years peak concentrations for bottom water soluble reactive phosphorus occurred in the fall, following summer maximum decomposition rates of organic matter in the hypolimnion. The lag in decreasing annual peaks in the bottom waters contrasted to the surface waters (1979 vs. 1974), may have been related to the build up of material in the Lake from excessive loading prior to 1972 and the slower response time to nutrient abatement programs by the sediments, contrasted to epilimnion waters in a deep lake like Ontario.

Dobson (1985) reported on long-term trends for nitrate and nitrite in the offshore region of Lake Ontario and noted that nitrite was probably very low. Nitrate and nitrite, however, have both increased steadily from 1968 to 1981 (Figure 23). Concentrations in the surface waters in summer were very low, about 10 ug N/L up to 1972, but in recent years there has been a residual nitrate and nitrite

level of 100 ug N/L. In contrast, bottom waters have shown no discernible trends either seasonally or annually.

The combination of decreasing phosphorus and increasing nitrogen in recent years in the offshore waters of Lake Ontario is also reflected in the long-term pattern of the spring nitrogen to phosphorus (N:P) ratios for the period of 1969 to 1982 (Figure 24). This ratio demonstrates a dramatic increase from a minimum of 10 in 1973 to a maximum of 30 in 1982. Since 1982, the spring TN:TP ratio has increased to a level greater than 35 (Stevens 1987). Furthermore, mid-summer TN:TP ratios in 1972 ranged from 12 to 14, suggesting that there may have been a seasonal shift from P to N limitation during the high phosphorus years (Lean, *et al.* 1987). In 1981 and 1982, summer TN:TP ratios were always greater than 20, with particulate N accounting for only 20 percent of TN, compared to 50 percent in 1972, indicating that nitrogen limitation may not be occurring presently (Stevens 1987).

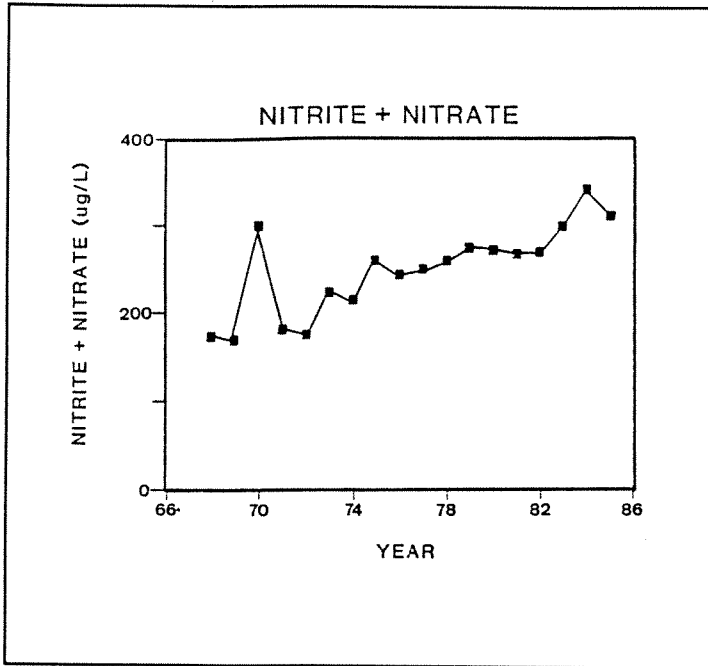


Figure 23. Mean nitrite plus nitrate concentrations in offshore, near-surface waters of Lake Ontario. From Dobson (1985).

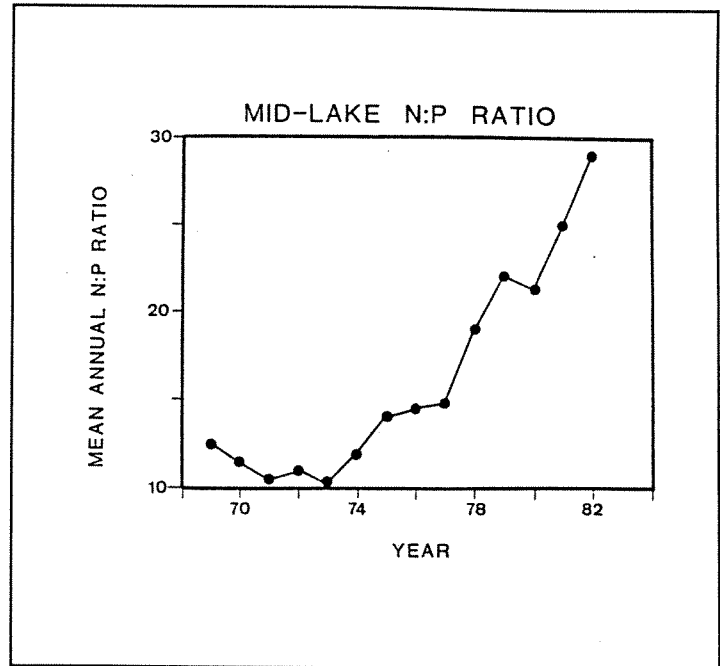


Figure 24. Changes in spring nitrogen phosphorous (N:P) ratio. Nitrogen calculated as nitrite plus nitrate plus ammonia. Phosphorous calculated as total P. From Dobson (1985).

Soluble reactive silica values were low in near-surface waters in summer during the period 1968 - 1982. A silica shortage could be limiting the stocks of diatoms in summer, permitting other kinds of phytoplankton to replace diatoms during this season (Dobson 1985). Soluble reactive silica values increased in the bottom waters in summer over this same period. This suggests that there was considerable internal loading (i.e., recycling, of silica) as described by the following scenario: a cycle of uptake near the lake surface, sedimentation of diatom frustules, and their dissolution near the bottom. The March/April values in surface waters fluctuated in the years 1968 to 1982, but the August/September surface values were always low, near 100 ug SiO₂/L.

From 1966 to 1981, oxygen in the offshore epilimnion waters of Lake Ontario followed a regular cycle in concentration values. Minimum values occurred in August/September in the range of 9 to 11

ug/L (Dobson 1985). In general values near 100 percent saturation in epilimnal waters during the summer interval in offshore waters. The deep water in Lake Ontario during late summer from 1966 to 1978 exhibited mean depletion rates of oxygen of about 1 ug/L every three months (Dobson 1985). Periods with the least hypolimnetic oxygen stratification were December/January and May/June, when temperatures very close to 4°C throughout the water column permitted overturn to the bottom. Based upon oxygen depletion rates in the deep water during the summer/autumn interval (Dobson 1985), between 1966 - 1981 there was no apparent long-term trend of anoxic conditions. The main basin of Lake Ontario appeared to have excellent oxygen conditions during this interval.

El-Shaarawi and Kwiatkowski (1977) examined the relative magnitude of seasonal and spatial variability for physical and biological variables in the surface waters of Lake Ontario. They found that, although seasonal variability was considerably

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seasonal variability was considerably greater than spatial, due in part to the bimodal seasonal cycle exhibited by most variables, these variables also exhibited significant spatial gradients. Therefore, Neilson and Stevens (1987), in order to more easily characterize the Lake Ontario system in more recent times (1981-82), attempted to divide the Lake into limnologically distinct water masses. They applied factor, cluster, and discriminant analysis to an extensive data base developed from 29 cruises conducted during 1977, 1981, and 1982 to arrive at the zonation pattern presented in Figure 25. The six zones produced by this analysis are readily interpretable in light of characterizations for water quality variable distributions, temperature patterns, and circulation processes.

Zone 1 represents the Kingston Basin region, receiving significant inputs from the Bay of Quinte and the Black River. Due to the influence of bottom morphometry, there is little impact from this region upon the main lake. Three nearshore zones are evident: zone 4 receiving inputs from Toronto, Hamilton Harbor and the Welland River; zone 3 being influenced primarily by discharge from the Niagara River, as well as some agricultural runoff along the south shore; and zone 2 receiving significant inputs from the Genesee and Oswego Rivers. Note that due to the predominantly counterclockwise circulation pattern in the Lake, and the strong nearshore currents along the south shore, these nearshore zones tend to extend to the right of the discharge, particularly zone 2. The east-west distinction of the mid-lake region

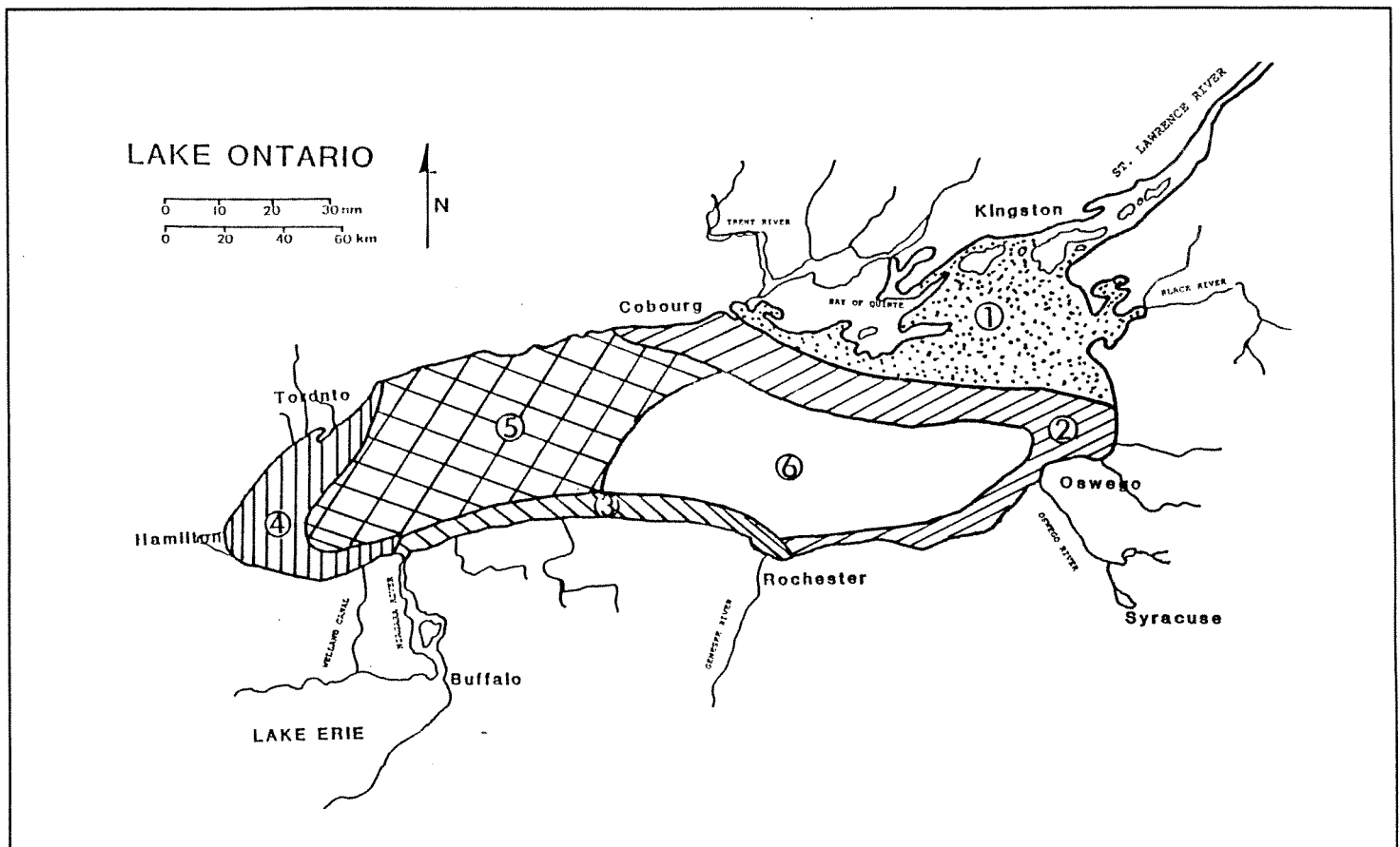


Figure 25. Statistically-derived (discriminant analysis) zonation scheme for Lake Ontario water quality differences within the Basin. From Neilson & Stevens (1986).

into two zones is principally a reflection of upwelling effects along the northwest shore, as it was found that the line of demarcation between the two zones varied from year to year in relation to the predominance of upwelling events encountered during the summer surveys.

Several regions in the Lake exhibited water quality of sufficient contrast to the remainder of the Lake to warrant them being considered as distinct from any of the six zones (Figure 25). These regions were the waters immediately adjacent to Toronto Harbor and Humber Bay, offshore of the Welland Canal, in the vicinity of Oswego Harbor, the entire region of Black River Bay and at the passage leading into Aldolphus Reach, adjacent to the Bay of Quinte. All five regions exhibit signs of serious pollution.

The observed differences between zones, particularly between nearshore and offshore zones, may be due to differences in rates of external loading (e.g., for chloride, sulphate); to a balance between rates of uptake by phytoplankton and internal/external loading (e.g., soluble nutrients); and/or to differences in the timing of seasonal cycles (e.g., phytoplankton biomass indicators, temperature). In reviewing the characteristics of individual zones, as described by Stevens (1987), it is appropriate to discuss them in counterclockwise order, reflecting the general direction of prevailing currents.

Zone 4 received industrial and municipal inputs from Toronto and Hamilton, as well as 85% of the total direct (i.e., not including tributaries flowing into the Bay of Quinte and Hamilton Harbor) discharge from Ontario tributaries. In 1981, of all six zones, this area demonstrated the highest average levels of ammonia, organic nitrogen and nitrate + nitrite, particularly during the stratified period. In addition, moderately elevated concentrations of all phosphorus forms were observed. Of note is the Don River which ranked highest of all tributaries in this

zone for all three nitrogen compounds and Twelve Mile Creek which contributed 4,900 t/yr of total nitrogen (66% of all nitrogen loading). In 1983, ammonia loadings from Hamilton sewage treatment plants totalled approximately 3,090 t/yr resulting in mid-harbor measured water concentrations of 1,130 ug/L. Stevens (1987) also indicated that most of the nearshore waters between Toronto and Niagara-on-the-Lake exhibited signs of enrichment of nitrate + nitrite and organic nitrogen, particularly in the vicinity of Toronto and Hamilton harbors. Relative contributions of total phosphorus to zone 4 were more readily quantifiable. Total tributary TP load, averaged over 1981-82, was approximately 490 t/yr. Again, the Don River exhibited the highest average concentration (391 ug/L) but Twelve Mile Creek showed the largest load (270 t/yr). Contributions from the Main and Humber sewage treatment plants (440 t/yr), however, exceeded the total tributary load. With inclusion of this load to Hamilton Harbor (240 t/yr) and that to Lake Ontario directly from plants at Mississauga, Oakville and St. Catherines, the total load approximated 1,215 t/yr.

Zone 3 displayed substantial gradients in water quality from the Niagara River to the Genesee River. During four surveys conducted in 1981, waters in the western end of zone 3 were found to be dominated by discharge from the Niagara River (Rockwell and Palmer 1984). Murthy, et al. (1985), in developing a transport model of the Niagara River plume, determined that the extent to which the plume dominated waters in zone 3 was related to the prevailing wind direction and the river discharge velocities. During the stratified period in 1981, Rockwell and Palmer (1984) found that the plume was confined primarily to the epilimnion within the study area (i.e., within 6-10 km of the river mouth). Due to reduced vertical mixing, the areal extent of impact of the Niagara River would be even greater. Numerous inputs from unmonitored creeks along the U.S. south shore, as well as the effects of nearshore/offshore mixing, contribute to the pronounced gradients

exhibited by several water quality variables, particularly phosphorus and chloride, in zone 3. During Lake Erie's isothermal periods, resuspension of sediment in eastern Lake Erie results in elevated levels of total phosphorus in the Niagara River and, hence, in zone 3.

Zone 2 exists as a narrow (5-10 km) nearshore band from the Genesee River (Rochester Harbor) to Mexico Bay. From there it appears to follow the circulation pattern discussed by Simons and Schertzer (1985), wherein 90% of transport along the south shore recycles into the midlake region. The apparent transport of zone 2 water to the north shore may be real or it may represent a gradient in response to mixing of waters from zone 1 and zone 5, rather than any physical transport. The two principal sources of input to zone 2 are the Oswego and Genesee Rivers with mean annual flows of 176 m³/s and 95 m³/s, respectively, for 1981 and 1982 combined. With a combined drainage basin area of 19.6 x 10³ km², these two rivers account for approximately 60% of the U.S. Lake Ontario drainage basin area. Flow from these two tributaries alone exceeded that of all Ontario tributaries to zone 4 in 1981 and 1982. Numerous (35) other tributaries also discharge to this zone. Tributary loadings could be calculated for only the Oswego and Genesee Rivers and Sandy Creek, but together they represent 83% of the drainage to zone 2. Average loadings of total phosphorus, nitrate + nitrite, ammonia and total nitrogen from these sources in 1981 and 1982 always exceeded those to zone 4. Despite the higher tributary loadings, however, water quality in zone 2 was almost consistently equal to or superior to that in zone 4 in 1981 and 1982. This may be due to the fact that sewage treatment plant discharges direct to the nearshore waters of zone 4 (17.7 m³/s) were, in 1981, almost four times that discharged to the nearshore waters of zone 2 (4.8 m³/s). For example, Ontario plants contributed an additional 675 t/yr of total phosphorus in 1981 to zone 4 while New York plants contributed only 150 t/yr to Zone 2.

Differences in mixing depth between the two zones would also contribute to water quality differences. The average bottom depth in zone 2 is 50 m compared with 33 m in zone 4. Consequently, during isothermal conditions, the average volume per unit surface area available for dilution of inputs is approximately 55% greater in zone 2. Similar conditions exist during the stratified period when the epilimnion thickness in zone 2 can average 36% to 62% greater than in zone 4.

Zone 1, representing the Kingston Basin, differed from adjacent areas principally with respect to concentrations of soluble nutrients (phosphorus, ammonia, nitrate + nitrite and silica). Onset of the spring diatom growth phase takes place much earlier in the Kingston Basin than other regions of Lake Ontario, generally peaking while the basin is still ice-covered (Munawar and Munawar 1981). Hence, biomass indicators are considerably greater in zone 1 than in adjacent zones and soluble nutrients exhibit greater depletion. The strongest evidence demonstrating the degree of separation between water masses in the Kingston basin and the main Lake is afforded by the respective rates of hypolimnetic oxygen depletion of the two water bodies. The volumetric hypolimnetic oxygen depletion (VHOD) rate in the main body of the lake was 8.4 ug/L/d and 11.3 ug/L/d in 1981 and 1982, similar to VHOD rates reported by Dobson (1985) for the early 1970s. In the Kingston Basin, however, VHOD rates of 76.0 and 71.5 ug/L/d were observed in 1981 and 1982. Furthermore, these rates were not significantly different from those observed over the period 1975 to 1979 (Stevens 1987), indicating that no apparent trend has occurred in VHOD rates coincident with decreasing phosphorus loads.

This region receives significant loadings of phosphorus from the Black River (164 t/yr) and water outflow (export) from the Bay of Quinte (480 t/yr; Minns, *et al.* 1986). This high loading is not reflected in ambient TP levels, however, presumably due to algal uptake, sedimentation of

particulate P (facilitated by basin morphometry), and downstream transport via the St. Lawrence River. The latter is especially significant for export from the Bay of Quinte, as there is a predominantly clockwise circulation pattern around Amherst Island which would serve to deflect inputs from the Bay of Quinte east through the North Channel directly into the St. Lawrence River.

Zones 5 and 6 (Figure 25) represent typical deep water environments for the Great Lakes overall. These zones, primarily because of their extent of water depth as well as large volumes of epilimnetic water during stratified conditions, exhibit good water quality throughout the year. The only characteristic of note would be the increasing nitrogen levels as discussed earlier, which are directly related to decreasing phosphorus concentrations in these offshore environments.

Toxic Materials

The following anecdotal information is important to consider in gaining an appreciation for our awareness of toxic contaminants in Lake Ontario. In the 1969 Lake Ontario Water Pollution Board report, only passing mention was made of persistent toxic substances in Lake Ontario, reflecting the state of knowledge at the time. Our initial awareness of the extent and magnitude of the organic contaminant problem in the Great Lakes was first afforded by Keith (1966) who reported breeding failures in herring gulls contaminated by DDT in Lake Michigan. Gilbertson (1974) also reported several reproductive failures in Lake Ontario colonies of herring gulls, attributed to eggshell thinning which, in turn, was correlated with the DDE content of eggs. During the 1972 International Field Year for the Great Lakes Program (IFYGL), fish, water, sediment, net plankton, Cladophora and benthic fauna were sampled to provide baseline information on the levels of DDT

group pesticides, dieldrin and PCBs (Haile, et al. 1975).

The signing of the 1978 Great Lakes Water Quality Agreement provided considerable impetus to develop more comprehensive monitoring and research programs. Nevertheless, the Toxic Substances Committee, established in 1980 by the International Joint Commission for the purpose of evaluating programs and activities, concluded that the underlying problem was the absence of an overall Great Lakes ecosystem strategy for toxic substances control activities that were being carried out under the various pieces of legislation among the jurisdictions. This problem of fragmentation of toxic substances programs and activities within the Great Lakes basin continues even now and prevents determination of total loadings to the Lake. Work is now underway, however, to develop an international toxics management strategy for Lake Ontario.

Tributary inflow is recognized as one of the largest sources of toxic substances to the Lake Ontario Basin (International Joint Commission 1987b), with the largest contributor being the Niagara River. A number of studies have used the pattern of sediment contaminant concentrations in Lake Ontario to suggest sources of contaminants to the Lake. The Niagara River has been identified as a major conveyance of PCBs, mirex, mercury, chromium, cadmium, vanadium, arsenic, lead, nickel, copper and zinc (Mudroch 1981; Thomas 1983) to Lake Ontario. The river in turn has received the contaminants from a plethora of sources in its drainage basin as well as from the upstream lakes. For example, sediment distribution patterns have identified the Niagara River as the major source of PCBs to the lake (Figure 26). Despite the fact that the Niagara River continues to contribute substantial quantities of DDT and endosulphans to the Lake (50-100 kg/yr

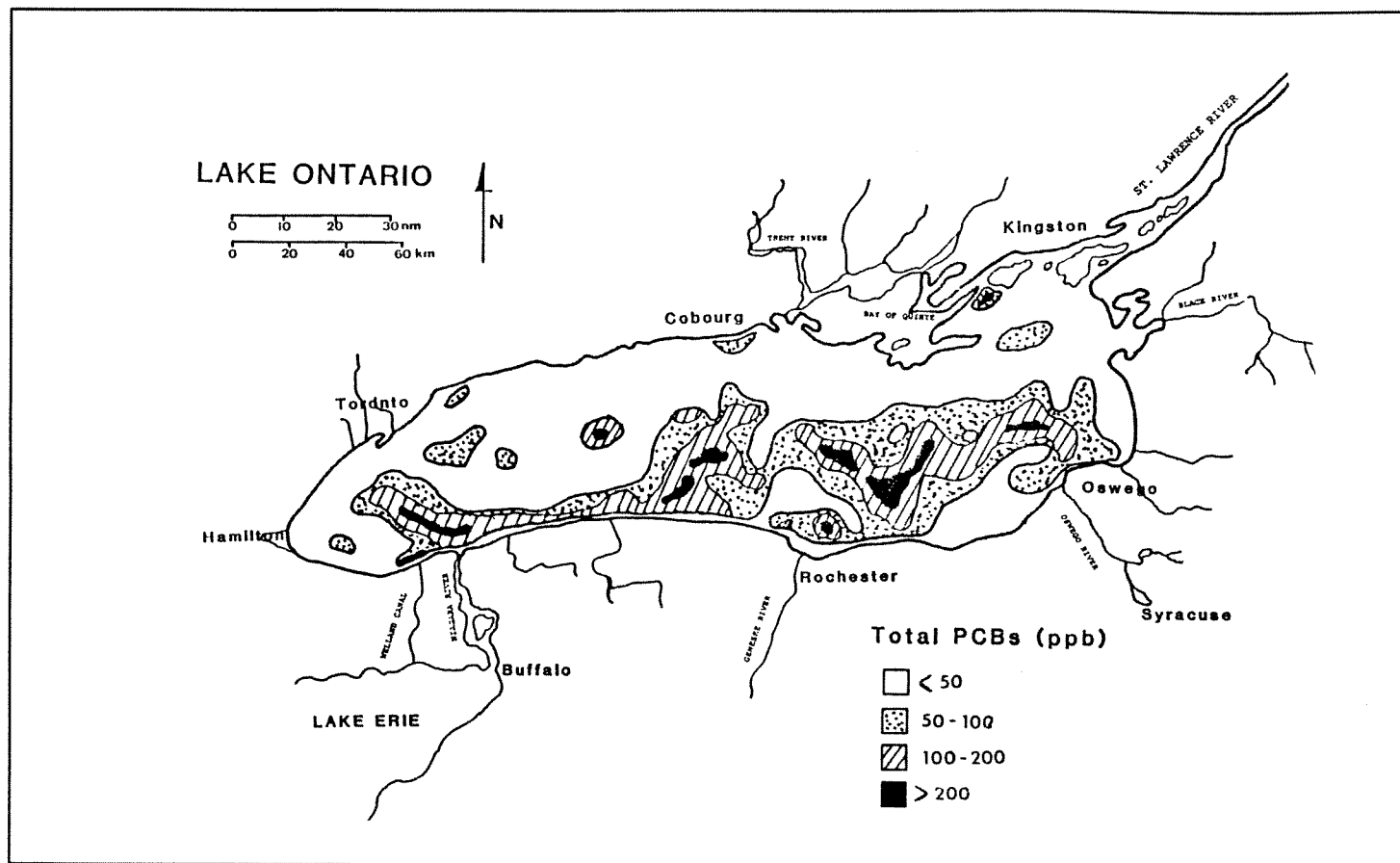


Figure 26. Distribution of PCBs in the Top 3cm of freeze-dried sediment. From Frank, *et al.* (1979).

and 55 kg/yr, respectively; Stevens 1987), no apparent source for either of these compounds could be identified from sediment distribution patterns (Frank, *et al.* 1979).

Atmospheric deposition is also recognized as a major source of some contaminants to the Great Lakes, yet considerable uncertainty remains as to the magnitude of this deposition and the role the atmosphere plays in the recycling of these substances. For Lake Ontario, it has been estimated that relative to other sources of contaminants, the atmosphere contributes 7% to the Lake for such chemical species as PCBs (International Joint Commission 1987b). In contrast, the atmosphere contributes up to 90% of PCBs to Lake Superior.

The role of the Niagara River in contributing to trace organic and metal contamination of Lake Ontario has been thoroughly documented (e.g., Oliver and Nicol 1984). Thomas (1983) showed that the means and standard deviations for several toxic metals and pesticides in surface sediment samples taken from the western basin of Lake Ontario compared well with the means and standard deviations of the same parameters in suspended sediment samples collected from the Niagara River. Routine monitoring of chlorinated pesticides, PCBs and chlorobenzenes in whole water and suspended sediment has been carried out since April 1979 (Kuntz and Warry 1983). One of the more significant findings from studies on the Niagara River was that, except for PCBs, hexachlorobenzene, methoxychlor, mirex and p, p'-DDE, the solute phase was the more important medium for the transport of

organochlorine pesticides (Kuntz and Warry 1983). Kuntz (1984) examined the role of suspended sediment in transport of trace metals to Lake Ontario. He determined that, except for cadmium, copper, and manganese, the bulk of the total trace metal load is associated with the suspended sediment.

It is thought that, with the exception of PCBs, DDT, -BHC and dieldrin, wet deposition contributes 10% to the total atmospheric load (Eisenreich, *et al.* 1981). This signifies that dry deposition and vapor exchange, which cannot be quantified with any degree of certainty, are responsible for the largest proportion of atmospheric loadings to the lake (Stevens 1987). For the compounds examined, however, such as PCBs, total atmospheric load to Lake Ontario is much less than that from the Niagara River.

The fate of trace organic contaminants in bottom sediments is governed by a variety of physical processes including volatilization, sorption, advective and diffusive mixing, sedimentation, resuspension, burial and bioturbation (Eadie, *et al.* 1983). While resuspension is the major process during the non-stratified period, other factors, such as the benthic nepheloid layer, are significant in the redistribution and transport of particulate matter during the stratified period. Using suspended sediment traps in Lake Ontario, Oliver (1986) reported that samples collected from within the nepheloid layer consistently exhibited higher contaminant concentrations than overlying waters for a wide range of chlorobenzenes and organochlorine pesticides. Consequently, this layer is likely a significant factor in the redistribution and transport of particulate matter and associated organic contaminants into depositional basins throughout the entire stratified period. Sediment resuspension, coupled with large-scale horizontal circulation patterns, tends to lead to homogeneity in contaminant levels (e.g. PCBs, Figure 26) in bottom sediments between the three major depositional

basins, Niagara, Mississauga, and Rochester (Stevens 1987).

Recent studies have shown Lake Ontario sediments to be highly contaminated by a variety of organic compounds, some considered specific to the Niagara River. Onuska, *et al.* (1983) identified and, to the extent possible, quantified a number of organic chemicals in sediment cores. The dominant compounds in the alkylbenzene group were xylenes and trimethylbenzenes in the chlorobenzene group di-, tetra-, and hexachlorobenzenes; and in the polynuclear aromatic group naphthalene, chrysene, phenanthrene, fluoranthrene, pyrene, and their corresponding methyl homologues (Onuska, *et al.* 1983). A reasonably consistent spatial distribution was apparent between the different compound groups. Highest concentrations were generally observed approximately 10 km north and northeast from the mouth of the Niagara River while lowest concentrations were consistently observed in the northwestern portion of the Niagara basin and the Whitby-Olcott sill, dividing the Niagara and Rochester basins, a non-depositional area (Onuska, *et al.* 1983). Low concentrations of contaminants in the Niagara Basin appeared to indicate that either little direct deposition from the Niagara River occurs in this area or resuspension processes rapidly redistribute sediment accumulation between the three other major depositional basins. In addition, when trace metal levels in sediment from the depositional basins are compared to those in the nearshore zone and in select harbors, little retention of these contaminants is thought to occur in shallow waters. For example, maximum sediment trace metal concentrations in the Ontario nearshore zone are less than the maximum reported concentrations in the depositional basins (Stevens 1987).

Ambient levels of trace organic (and metal) contaminants in the offshore waters of Lake Ontario are generally the result of anthropogenic loadings and internal release

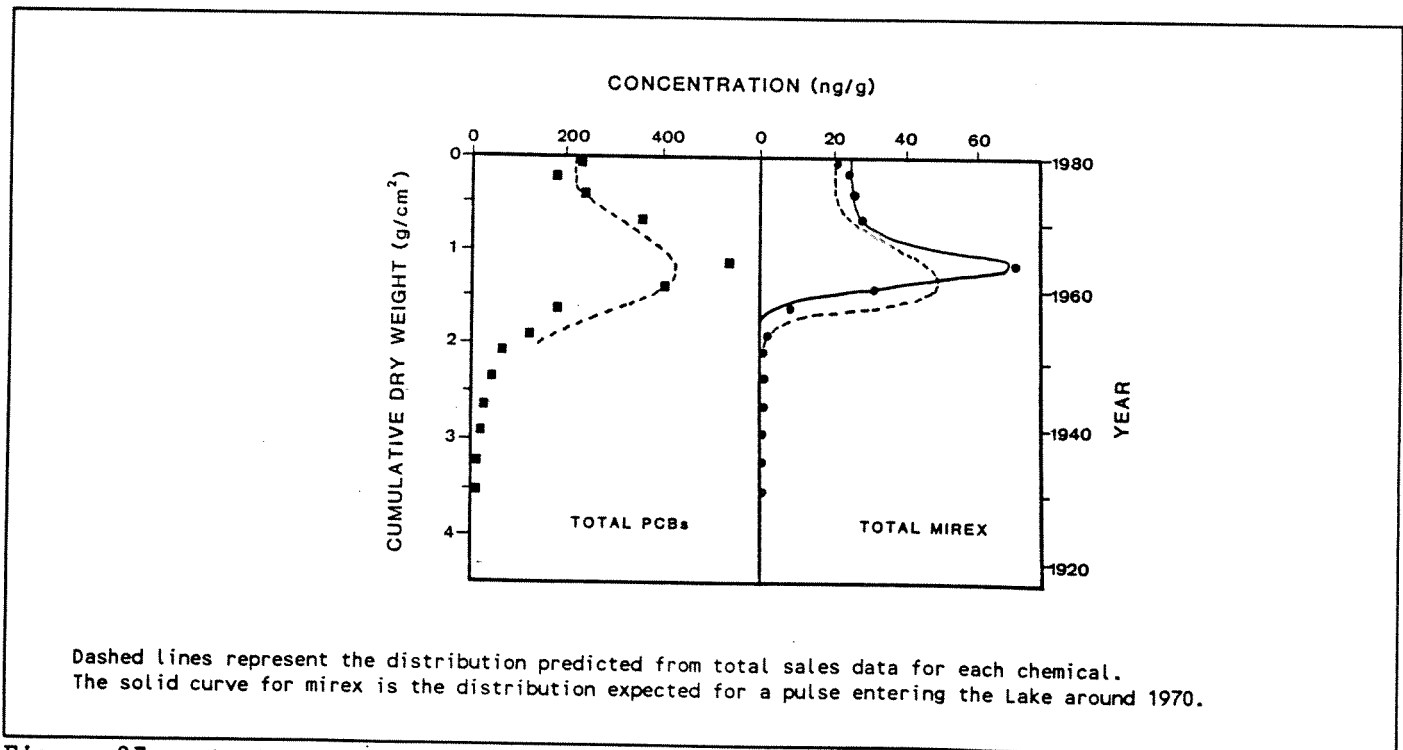
Lake Ontario

from resuspended sediments, ameliorated by sedimentation losses and subsequent burial, downstream transport, biological uptake and, for trace organics, losses by volatilization and/or degradation of compounds. The rapid rate of most of the processes that tend to remove organic contaminants from the water column (e.g., biotic uptake and volatilization) cause ambient concentrations in the offshore waters of Lake Ontario to be extremely low. For example, Oliver (1986) observed that 80-90% of the Niagara River loading of chlorobenzenes to the Lake were lost through volatilization in a relatively short period of time. Despite the obvious gradients displayed by trace metals in surface waters of Lake Ontario in response to nearshore inputs, concentrations of almost all metals are also very low in the offshore water column (Stevens 1987).

Unlike nutrients, no long-term records of organic and metal contaminants exist for Lake Ontario that can reliably be subjected to trend analysis. To assess lake-wide response to remedial efforts at

reducing contaminant loadings, indicator organisms have been used as surrogate measures of contamination and sediment cores have been employed to date the first appearance of some contaminants in the Lake.

To determine trend patterns for organic contaminants for years prior to those provided by biological samples, it is necessary to examine the historical record contained within the sediments. Interpretation of contaminant levels in sediment core samples provides insight into the pattern and magnitude of historical loadings. For this assessment, ^{210}Pb and ^{137}Cs profiles have been used to construct age/depth profiles and to determine sediment accumulation rates. The ability of sediment profiles to assess trends in contaminant reduction was demonstrated by Eadie, *et al.* (1983), who determined the vertical profiles of PCBs and mirex in cores from the Rochester basin (Figure 27). Other examples include chlorobenzene



Dashed lines represent the distribution predicted from total sales data for each chemical. The solid curve for mirex is the distribution expected for a pulse entering the Lake around 1970.

Figure 27. Distribution of mirex and total PCBs in a Lake Ontario sediment core. From Eadie, *et al.* (1983).

concentrations in cores from the Niagara and Rochester basins which showed a sharp drop-off below sediment depths corresponding to the early 1940s (Oliver and Nicol 1982). Concentrations at the 1-2 cm core depth were generally higher than at the core surface (0-1 cm), indicating that chlorobenzene loadings to the Lake are less now than in the recent past.

The PCB profile produced by Eadie, *et al.* (1983) agrees favorably with the more recent work of Durham and Oliver (1983). Accurate sediment dating provided the detailed profile presented in Figure 28, along with limited available sales data. Peak PCB concentrations occurred in the sediments during the years of its peak sales (Durham and Oliver 1983). The sediment pattern in PCBs since 1975, however, does not parallel the trend exhibited by herring gulls and lake trout. This would suggest that the level of detail within the top 5 cm is insufficient to

document recent annual changes in ambient level (Stevens 1987).

Profiles of mirex (Figure 29) and chlorobenzenes, chlorotoluenes, octachlorostyrene, and hexachlorobutadiene all exhibit a pattern similar to that exhibited by PCBs. Peak concentration generally occurred during the early 1960s with fairly rapid declines occurring shortly thereafter, both in response to declines in production and, presumably, in response to more stringent effluent controls. Most investigators conclude that pollution of Lake Ontario with chlorinated hydrocarbons from the Niagara River was still occurring, but the worst contamination occurred in the 1960s. Mudroch (1983) has reported on meta concentrations in cores from nine sites in

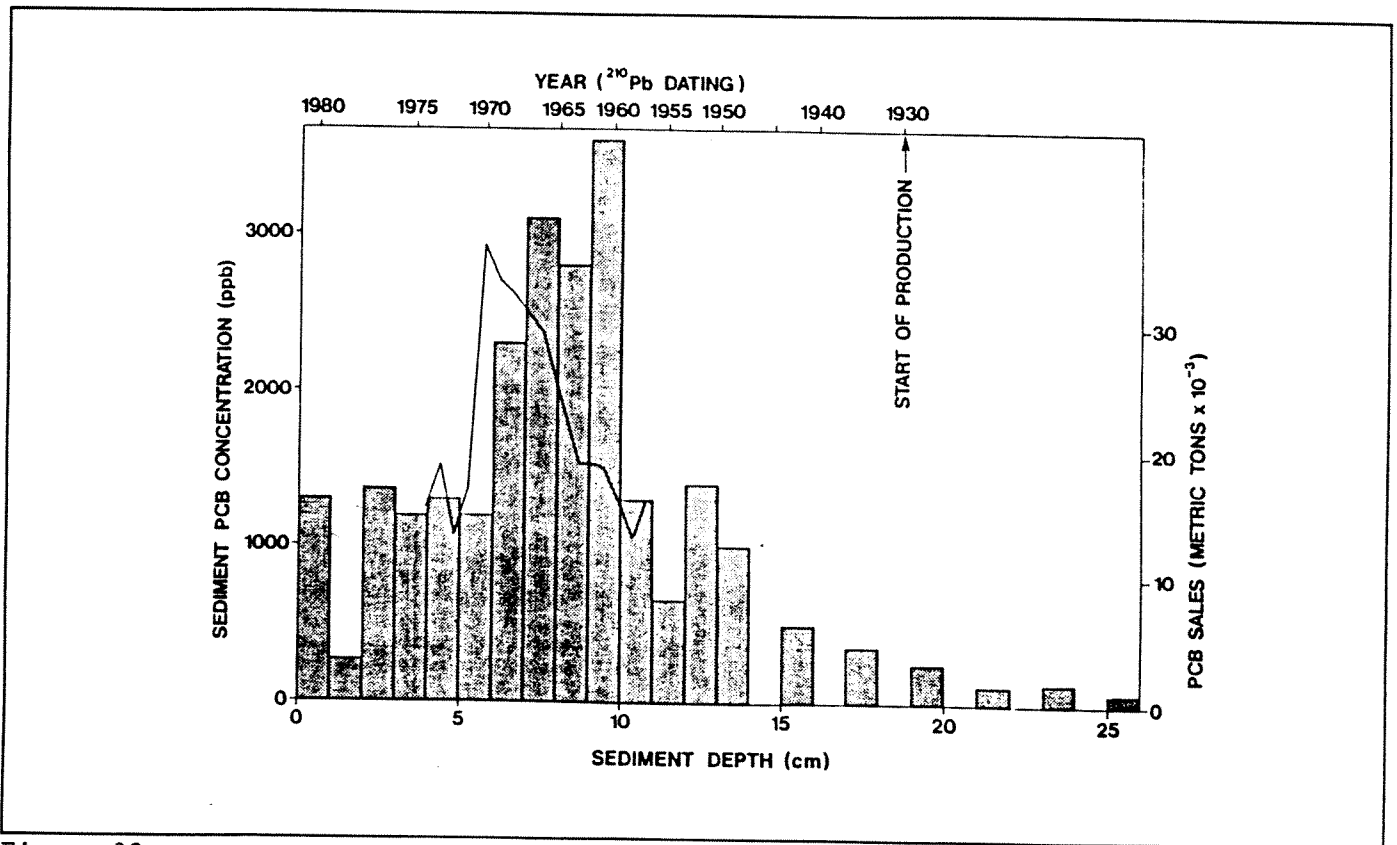


Figure 28. Total PCBs versus sediment depth in the core (bottom axis) and sediment age (top axis). Sales data for PCBs are superimposed (right axis) with a line plot. From Durham and Oliver (1983).

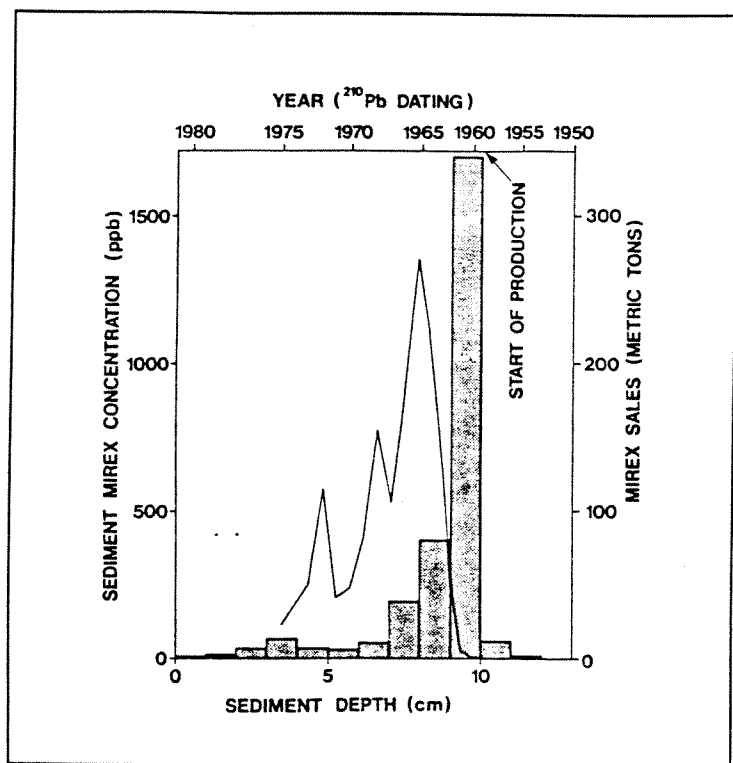


Figure 29. Mirex concentration versus sediment depth in the core (bottom axis) and sediment age (top axis). Mirex sales data are superimposed (right axis) with a line plot. From Durham and Oliver (1983).

coho salmon (*Oncorhynchus kisutch*) from all the Great Lakes, levels of PCBs, DDT, mirex, photomirex, hexachlorobenzene and mercury were highest in those fish taken from Lake Ontario (DeVault and Weishaar 1982; Clark, *et al.* 1984). In 1980, total PCBs averaged 2,900 ng/g wet weight. Total DDT averaged 800 ng/g with 85% as p, p'-DDE. PCB levels as high as 10,000 ng/g have been reported for lake trout fillets, although average concentrations for most sites ranged from 2,000 to 3,000 ng/g. These high contaminant levels in fish have resulted in consumption advisories for Lake Ontario.

Whittle and Fitzsimons (1983) used a number of species representative of several trophic levels to demonstrate the influence of the Niagara River on residue levels in Lake Ontario biota. They found that concentrations of PCBs, DDT and p, p'-DDE were significantly higher in coho salmon, rainbow smelt (*Osmerus mordax*) and the benthic amphipod (*Pontoporeia affinis*) from the western basin of Lake Ontario than the eastern basin of Lake Erie. In addition, mirex was detectable in all three species from Lake Ontario but was undetected in those from Lake Erie.

the western basin of Lake Ontario. She observed a significant decrease in mercury concentration in the upper layers of the cores, suggesting a significant decrease in mercury loading from the Niagara River during the past 20 years (Figure 30). Similar but smaller decreases were observed for lead, zinc, chromium, and copper, indicating that remedial measures are reducing inputs. Of the metals measured, mercury concentrations were most elevated above background levels. Despite the observed decline, elevated concentrations (up to 7 mg/kg above background levels) remained in the top 10 cm of sediment at eight sampling sites.

The longest record for contaminants in Lake Ontario biota is provided by the monitoring of herring gulls. Herring gulls are a top predator in the food web and, although usually opportunistic in feeding habits, they are primarily piscivores. Concentrations of all six major organochlorine residues in herring gull eggs from Lake Ontario colonies have shown declines since 1974 (Table 6). As an example of changes, levels of DDE and PCBs have decreased 79 percent and 78 percent respectively, between 1974 and 1983.

As a consequence of significant contaminant loadings to the lake, body burdens of many contaminants in Lake Ontario biota are the highest of all the Great Lakes. For example, based on analysis of dorsal fillets from 3 year old

Trends in trace organic contaminants in lake trout (*Salvelinus namaycush*) generally parallel those of the herring gulls, although levels exhibit considerable year-to-year variation. Comparison of contaminant levels in a single age class, and thus a similar exposure period, over a long-term period, considerably reduces the

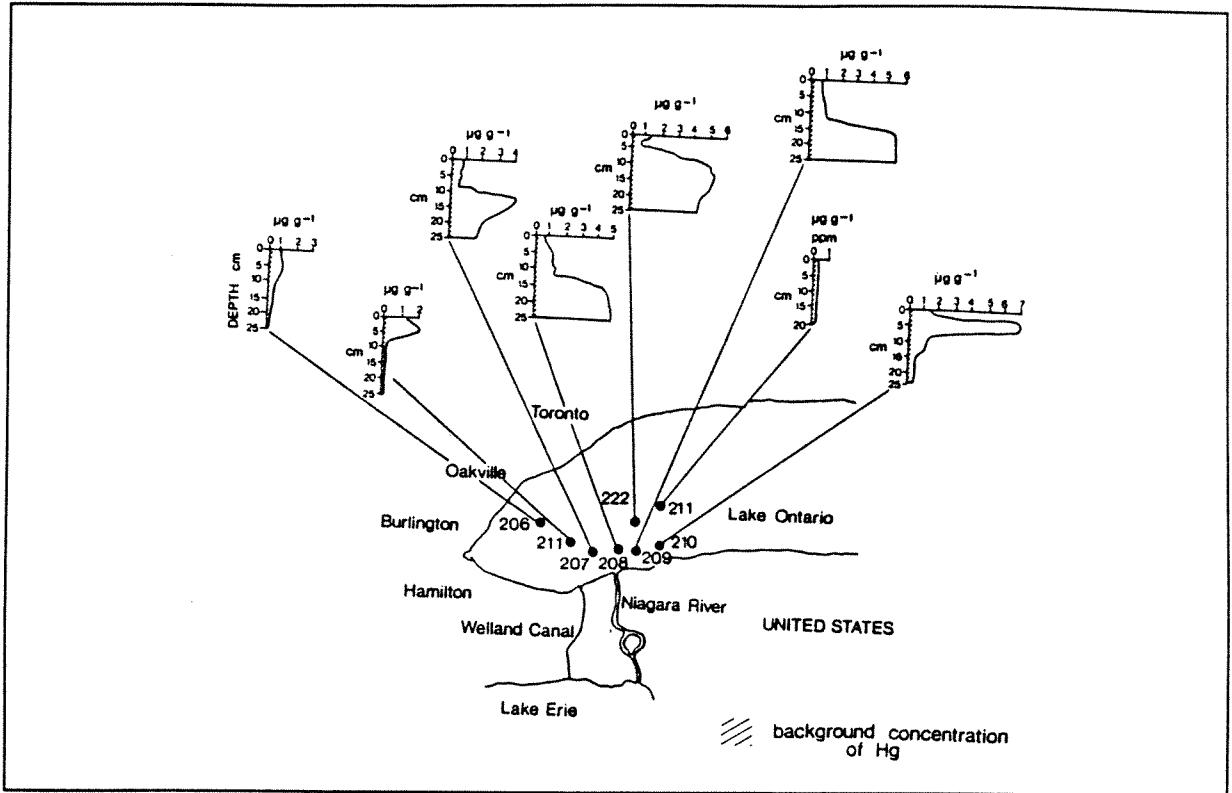


Figure 30. Concentration profiles for mercury (Hg) in sediment samples taken from the western basin of Lake Ontario. From Mudroch (1983).

Table 6. Mean levels (ug/g wet weight) of 3 major organochlorine residues in herring gull eggs from annual monitored colonies in Lake Ontario, 1974-1983.

COLONY LOCATION & YEAR	N	DDT	MIREX	PCBs
MUGG'S ISLAND				
1974	9	1.20	7.4	170
1975	10	0.13	3.4	110
1977	10	0.11	2.1	87
1978	8	0.10	1.4	75
1979	10	0.08	1.8	76
1980	9	0.10	1.7	60
1981	10	0.08	2.5	72
1982	9	0.11 ¹	3.7	64
1983	11	0.02	1.4 ²	39
SNAKE ISLAND				
1974	10	1.00	6.6	140
1975	10	0.24	6.0	180
1977	10	0.11	2.9	120
1978	10	0.07	1.7	71
1979	10	0.06	2.0	63
1980	10	0.14	1.6	53
1981	10	0.11	2.8	86
1982	10	0.10	2.5	61
1983	11	0.03	1.5 ³	46

¹ Significantly different from 1981 value (according to t-test, P < 0.05).
² Number (N) sampled equalled 10.
³ Number sampled equalled 9.

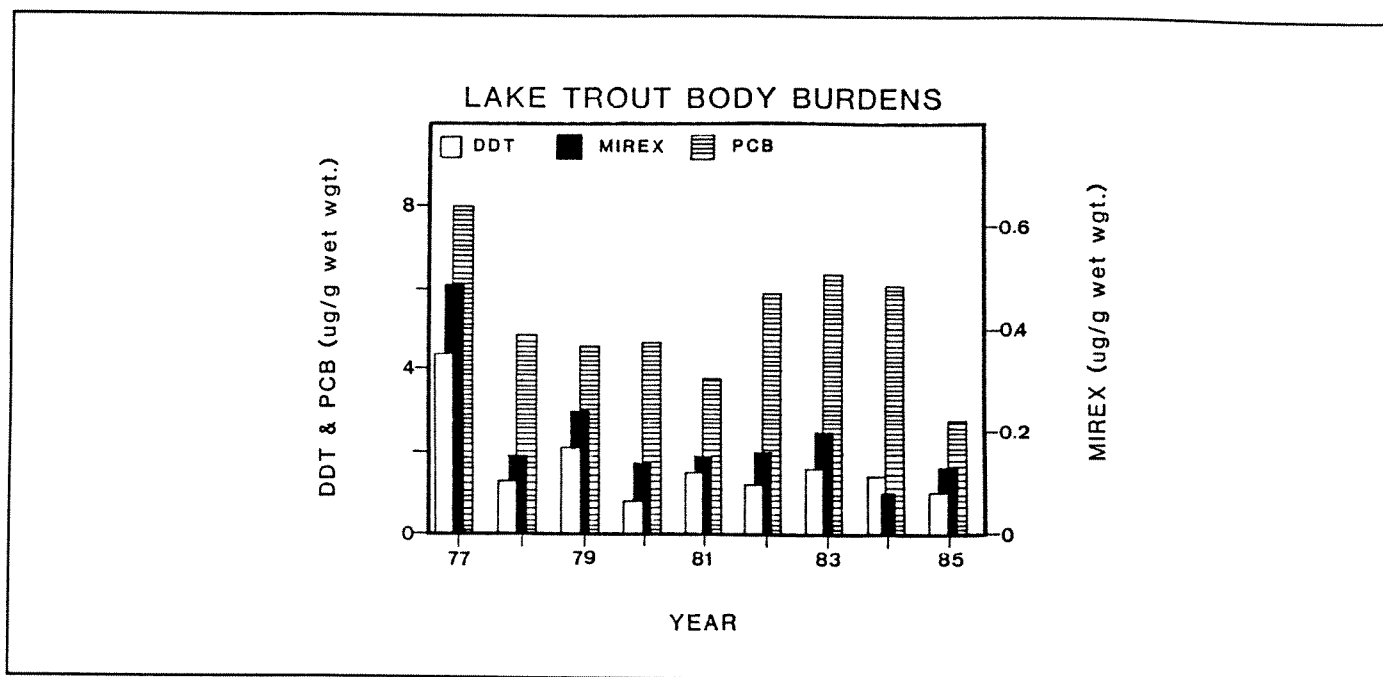


Figure 31. Body burden changes in age 4+ lake trout from Lake Ontario for DDT, mirex and PCB concentrations. From International Joint Commission (1987a).

between-year variation. Therefore, age 4+ year lake trout samples were utilized to describe temporal contaminant trends (Figure 31). PCB concentrations declined steadily from 1977 to a significantly lower level ($P < 0.001$) in 1981. Whole fish PCB levels in age 4+ lake trout increased significantly (67.3%) from 3.67 ug/g in 1981 to 5.87 ug/g in 1982 and subsequently to 6.44 ug/g in 1983 samples. Similarly levels of DDT and mirex in 4+ lake trout all increased significantly in 1983 as compared to levels reported in preceding years. With the exception of the 1984 minimum value, annual mean mirex levels in lake trout have not changed significantly ($p < 0.05$) from 1980 to 1985. This suggests a constant input or a recycling of mirex within the Lake Ontario ecosystem, with probably the latter being the case.

Changes in contaminant levels in rainbow smelt reflected those of lake trout (Figure 32), with PCBs, DDT, and mirex all exhibiting increases in 1983. As with lake trout, levels of most routine contaminants determined in rainbow smelt samples collected in 1985 were the lowest recorded over the nine year period of the survey.

The decline for contaminants such as PCB and DDT has been inconsistent, however, with significant fluctuations during the period 1979 to 1981. Conversely, levels of mirex and Hg measured in smelt samples have shown a consistent decrease, with a significant ($P < 0.05$) decline over the latest two years of the survey.

Data from the 1972 International Field Year for the Great Lakes (Haile, *et al.* 1975) and similar historical surveys of Lake Ontario contaminant levels (Reinert 1970) provide a reference point for determining the significance of current data. Consistent decreases in DDT, PCBs and dieldrin levels measured in samples of sculpins and smelt over almost two decades were noted. Only dieldrin levels in sculpins increased slightly from 1972 to 1982 while concentrations in smelt remained constant.

Still further down the food chain a comparison can be made in the invertebrate prey in Lake Ontario between 1972 (Haile, *et al.* 1975) and more recent (1983) data collections (M. Whittle, CCIW, Burlington, personal communication, 1985). Significant

increases occurred for DDT, p, p'-DDE and dieldrin residues identified in benthic invertebrate (*Pontoporeia*) samples collected one decade apart. Conversely PCB residues decreased by more than 500%. Haile et al. (1975) indicated that benthic invertebrate PCB levels in the western basin of Lake Ontario were approximately four times greater than those determined for the other two stations located off Oswego and Rochester. These levels were similar to those found in *Pontoporeia* from western Lake Ontario stations in 1983.

Reproductive success in both herring gulls and cormorants has increased in recent years contrasted to the early 1970s (Table 7). Contaminant levels in herring gull eggs between 1974 and 1986 for the two Lake Ontario colonies showed significant declines for DDE, mirex, HCB and PCB, ranging from 57 to 73% at both colonies, since 1981. Only heptachlor epoxide has not decreased, and no compound increased significantly from 1981 to 1986. There has been a population explosion of healthy herring gulls in recent years. This correlates with efforts to control the entry of persistent toxic substances into Lake Ontario from point sources, agricultural pesticide use, and solid waste (garbage) dumps and, thereby, reduces the availability of contaminants to the gulls.

Toxic contaminant levels (PCB) were also monitored in terrestrial mammals in close proximity to waters containing high toxic loads. PCB levels are consistently greatest in mink in close proximity to known sources of elevated PCB levels in fish, (e.g., Lake Ontario and the Hudson River; Foley, et al. 1988). The PCB levels measured in wild mink were similar to those which caused reproductive problems in laboratory feeding studies on this species. In attempting to gain a better understanding on the source of contaminants to the mink, Foley, et al. (1988) examined contaminant relationships of this animal to those of fish. Significant correlations between fish and mustelids were observed, indicating a relationship between the

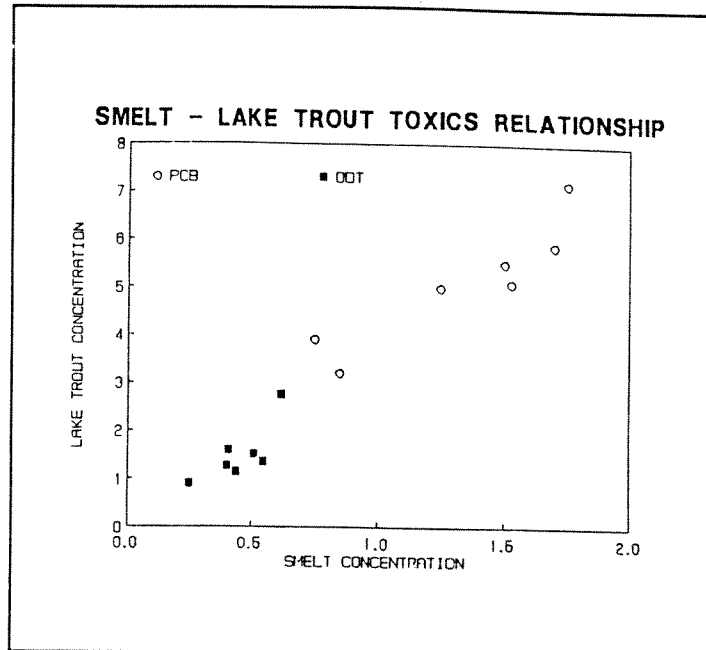


Figure 32. Relationship between levels of PCBs (ug/g) and DDT (ug/g) in Lake Ontario rainbow smelt and lake trout. From International Joint Commission (1987a).

accumulation of PCB and DDT in mustelids and levels of these chemicals in potential fish food source.

The case history of mirex contamination in Lake Ontario provides one of the clearest examples of fate of a toxic contaminant that emanated from human abuse of the environment. Mirex is a chlorinated hydrocarbon used as a pesticide for controlling fire ants and also employed as a fire retardant. Mirex is hydrophobic and hence tends to accumulate in fatty tissues at much higher concentrations than found in water. Bioconcentration occurs with highest levels found in top predator (Scudato, et al. 1986). Photomirex is produced from mirex as a result of a light induced dechlorination at a single position (Mudambi and Hassett 1986).

Mirex contamination was first reported in fish from the Bay of Quinte (Kaiser 1974). In 1976, New York State announced a fisheries ban and the Province of Ontario issued fish consumption

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Table 7. Reproductive success (number of young raised/pair of nesting adults) of herring gulls and double-crested cormorants on Lake Ontario islands and in Canadian waters. From Weseloh and Struger (1985).

YEAR	HERRING GULLS			CORMORANTS	
	MUGG'S ISLAND	SCOTCH BONNET ISLAND	SNAKE ISLAND	CANADIAN ISLAND	WATERS
1954					> 0.12
1957					0
1971					0
1972		0.12		0.18	0
1973				0.12 ¹	
1974					0
1975		0.15			0
1976					0
1977	1.52	1.10		1.01	0
1978	1.47	1.01		0.86	0.13
1979	1.56			1.60	1.25
1980				1.49	1.30
1981	1.40	2.14		1.73	1.45
1982					1.45
1983				1.34	
1984	1.17				

¹ Success determined on nearby West Brothers Island.

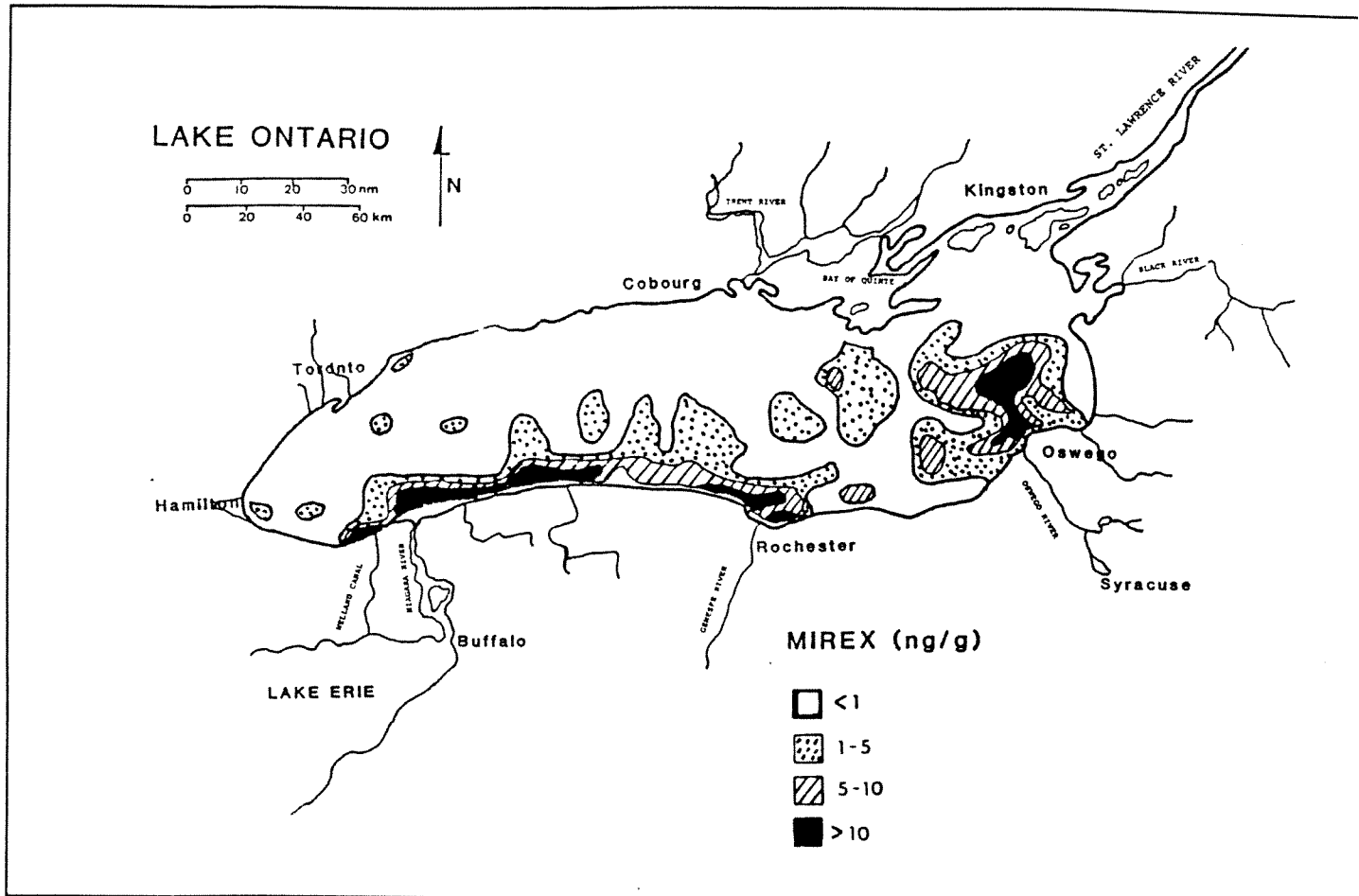


Figure 33. Distribution of mirex in the surficial 3cm of sediment in Lake Ontario. From Holdrinet, *et al.* (1979).

guidelines due to mirex contamination. At this time, there were no known sources of mirex in the Lake Ontario basin. In July 1976, the Pollution from Land Use Activities Group (PLUARG) presented preliminary results to the International Joint Commission indicating the presence of mirex in a number of bottom sediment samples collected in 1968 from Lake Ontario. Subsequent interpretation and confirmation of these data by Holdrinet, *et al.* (1978) clearly identified the Niagara River as the principal source of mirex to Lake Ontario (Figure 33), with sediment concentrations in excess of 10 ng/g extending along the south shore eastward from the Niagara River to beyond the Genesee River (Holdrinet, *et al.* 1978). There was a second, less extensive, area of contamination identified in the Rochester

basin, extending northwest from the Oswego River, suggesting an additional source within its watershed (Figure 33; Holdrinet *et al.* 1978). The majority of bottom sediment samples in the rest of the lake exhibited no detectable levels (i.e., less than 1 ng/g) of mirex during the measurement periods.

These results, in conjunction with more intensive sampling in 1976 (Holdrinet *et al.* 1978) and related investigations by the New York Department of Environmental Conservation, identified two industrial sources of mirex: Hooker Chemical on the Niagara River and Armstrong Cork on the Oswego River. During the production and processing of mirex at Hooker Chemical, it was estimated that 688 kg was lost over the period 1959 to 1976, leading to extensive

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Table 8. Distribution of photomirex (ug/g) in Lake Ontario. From Flint, *et al.* (1987).

	N	PHOTOMIREX		MIREX	
		MEAN	RANGE	MEAN	RANGE
SEDIMENTS	13	0.01	0.01 - 0.1	0.7	0.01 - 3.5
BENTHOS	51	1.9	0.3 - 29.6	4.9	0.4 - 29.6
AMPHIPODS	19	3.7	0.4 - 20.4	5.6	0.6 - 34.6
ALGAE	53	2.1	0.1 - 9.2	2.0	0.1 - 9.0
ZOOPLANKTON	52	3.2	0.3 - 18.4	3.5	0.5 - 21.8
ALEWIFE	40	92.5	5.0 - 73.3	133.3	5.9 - 594.0
YELLOW PERCH	40	29.9	1.8 - 73.3	55.7	2.3 - 114.0
BROWN TROUT	72	170.7	102 - 750	297.0	58 - 1,213

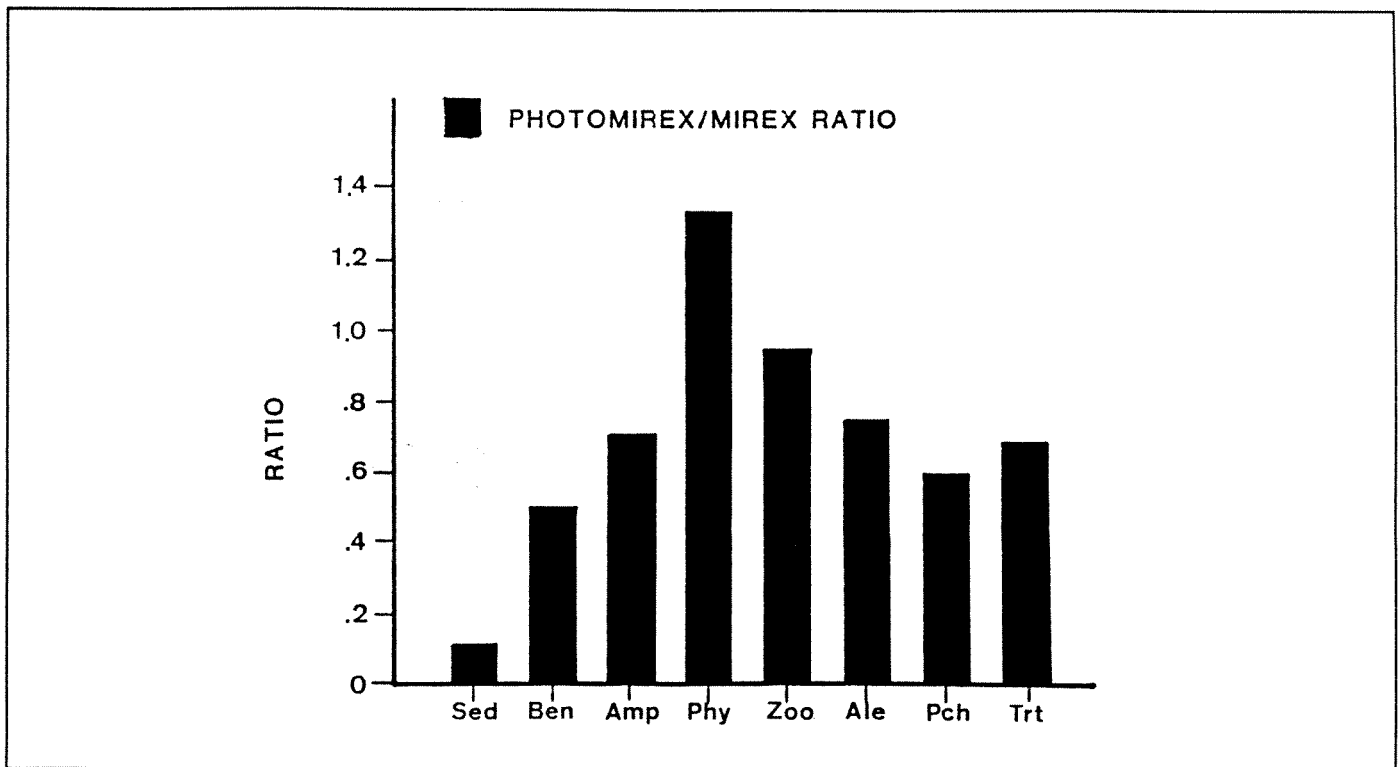


Figure 34 Photomirex/mirex ratio distribution in Lake Ontario sediments, phytoplankton, invertebrates, and fish. From Flint, *et al.* (1987).

contamination of the lake. Contamination from the Oswego River, in contrast, was attributed to a short-term, substantial loss occurring in the early 1960's, at an industrial plant located 14 km upstream of the river mouth (Holdrinet, et al. 1978).

Although production of mirex at Hooker Chemical ceased in the summer of 1976, it continues to be detected in suspended sediment samples taken at the mouth of the Niagara River (Kuntz and Warry 1983). During the period 1982 to 1984, it was estimated that the annual load of mirex to Lake Ontario from the Niagara River was 6.2 kg/yr. The situation in Oswego River appeared similar. Scrudato and DelPrete (1982) conducted an extensive sediment sampling program in 1979 and found that the Oswego River and Harbor were still highly contaminated with mirex. Highest concentrations, 1,290 to 1,834 ng/g, were observed immediately downstream of the Armstrong Cork Company discharge. Levels in the harbor, as high as 48 ng/g, indicated that mirex-contaminated sediments were continuing to accumulate within these protected waters. Lower concentrations (20 ng/g) were found in the upper 3 to 4 cm of sediment at two offshore, deepwater stations, suggesting that decreasing amounts of mirex were being deposited in the Lake. Scrudato and DelPrete (1982) nevertheless suggested that mirex could continue to be available to lake biota for many years to come.

Both mirex and photomirex have been shown to accumulate in tissues of higher organisms, particularly fish (Scrudato, et al. 1986). Highest concentrations were observed in fish with photomirex ranging from 92-170 ug/kg and mirex ranging from 55-257 ug/kg (Table 8). Lowest concentrations were seen in sediments and smaller size classes. For example, photomirex means of 0.1 ug/kg were observed for sediment and 2.07 ug/kg for phytoplankton while corresponding mirex

means of 0.68 ug/kg for sediments and 1.97 ug/kg for phytoplankton were noted.

It was suggested that the ratio of photomirex to mirex may serve as a tracer for movement of this contaminant through the Lake Ontario food web (McDowell, et al. 1986; Flint, et al. 1987). The ratio of photomirex/mirex (PM/M) was lowest in sediments (Figure 34) and significantly higher in all organisms (Flint, et al. 1987). Phytoplankton exhibited the highest PM/M ratio, consistent with their high exposure to light. In contrast, zooplankton exhibited a significantly lower ratio ($P < 0.05$, ANOVA), suggesting some assimilation of food which had not been in the euphotic zone for a long period (i.e. resuspended benthic sediments). All three species of fish examined (alewife, yellow perch, and brown trout) showed PM/M ratios not statistically different from that of zooplankton (Figure 34). This could be expected on the basis of food sources in the trophic web. Yellow perch and alewife diets were dominated by zooplankton, and brown trout fed almost entirely upon alewives. These results suggested that little fractionation of mirex and photomirex occurs in passing through the food web.

The tracing of the PM/M ratio through the Lake Ontario food web, has provided evidence for the long-term fate of toxic substances in this system. What is most interesting is that although the mirex is originating from the sediments at present, because external inputs of mirex have not occurred for years, the majority of transfer of mirex in the food web is occurring through pelagic routes, as suggested by higher concentrations of photomirex in the measured ratios. The tracing of this contaminant ratio further illustrates the complexity of transfer for these toxic materials in general within the Ontario ecosystem and very nicely demonstrates their long-term persistence in the system.

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CHAPTER 6

INTEGRATION OF SYSTEM TRENDS

In light of the ever-increasing demand and multiple use of Lake Ontario natural resources, a clear understanding of the complex biological and physiochemical interactions among and between ecosystem components is vital. Basic research on complex lacustrine interactions still falls short of providing a sound scientific basis for effective management of Great Lake systems, including problems that relate to pollution management and ecosystem rehabilitation. Instead of focusing upon a single chemical or a particular fish species, we must begin to consider how the entire ecosystem works, how components are interconnected and affect one another, and then attempt to preserve the quality of natural resources using this holistic knowledge.

The most important questions needing answers by environmental managers are fundamentally interdisciplinary in nature. Before we formulate good policy, however, we need research that goes beyond the identity of problems and defines processes that can magnify these problems over the long term (Loucks 1985). Causal mechanisms should be the basis for research questions addressing the level of certainty needed to understand processes in the Great Lakes (Carpenter, et al. 1985). Only through these approaches will we be able to make predictions about the future which can then be used to formulate policy for management. Until the interactions within the various components of an ecosystem like Lake Ontario are fully understood, however, we must proceed cautiously with respect to decisions and actions that might be based upon over-simplified concepts. Taking a holistic, ecosystem approach (Risser 1985) to evaluating Lake Ontario natural resource dynamics requires a change in focus rather than a change in methodology.

In evaluating processes that are important in influencing the dynamics of a system such as Lake Ontario, researchers typically look at temporal averages. These time scales, which can be viewed as a "snapshot" of the system being studied, may not reveal important but transient responses by the ecosystem. For example, it is not reasonable to expect today's algal populations to necessarily correlate with today's biomass at other trophic levels, but rather, algal production today may be the result of dynamics at zooplankton or fish levels that occurred in previous days or weeks. In order to overcome the inadequacies of correlation studies, there must be recognition that the kind of research required to understand a Great Lake such as Lake Ontario, must not consist of monitoring, surveillance, and inventory studies alone. Research, instead, must be organized around questions and hypotheses which foster both the expansion of ecological theory and the solution of problems of environmental resource management (Callahan 1984).

The question here is, where do we proceed from this point forward? At what state is our scientific knowledge on Lake Ontario and what level of integration can we achieve with our present understanding, that will begin to address the complex problems related to system management?

Anthropogenic Change

Analyses of chemical loading data and analysis of chemicals in the water and sediments of Lake Ontario indicate that great improvements have been made in slowing anthropogenic degradation. It is widely recognized that for years Lake Ontario was probably the most heavily

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impacted of the Great Lakes from the cumulative effects of all forms of pollution. Although Lake Erie was considered "dead" in the early 1970s, this Great Lake was mostly affected by the excessive loading of nutrients beyond its capacity to assimilate these nutrients. In contrast, Lake Ontario also was classified eutrophic and its loadings were just as great as Lake Erie (Glooschenko, *et al.* 1974), but the greater volume of Ontario softened the impacts of nutrient inputs. The eutrophic status of Lake Ontario, coupled with the excessive input of toxic contaminants in the last four decades, has resulted in probably the worst water quality of all the Great Lakes when all forms of pollution are combined. There are a number of reasons for the degradation of this Great Lake and there are also a number of reasons for reversal of these trends, all of which need to be considered together in order to understand why the changes have taken place.

First and foremost, Lake Ontario is the lowermost of the five Great Lake basins and thus receives the combined outcome of most anthropogenic impacts realized on any one of the other basins individually. Lake Ontario because of its downstream location,

reflects the composite of environmental insults levied against any of the other Great Lakes. In addition to its downstream location, in relative terms Lake Ontario has the largest watershed area to lake surface area ratio of any of the other lakes (Table 9). Therefore, the cumulative effects from development activities in the drainage basin of Lake Ontario have a much greater chance of impacting on the water quality of this Lake than a similar amount of areal activity in any of the other Great Lake drainage basins.

Coupled with its downstream location and greater potential for watershed effects, as described above, Lake Ontario has also had the longest history of cultural development in its basin. Schelshe, *et al.* (1986) for example, through paleolimnological studies were able to demonstrate that Lake Ontario was affected by nutrient enrichment about 100 years before comparable changes were observed in Lake Michigan. It is important to note, however, that demographic trends for major industry in the basin over the last 40 years (Figure 3) have shown a significant downward pattern. For example, industrial manufacturing which relies

Table 9. Ratios of terrestrial watershed to lake surface areas for the five Great Lakes¹.

LAKE SUPERIOR	1.6
LAKE MICHIGAN	2.0
LAKE HURON	2.2
LAKE ERIE	2.3
LAKE ONTARIO	3.2

¹ Data for calculations obtained from Herdendorf, *et al.* (1987).

heavily upon water for processing and waste disposal is steadily being replaced in the Lake Ontario watershed with more service-oriented industries which are much less harmful to the environment.

Another factor that has had a significantly positive effect upon Lake Ontario water quality in recent years has been the binational enactment of the Great Lakes Water Quality Agreement. Most important through this process has been the recognition that, in order to solve water quality problems in any of the Great Lakes, including Lake Ontario, we can no longer attempt to manage the resources of nature, but rather must focus our attentions on managing the activities of people. In essence, the Canada-U.S. Great Lakes Water Quality Agreement, signed in 1972 and amended in 1978 and 1987, is the principal vehicle for ensuring a coordinated, binational approach to water quality management in the international Great Lakes Basin. Its purpose is to "restore and maintain the chemical, physical, and biological integrity of the Great Lakes Basin ecosystem." This is to be accomplished by developing programs, practices, and technology necessary for a better understanding of the Basin ecosystem and by eliminating or reducing "to the maximum extent practicable" pollutant discharges into the Great Lakes system. It is this agreement, coupled with demographic changes, that has brought about much of the improvement in water quality within the Lake Ontario Basin that, for so many years, had deteriorated through much of the historical development summarized previously.

Large Scale Limnological Patterns

It is generally recognized now that data on pattern and control of inorganic and organic material movement through ecosystems like Lake Ontario are required to clearly understand system function and provide capabilities to predict future response to change. Material transport, whether it be inorganic nutrients, organic

carbon, or toxic contaminants, and energy flow dynamics, that are the basis of biological productivity, should be the organizing foci for scientific synthesis in support of decision making in Lake Ontario.

Water quality changes have been found to be occurring in Lake Ontario at rates and in ways that are different from what would be expected. A typical example is the observation that spring surface nitrate and nitrite continues to exhibit a significantly increasing linear trend of 9.18 ug/L/yr as determined for the period 1968 to 1986. This represents an annual rate of increase of 15×10^3 t/yr in nitrogen. Whether these concentrations can be attributed to loading impacts or in-lake processes is not clear. The increasing nitrate concentration in Lake Ontario caused the phytoplankton in 1982 to be phosphorus deficient; whereas, 10 years previously, nitrogen was most likely limiting the summer biomass of phytoplankton (Lean, *et al.* 1987).

Lean and Knowles (1987) observed that seasonal changes in concentration of the various nitrogen fractions in the lake were related to nitrogen flux rates from one form to another. For example, ammonium uptake was observed principally in chemoautotrophic nitrifying bacteria rather than phytoplankton. Nitrate, rather than ammonium, appeared to be the principal form of nitrogen used for algal growth in Lake Ontario. Quantities of nitrate used for protein synthesis were related to seasonal nitrate declines and provided an upper limit for protein available to support the food web. Lean and Knowles (1987) further indicated that seasonal increases in ammonium and nitrite were related to rates of zooplankton excretion.

Instead of being restricted to the metalimnion as in 1966 and 1967 (Dobson 1985), elevated nitrite concentrations now extended to the surface. Unlike trends in 1972 (Stadelman and Frazer 1974) and before, nitrate concentrations now remained above 130 ug N/L (Stevens and Neilson 1987). As such, nitrite utilization would

be reduced in the surface waters. Previously, when nitrate levels were low (less than 5 ug N/L), nitrite uptake would probably outweigh its production. Presently, however, denitrification and nitrogen fixation are not judged to be important processes in supporting algal growth (Lean and Knowles 1987) in Lake Ontario.

While at present, the effect of continued loadings of phosphorus and nitrogen to Lake Ontario, in relation to long-term algal productivity cannot be determined, 1985 mean summer chlorophyll *a* in the offshore region of the epilimnion was the lowest observed (2.5 ug/L) for the period 1975 to 1985 (Neilson and Stevens 1987). And yet very little significant change has occurred within the zooplankton population of the Lake (Johannsson 1987). Johannsson as well as Taylor, *et al.* (1987) suggest that a lack of change in zooplankton communities may be the synergistic outcome from two independent factors working in Lake Ontario.

According to Johannsson's (1987) synopsis, the present examination of total zooplankton abundance and community composition indicates that there has been no consistent change in either parameter. Neither reductions in phosphorus loadings to the lake nor the introduction of large numbers of top predators has exerted a detectable effect on the zooplankton community to date. The zooplankton community is buffered from these perturbations by trophic levels both above and below. These trophic levels should show more pronounced responses to the perturbations than the intermediate zooplankton level (McQueen, *et al.* 1986). Yet there is little evidence of marked changes even in the adjacent trophic levels at the present time. The chances are good that, with continuing application of perturbations to the Lake Ontario ecosystem, changes in zooplankton may well occur in the near future. At present these populations appear to be a good surrogate for measuring total Lake response to continued nutrient abatement, as well as

effects on the food web from introduction of large predator biomass.

Focusing on another component of the ecosystem, Scavia (1979) concluded that in Lake Ontario, rapid internal recycle rates, especially due to zooplankton dynamics during the summer, provided sufficient phosphorus to maintain moderate phytoplankton production. He observed that internal recycling rates of phosphorus were much higher than rates of external loading and/or influx from the hypolimnion. IFYGL measurements suggested that, for both phosphorus and nitrogen cycles, the nutrient pools within Lake Ontario were not sufficient to maintain observed primary production (Stadelmann and Fraser 1974). Thus, it was concluded that there must be considerable recycling of nutrients in the epilimnion. Estimates for recycling rates indicated that 93% of phosphorus metabolism and 81% of nitrogen metabolism for phytoplankton was accounted for by the recycling of these nutrients within the ecosystem.

In a more detailed examination of specific nutrient contributions from various sources, Thomann, *et al.* (1974) observed that during spring biomass, the hypolimnion of Lake Ontario contributed almost twice as many nutrients to the photic zone as were contributed from external loading sources to the Lake. In contrast, by early summer, biological recycling of nutrients in the epilimnion reached five times the rate of nutrient supply to the Lake from either external loading or from hypolimnion sources. These results indicated quantitatively the significance of recycling and vertical mixing relative to the level of external nutrient loading.

It should be emphasized, however, that there are no simple distinctions between nutrient loading dynamics to a lake, such as Lake Ontario, and the cascading trophic interactions, coupled with nutrient recycling, that occur and also control productivity. Potential production at all trophic levels is set by

nutrient supply. Actual production, however, depends on the recycling of nutrients and their partitioning among various trophic levels and populations that vary in growth rates (Carpenter, et al. 1985). The phosphorus available to phytoplankton, for example, is set by processes that operate over a wide range of temporal and spatial scales (Kitchell, et al. 1979). Nutrient regeneration through excretion by zooplankton is a major recycling process that is thought to be strongly influenced by selective predation on zooplankton by fish. Thus, by regulating recycling rates, predators regulate phytoplankton production. Further complexity arises because effects of zooplankton on phytoplankton biomass and productivity are not intuitively clear. These effects are generated by two contrasting processes: grazing and nutrient recycling. Herbivorous zooplankton alter phytoplankton species composition and size structure directly by selective grazing and indirectly through nutrient regeneration (Carpenter and Kitchell 1984; Lehman and Sandgren 1985).

In examining the long-term trends for Lake Ontario system-wide variables that we possess extensive data on, some interesting patterns emerge (Figure 35). Phosphorus loading to the Lake Ontario ecosystem has decreased substantially since the early 1970s. Total phosphorus in the mid-lake epilimnion has followed a similar trend, while as discussed above, inorganic nitrogen has increased significantly. Biological components have also shown interesting patterns (Figure 35). For example, phytoplankton biomass has exhibited a decline over the early 1970s and chlorophyll a measures, although more variable, have also demonstrated a decline with time. Top predator biomass on the other hand, has shown a major increase in the Lake over the last decade, as reflected by increased stocking rates. Ironically, decreases in phosphorus and phytoplankton measures, although corresponding to decreased loading to the system, also appear to mirror the increases in fish biomass through stocking (Figure 35).

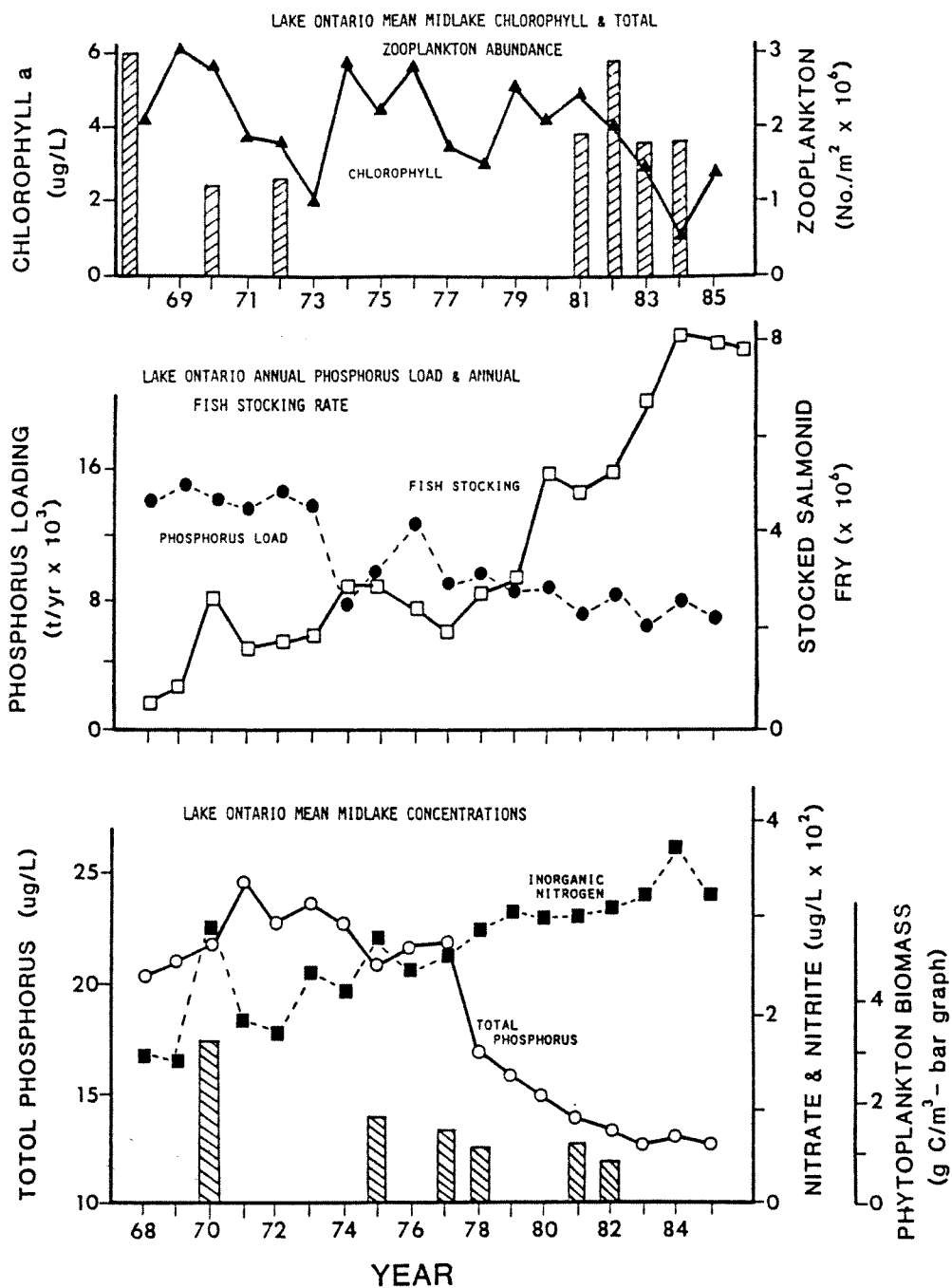
Whereas it appears that changes in external nutrient loadings for Lake Ontario may affect structure and productivity of phytoplankton communities, it is also clear from more recent Great Lakes data that the structure of higher trophic levels in the lake also potentially exert influences over the structure and productivity of phytoplankton (Kitchell, et al. 1988). Therefore, it is extremely important that we take the holistic, ecosystem approach that all factors influencing biological productivity are integrated into a picture of comprehensive system function. From this picture, it will then be possible to identify factors contributing to control of system dynamics and to attempt to gain perspective on the predominant influence on functioning.

Contaminants and Ecosystem Health

During recent years researchers have concentrated on gaining insight on the sources and mechanisms involved in contaminant transport and distribution in Lake Ontario. Results of these studies have contributed immensely to our understanding of processes affecting contaminant dynamics. It is clear, for example, that contaminants will persist in the Lake environment for decades, even if all point sources were eliminated now. For example, Mirex and PCBs are no longer being manufactured or used, but because of buildup in environmental reservoirs (e.g. sediment pools), their persistence will continue as long as they are recycled through the environment. In addition, new high-priority contaminants will be identified in the future and as analytical instrumentation improves, more contaminants may be linked with threats to human health.

The invertebrate forage base serves as the source for subsequent bioaccumulation and biomagnification of toxic contaminants in the Lake Ontario

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Zooplankton from Johannsson (1987), chlorophyll a measures from Dobson (1985), phosphorus loading data and mid-lake total phosphorus and nitrogen data from DePinto, *et al.* (1986), and phytoplankton biomass data from Munawar and Munawar (1986).

Figure 35. Long-term trends for various Lake Ontario ecosystem components.

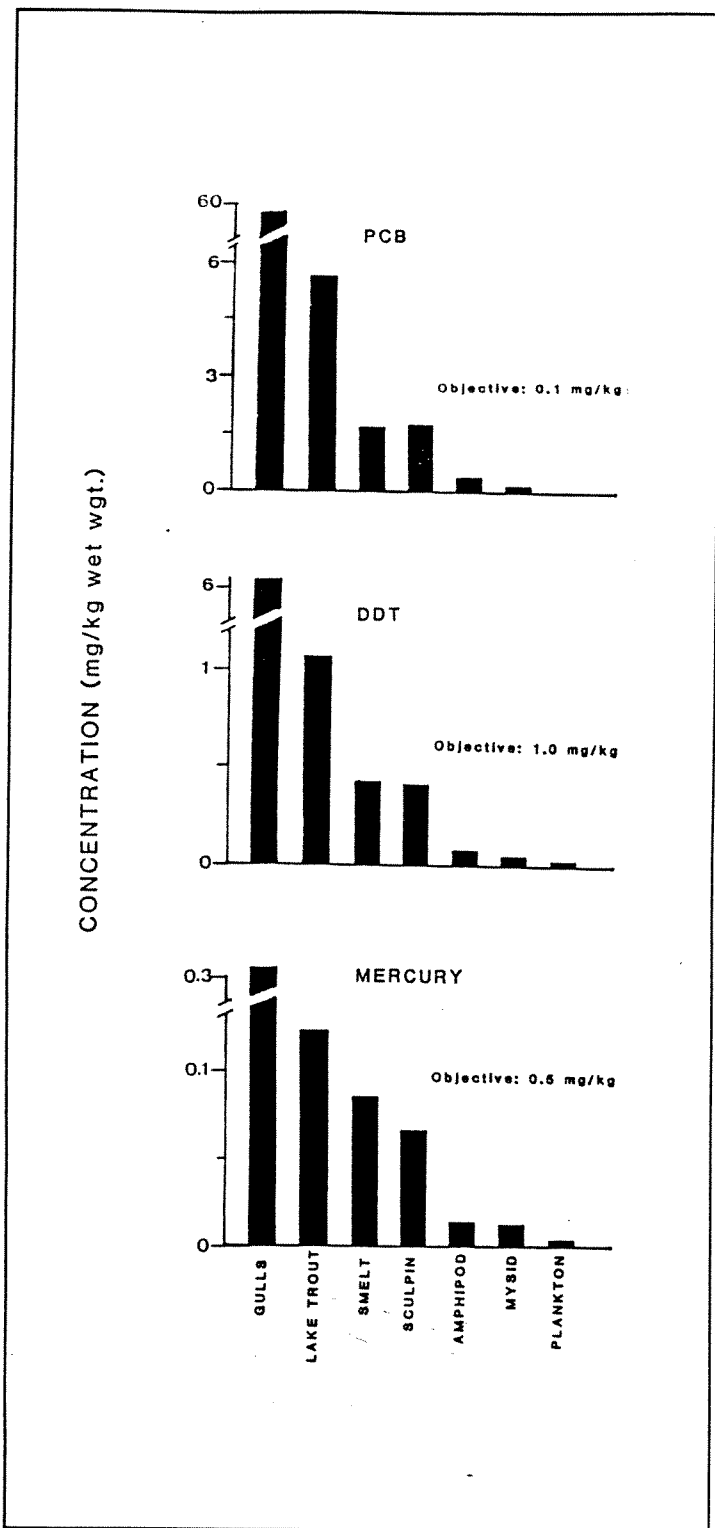


Figure 36. Illustration of the biomagnification of PCB, total DDT and mercury through the Lake Ontario food web. From the International Joint Commission (1987a).

ecosystem. Net plankton, zooplankton (*Mysi relicta*), and benthic invertebrate (*Pontoporeia hoyi*) form the first three step in food chain contaminant biomagnification and serve as biological surrogates for the measurement of persistent toxic chemicals in the water column. Mean bioconcentration factors for organochlorine compounds such as PCB or DDT are 10^5 within the aquatic food chain (Figure 36) up through herring gulls which represent the highest trophic level. Similarly, trace metals are also rapidly bioconcentrated within the aquatic food chain with factors exceeding 10^3 for mercury (Figure 36).

Accumulation of contaminants in the food web, combined with the harvest of many fish species that serve as top predators in this food web, represents a potentially significant removal of contaminants from the ecosystem, thus breaking the recycling of these materials and decreasing their persistence. Although this process of fish harvesting may provide a mechanism for toxic contaminant removal from Lake Ontario, the important point to emphasize here is that all warnings and bans against consumption of these fish be followed carefully so that this potential biological control on contaminant will not turn into a major human health problem.

Similar to the role of fish harvesting as a possible biological control on contaminant flow in the system, the macroalga *Cladophora* may also represent a means of controlling contaminant flow and breaking the recycling pattern of these materials in the Lake ecosystem. Extensive growths of the attached green alga, *Cladophora glomerata*, are found in the nearshore zones of Lake Ontario wherever sufficient light and firm substrate occur. Estimates of standing stocks have ranged from a mean of 224 g/m^2 (dry weight) at a 1.5 m water depth to 100 g/m^2 at 5.0 m water depth (Shear and Konasewich 1975). Based upon these estimates and the known habitat area for *Cladophora* in Lake Ontario (187 km^2) it was calculated that production of this macroalga exceeded 19,510 metric tons/yr (dry weight).

This flora received considerable attention when it was recognized as a nuisance associated with nitrogen and phosphorus enrichment (Neil and Owen 1964; Bellis and McLarty 1967; Herbst 1969; Gerloff and Fitzgerald 1976). *Cladophora* is a well studied alga (e.g. Shear and Konasewich 1975; Wolfe and Sweeney 1982; Lowe, et al. 1982; Millner, et al. 1982), and there is evidence that it bioaccumulates trace metals, DDT, PCB and other materials to several orders of magnitude over ambient water concentrations (Yaguchi, et al. 1974; Kinney, et al. 1976; Anderson, et al. 1982). Within the Niagara River and at its mouth in Lake Ontario, *Cladophora* biomass has been shown to contain much higher levels of heavy metals and PCBs than in similarly sampled areas of Lake Erie (The Niagara River Toxics Committee 1984). Kinney, et al. (1976) demonstrated that dissolved trace levels of heavy metals including zinc, cadmium, lead, and copper were accumulated from 1.0×10^3 to 49×10^3 fold by *Cladophora*. Similarly, mercury was found to be accumulated by *Cladophora* from western Lake Erie (Burkett 1973) and PCB residues were found to accumulate in excess of ambient water concentrations in a *Cladophora* community in Lake Huron (Anderson, et al. 1982).

Documentation that *Cladophora* can accumulate aquatic contaminants and that this flora is viewed as a nuisance whenever it reaches high levels of population growth, suggests it may serve as a suitable candidate to consider in biological control strategies. The bioaccumulation mechanisms of *Cladophora*, coupled with the harvest and removal of this biomass from the aquatic environment, could serve as an ideal control alternative. For example, if one were to assume that *Cladophora* could accumulate 10 ppm of a toxic contaminant in its plant tissue, then based upon the estimated annual standing stock of this alga in Lake Ontario (39.0×10^6 kg), approximately 39×10^4 gm of contaminant would be tied up in *Cladophora* biomass annually. If one could remove 10% of this natural biomass from the Lake, 39 kg of

toxic material could be removed annually, rather than continuing to recycle in the habitat.

An additional consideration for the use of natural populations of this macroalga is its potential to serve as a detector of hazardous chemicals in surface waters used for drinking before they reach crisis level, by the routine monitoring of *Cladophora* body burdens for potentially dangerous chemical concentrations. In this way, water quality evaluation would become more holistic, producing a temporal "snapshot" which could serve as an accurate gauge of the health of these surface waters. This same approach is applied in coastal marine environments with the infamous "Mussel Watch" program (EPA and now NOAA).

There seems to be sufficient evidence to warrant the use of *Cladophora glomerata* as a biological control for managing toxic contamination in Great Lakes' environments. Through the development of this strategy, the stimulus may be provided to begin harvesting this macroalga, thus lessening the nuisance caused by its heavy growth, and provide a reasonable alternative for environmental managers to use in addressing toxic contaminant problems in surface waters.

Ecosystem Objectives

Considering the number and variety of cultural stresses that have impacted Lake Ontario, with many of the results of these insults not detected until years after the initial impact to the system, ecosystem objectives that indicate system quality are now being considered as a necessary management strategy (Ryder and Edwards 1985). A prerequisite for appropriate application of ecosystem objectives is the need to determine the current state of health of the environment and its contained resources, and to identify areas where the ecosystem may be unduly stressed by cultural interventions.

With the implicit understanding that the various biotic compartments of an ecosystem may reflect the effects of cultural stresses at several hierarchical levels, including those of species, stocks, populations, and communities, it only makes sense to investigate these various components to identify appropriate indicators of ecosystem quality. At the community level of organization, any observable departure from base-line conditions of steady state or natural trends and cycles may provide ready foci for identifying stress effects. Within stocks and populations, density dependent changes in response to cultural stresses, at least in some instances, would likely provide a suitable early measure if appropriately calibrated. Loss of particular stocks identified as either genotypically or phenotypically distinct, through mechanisms of extinction or introgression, may be a symptom of more advanced environmental stress.

In addition to the species, stock, population, and community ecological measures for ecosystem quality, physiological measures and surrogates implying physiological factors can also be considered. Concentrations of xenobiotic chemicals in species like lake trout have been used as a surrogate measurement for general environmental pollution (Ryder and Edwards 1985), which presumably translates into ecosystem health and serves as preliminary clues to the cause for declines in stock abundance. For the chlorinated hydrocarbons or other lipophilic chemicals, understanding of the bioconcentration mechanism is essential if accurate projections of ecosystem health are to be expected.

Other surrogate measures of contaminant mixture effects on fish stocks include lowered resistance to disease or additional pollution, inability to complete critical life functions, such as spawning, hatching, rearing, or the attainment of sexual maturity, and increased incidence of deformities which may translate into indirect and undetermined mortality through

selective predation. In addition physiological indicators of single species stress may be instructive in depicting the state of the environment, especially when that species, as a keystone predator, is the ultimate integrator of community dynamics. Two biochemical measures of stress that have recently received serious consideration by scientists are the mixed function oxidase system (MFO) and the adenylate charge procedure which compares levels of ATP/ADP/AMP as indicators of stress (e.g., Hochachka and Somero 1977; Payne 1977).

In the last several decades, a number of candidate species, both aquatic and terrestrial, have been proposed as indicator species for environmental quality which include: vegetation for the detection of pollutants (Jones and He 1981); *Cladophora* as an indicator of levels of biologically available phosphorus (Aue et al. 1982); insects, amphipods, and isopods as indicators of stream quality (Hilsenhoff 1982); herring gull as representative of the highest trophic level of both aquatic and terrestrial food webs (Gilman, et al. 1979); zooplankton as indicators of PCB levels (Clayton, et al. 1977); mussels as indicators of concentration changes in trace metal DDEs, PCBs, hydrocarbons and artificial radionuclides (Farrington, et al. 1983); fecal bacteria in shellfish (mussel cockles, clams, etc.) as indicators of levels of fecal bacteria in the surrounding water (Al-Jebouri and Trollope 1984); and fish communities for multiple ecosystem stresses imposed by man (Regier 1979; Ke and Dickie 1984).

Indicator species to consider for Lake Ontario should comprise one of two types of organisms: specialized species with only narrow tolerances for environmental properties (stenoeccious), and those species that are less specialized with broader tolerances (euryecious). Both types of organisms are useful indicator species but in different ways (Ryder and Edwards 1985). Ideally, the most satisfactory results would be obtained through

monitoring both types of organisms: the information obtained from stenocious organisms would complement that obtained from euryecious organisms. In a practical sense, however, certain organisms stand out as prime potential indicators such as the lake trout which is representative of terminal predators (fourth trophic level of the food web) for oligotrophic environments. Similarly, the walleye is a terminal predator indicative of mesotrophy and early eutrophy, while the crustacean, *Pontoporeia hoyi*, is an invertebrate representative of the secondary trophic level of the food web for deep water benthic and pelagic zones of oligotrophic environments. Additionally, Forster's tern has been shown to be a good indicator for wetland marshes bordering the Great Lakes and in particular, was indicative of the level of water turbidity realized from cultural eutrophication and other sources.

Specific criteria are important to consider in order for an organism or a population to serve as a good indicator of environmental conditions in a Great Lake environment. These criteria include the following and, although it might not be possible to select species that contain all these characteristics, the organisms chosen should possess as many as possible.

1. Consideration should be given to the most sensitive life stages of the most sensitive species in the ecosystem.
2. Reproductive and early life stages should be vulnerable to environmental stress.
3. The species should reflect an environment developed over evolutionary time in contrast to an exotic species.
4. The species should be an integrator of material flows through an ecosystem's food web.

5. The species should be able to provide a composite picture of the ecosystem by its migration patterns throughout the various parts of the ecosystem.
6. Need indicator organisms that are site specific and can reflect temporal changes in a given environment, such as the benthos.
7. An ecosystem indicator that can provide a long record of ecosystem quality, possibly through paleolimnological records, would be valuable for trend analysis.
8. Any organism chosen as an ecosystem indicator must be bi-directional, capable of showing both improving and worsening conditions of system health.
9. Ecosystem indicator species chosen should exhibit interpretable results, results which can be understood.

An ecosystem objective, by definition, infers a broad scientific approach encompassing all aspects of the environment and the indigenous biota. Such an objective for any of the Great Lakes may be impossible to accomplish, although it is philosophically satisfying. Alternatively, establishment of ecosystem objectives for a Great Lake such as Lake Ontario using a single species or subset of species may be justified, provided that ecological characteristics of the species can be adequately described. The development of an appropriate ecosystem objective would allow for the monitoring of Lake Ontario environmental health in a holistic fashion by employing an integrator of a number of different system functions that would be of concern in assessing the Lake's quality over time.

CHAPTER 7

ECOSYSTEM APPROACH TO MANAGEMENT STRATEGIES

The Great Lakes have experienced a diversity of perturbations to their natural function which include excessive nutrients, influx of toxic contaminants and the restructuring of fish communities through both exotic invasions and purposeful stocking. Although both federal and state/provincial agencies in Canada and the U.S. have made significant progress toward the reversal of eutrophication in the Great Lakes and are presently taking actions to halt the input and cycling of toxic materials in these lakes, not enough emphasis in management strategies has been placed on the interconnections of many of these phenomena. This recognition is imperative if an "ecosystem approach" is truly our guide to Great Lakes resource protection. In this regard, the regulatory agencies have accepted the fact that it is unrealistic to assume that man can effectively manage natural systems as complex as any of the Great Lakes. It is now believed that what we can do is influence human uses and abuses of the natural resources of these Great Lakes ecosystems. In order to accomplish this strategy an integrated understanding of system function is imperative.

An ecosystem context for inquiries about Lake Ontario provides a plausible framework for the stewardship of water quality and fisheries resources. Information about causal relationships, however, is generally lacking and management decisions have most often been guided by extrapolated correlations focused upon individual problems. For example, natural resource managers are attempting to apply ecological theory to their problems of system management. Often the application of ecological theory to one resource problem is performed in isolation

from other problems (i.e., external nutrient loading vs. food web manipulation). Thus, confounding effects between the problems may ultimately cause failure of strategies to address the single problem. It is suggested that natural resource managers must develop a better appreciation for the interconnectedness of their problems associated with managing ecological processes, which are often viewed otherwise in isolation from one another. Below, several examples are presented for Lake Ontario that demonstrate the need for a more holistic perspective: addressing problems of concern for the Great Lake.

Nutrient Control vs. Fish Stocking

Is it possible that present eutrophication control in many of the Great Lakes is the result of fish stocking strategies and food web manipulation, as much as from effects of continued nutrient abatement programs? Is it also possible that continued abatement of nutrients by "shutting off the tap" while stocking millions of recreational fish, could result in no food for these fish and destruction of a multimillion dollar economic infrastructure around the Great Lakes? Likewise, abatement of nutrients beyond some presently unknown threshold at the costs of millions of additional dollars may be unnecessary when manipulation of fish communities are, by themselves, having a positive effect on improving water quality.

Nutrient limitation has been linked to deficiencies in phosphorus (Thomann, *et al.* 1981) as one factor most important in controlling phytoplankton productivity.

This pattern may be changing now, however, with nitrogen becoming more important (Figure 35). As of 1982, phosphorus levels in Lake Ontario were judged to be approaching the target established by the 1972 Great Lakes Water Quality Agreement between Canada and the United States (Dobson 1985). Decreased phosphorus and reversal in eutrophication have been the result of an intensive abatement program. By this strategy, system productivity has been significantly decreased as measured by improvements in water quality, including phosphorus decreases, decreases in chlorophyll a (Figure 35), and phytoplankton community composition changes.

In the mid-1970s extensive stocking programs were initiated to introduce and in some cases restore large salmonine species to Lake Ontario. Since the late 1970s millions of salmon and trout fry have been jointly stocked in Lake Ontario annually by Provincial, State and Federal governments in Canada and the United States (Figure 35). The purpose of this stocking strategy has been to increase system productivity as reflected in the recreational harvest of large predatory fish (Jude and Tesar 1985).

Presently, nutrient abatement programs are forcing lower levels of biological productivity that are meant to support predators, and these lower levels of productivity may not be sufficient to maintain significantly increased predator biomass (Eck and Brown 1985). It is worth considering the merit of this issue using the data available on various components of the Lake Ontario food web/nutrient cycling system and comparing changes in these processes over time. Eadie and Robertson (1986), Scavia (1979), Borgmann et al. (1984) and Flint (1986) have addressed portions of this problem for Lake Ontario through either the development of carbon budgets, the qualitative description of nutrient flux, or calculations to estimate salmonine biomass from zooplankton production measures.

Estimated phosphorus loading rates for Lake Ontario can be obtained from the data presented in Figure 35 for three years where we have corresponding information of phytoplankton and fishery production rates (Table 10). According to the conceptual scheme for flow of carbon through the Lake Ontario food web hypothesized by Flint (1986) and illustrated previously in Figure 18, phytoplankton production was estimated to equal $178 \text{ g C/m}^2/\text{yr}$ in 1972. Lean, et al. (1987) conducted phytoplankton production measures in Lake Ontario in 1982 for a six month period. Extrapolating their results to a yearly total, with aid of winter production rates from Munawar and Munawar (1986), gave a primary production estimate of between $78\text{-}90 \text{ g C/m}^2/\text{yr}$. For purposes of comparison between years it was decided to use the upper value to represent 1982 production estimates and the lower value to characterize 1987 lake-wide production estimates (Table 10), assuming that phosphorus loading and mid-lake phosphorus concentrations have continued to decline (Figure 35).

Fisheries production in Lake Ontario for 1972, 1982 and 1987 (Table 10) can be obtained from several sources. Based upon historic fishery yields (Christie 1973), production was estimated to be equivalent to $0.04 \text{ g C/m}^2/\text{yr}$ for 1972 (Flint 1986 and Figure 18 above), prior to any major impact on the Lake from salmonid stocking (Figure 35). From early 1980 salmonid survival estimates (Echert 1984), Flint (1986) calculated that salmonid production in Lake Ontario was equivalent to $0.02 \text{ g C/m}^2/\text{yr}$. Combined with historic fish community production of $0.04 \text{ g C/m}^2/\text{yr}$, the total 1982 fish production was calculated to equal $0.06 \text{ g C/m}^2/\text{yr}$ (Table 10). Estimates for total fish yield in Lake Ontario in 1987 (C. Schneider, personal communication, NY Department of Environmental Conservation, Cape Vincent, NY, March, 1988; W. Dentry, personal communication, Ontario Ministry of Natural Resources, Picton, ON, March, 1988) indicated that $4.09 \times 10^6 \text{ kg}$ of fish were harvested during

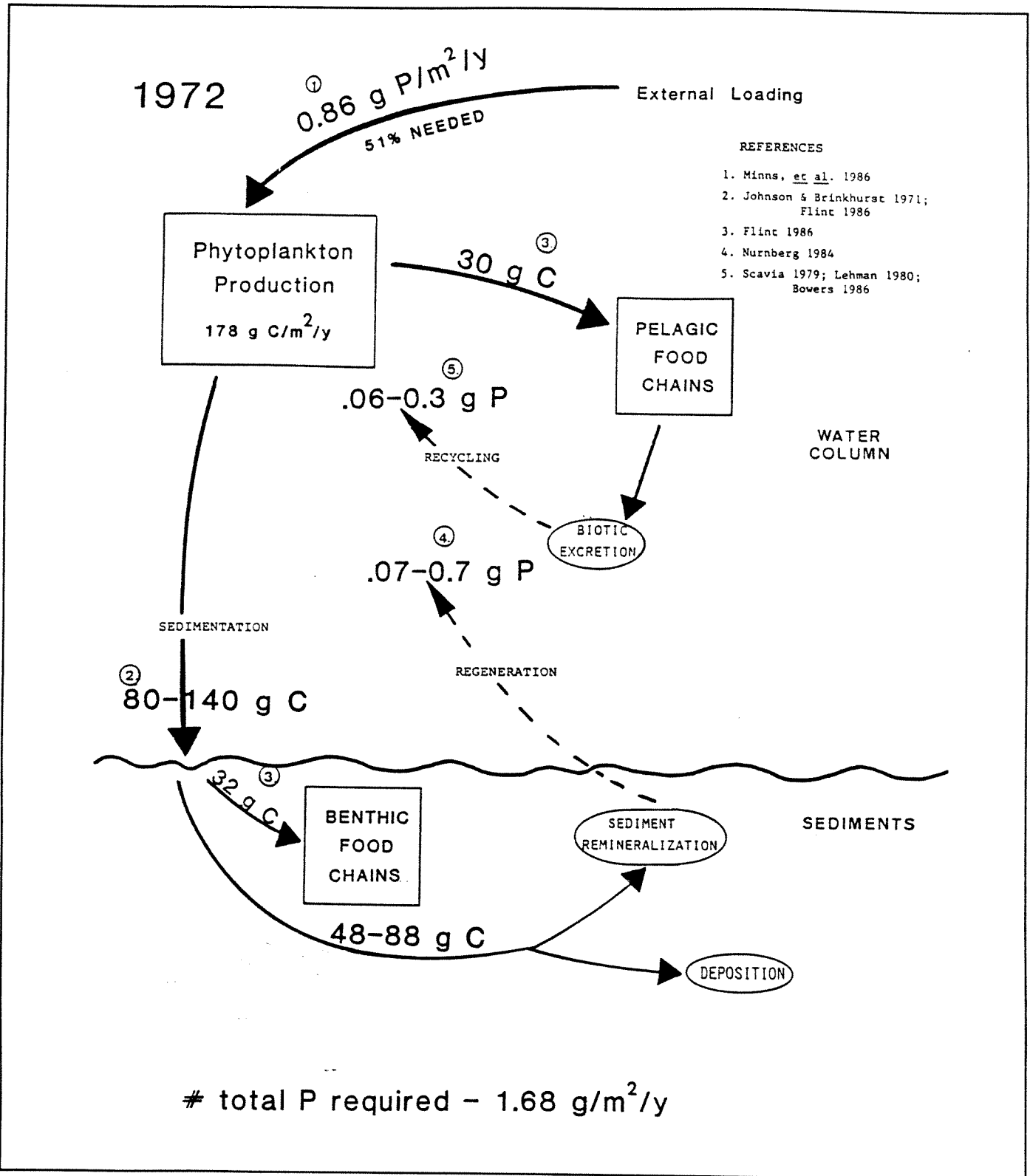


Figure 37. Conceptual scheme for the nutrient recycle pattern in Lake Ontario that illustrates estimated compartment values for 1972.

Although the above simulation suggests that decreased loading of nutrients to Lake Ontario coupled with increased biomass of predatory fish may be burdening the ecosystem, it should be noted that changes in nutrient loading dynamics also affect other system components, such as the partitioning of nutrients from the various sources that fuel primary production. Of most concern here is, that with recent abatements in nutrient loading, as suggested by Figure 35, is it possible that there are now not enough nutrients cycling in the system to make up for decreases in external sources needed to support phytoplankton production and in turn, the large recreational fishery being developed?

Growth and reproduction of phytoplankton in an aquatic system are the basis of carbon production ultimately realized up through the food web to fisheries. The appropriate interaction of essential nutrients is vital to the production of this carbon. As time has progressed, recognition of links between nutrients and the entire trophic web of aquatic systems, through phytoplankton dynamics, has caused studies to encompass more than phytoplankton, taking a broader perspective and including zooplankton, nekton, benthos, and bacteria (Flint 1985a). Simultaneously, mass balance studies of nutrients in aquatic systems have implicated the importance of excretion and remineralization of organic matter by biological processes in supplying nutrients to aquatic systems in support of biological productivity (Robertson and Scavia 1984; Flint 1985b; Flint, *et al.* 1986).

Whereas changes in external nutrient loadings for Lake Ontario may affect composition and productivity of phytoplankton communities, examination of nutrient cycling from a biological rather than physio-chemical perspective conclusively, suggests that there are also controlling factors within lakes that are generated by higher trophic levels (Taylor 1984; Carpenter, *et al.* 1985; Scavia, *et al.* 1986). In particular, an increase in

phytoplankton, with shifts towards larger sizes is coincident with a decrease in large herbivorous zooplankton (Lynch and Shapiro 1981; Elliott, *et al.* 1983). Likewise, a change in the biomass of large zooplankton is the result of changes in the fish community (Henrickson, *et al.* 1980; Stenson 1982) most often affected by man's manipulation of the ecosystem (Eck and Brown 1985). These changes are thought to significantly influence nutrient recycling by zooplankton, which may be an important nutrient source to phytoplankton.

Also important are influences of sediment nutrient regeneration. Although not always available because of thermal stratification, these nutrients are able to recycle into the epilimnion during the winter and early spring, preceding the onset of maximum phytoplankton production. They could play a major role in annual system production. It should be emphasized that there are no simple distinctions between nutrient loading dynamics to a lake, such as Lake Ontario, and the cascading trophic interactions that occur and also control productivity. Potential production at all trophic levels is set by nutrient supply. Actual production, however, depends on the recycling of nutrients and their partitioning among various trophic levels that vary in growth rates (Carpenter, *et al.* 1985).

The few mass balance calculations that exist for Great Lake food webs and nutrient cycles fall short of demonstrating synergistic function. Probably the best attempt was presented by Scavia (1979). He constructed an ecological model derived from experimental results on Lake Ontario during the 1970s. The outcome of this effort was important for delineation of some of the factors that control phytoplankton production and phosphorus cycling. Unfortunately, these modeling efforts were not able to incorporate some of the more recent ecological theories concerning the magnitude of control that successive levels of the food web exert on primary production and nutrient cycling in the Great Lakes. Significant questions now

State of a Great Lake

Table 10. Results of simple simulation model comparing nutrient loading to Lake Ontario phytoplankton production in the deep water environment and fisheries production as reflected by fish harvest.

YEAR	ESTIMATED ^{a.} PHOSPHORUS LOADING (g P/m ² /yr)	TOTAL P ^{b.} REQUIRED TO SUPPORT PPR (g P/m ² /yr)	MEAN ANNUAL ^{c.} PHYTOPLANKTON PRODUCTION (g C/m ² /yr)	TOTAL ^{d.} FISHERIES PRODUCTION (g C/m ² /yr)	PPR REQUIRED ^{e.} TO SUPPORT OBSERVED FISH PRODUCTION
1972	0.86	1.68	178.0	0.04	40
1982	0.43	0.94	90.0	0.06	60
1987	0.38	0.74	78.0	0.11	110

a. From International Joint Commission 1987a.

b. Based upon the Redfield ratio (Redfield, et al. 1963).

c. From Munawar and Munawar (1986), Flint (1986), and Lean, et al. 1987).

d. From Lake Ontario Committee annual reports, Christie (1973), Borgmann, et al. (1984), Flint 1986) and Loftus, et al. (1987).

this year. Based upon conversion assumptions described in Flint (1986), this harvest was equivalent to 0.11 g C/m²/yr of fish production (Table 10).

Comparison of the nutrient-food web components in Table 10 for the three different years of calculation provides initial evaluation of the question of adequate food web support for fisheries production in Lake Ontario. In 1972, 51 percent of the phosphorus required to support phytoplankton production was derived from external loadings and there was more than enough phytoplankton carbon produced to support estimated fishery production. In 1982, 46 percent of required phosphorus was derived from external loading, and again there appeared to be enough phytoplankton carbon production to support estimated fishery production in the Lake. Over the 15 year

period (1972-1987), the 1987 calculation give the first indication that there may be more fish production in Lake Ontario than long-term nutrient loading and phytoplankton production can support (Table 10). Loading appeared to provide 5 percent of required phosphorus for phytoplankton while the fish production estimated for 1987 required 32 g C/m²/yr more than what was apparently produced by the system. At present stocking rates of salmonids for Lake Ontario (Figure 35) or can assume that fisheries yields will remain at 1987 levels for the next several years. If this is the case, then based upon the calculations presented in Table 10, Lake Ontario's present productive dynamics are stressed by lack of sufficient nutrients available to sustain the primary production required to support observed yields of fish from the system.

Lake Ontario

Table 11. Representation of three different nutrient cycling scenarios for the deepwater environment of Lake Ontario based upon known changes in nutrient loading to the system for three different years and data on other processes in the system that affect rates of nutrient recycling. All Carbon to Phosphorous conversions done in the construct of this simulation are based upon the stoichiometric relations of Redfield (Redfield, et al. 1963).

YEAR	Phytoplankton ^a Production (PPR) (g C/m ² /yr)	Required P to support PPR (g P/m ² /yr)	External Loading (g P/m ² /yr)	Required PPR ^b Support for Pelagic Food Web (g C/m ² /yr)	Pelagic P ^c Recycling (g P/m ² /yr)	PPR ^d Sedimented to Benthos (g C/m ² /yr)	Benthic p ^e Recycling (g P/m ² /yr)
1972	178	1.68	0.86	30	0.06-0.30	80-140	0.07-0.7
1982	90	0.94	0.43	70	0.20-0.70	10-30	0
1987	78	0.74	0.38	35	0.10-0.35	22-39	0-0.06

a. Based upon Table 10 calculations

b. From Flint (1986) and Johannsson (1987).

c. From Scavia (1979), Lehman (1980), and Bowers (1986).

d. From Johnson and Brinkhurst (1971c) and Flint (1986).

e. Based upon benthic food chain requirements of 32 gC/m²/yr (Flint 1986). and measures of benthic nutrient regeneration by Nurnberg (1984).

arise concerning the proportional sources of nutrients required to support a Great Lake food web.

Table 11 presents the results of three different nutrient cycling scenarios in Lake Ontario, based upon the best available data for those years listed. The pattern of nutrient cycling and the estimated compartment values in Lake Ontario for 1972 are illustrated in Figure 37 to demonstrate the approach used here to calculate nutrient recycling schemes through time. Phytoplankton production and the amount of total phosphorus required to support this production were derived as discussed above for Table 10. The amount of phytoplankton carbon required to support

the pelagic food chain was derived from original estimates in Flint (1986) with extrapolations to other years based upon zooplankton abundance data of Johannsson (1987). According to Johannsson total zooplankton abundance reached a peak of approximately $3.0 \times 10^6/m^2$ in 1982 in Lake Ontario. In 1972 total abundances were one third of this value and in 1984 abundances ranged around $1.5 \times 10^6/m^2$. Since we do not possess extensive data on biomass or production of zooplankton for the total period 1972-1987, we extrapolated from the abundance measures of Johannsson (1987) using a few biomass and production measures cited for the early 1980s (Flint 1986). Based upon these extrapolations, 30 g C/m²/yr of primary production was required

to support the pelagic food chain in 1972, 70 g C/m²/yr in 1982 and 35 g C/m²/yr in 1987 (Table 11).

Pelagic phosphorus recycling rates (Table 11) primarily from the zooplankton community, were estimated from published rates of Scavia (1979), Lehman (1980) and Bowers (1986). Benthic regeneration rates of phosphorus (Table 11) were based upon the measures of Nurnberg (1984) and the estimated sedimentation rate of carbon produced by phytoplankton (Johnson and Brinkhurst (1971c; Flint 1986), as well as the amount of carbon required to support benthic community nutrition (Flint 1986).

From comparison of potential nutrient dynamics in support of system phytoplankton production (Table 11), it is apparent that there is considerable variation between years, and changes in external phosphorus loading rates correlate with significant changes in the various recycling components of the Lake Ontario system. For example, in 1972 external loading plus the maximum contribution from pelagic and benthic phosphorus recycling rates provided more than enough phosphorus to support projected phytoplankton production (Table 11; Figure 37). Even though no benthic regeneration of phosphorus appeared possible in 1982, based upon mass balance calculations, there still seemed to be sufficient phosphorus coming from external loading plus pelagic recycling to support observed phytoplankton production (Table 11). In 1987, however, the contributions from all sources of phosphorus appeared to be very close to, and possibly short of, the amount required to support estimated phytoplankton production.

This description of changes in nutrient dynamics that have supported phytoplankton production in Lake Ontario over the period 1972-1987, does not take into account effects from food web manipulations that have also occurred in the ecosystem and does not take into consideration the fact that phosphorus to nitrogen ratios are also changing. Both of these factors could play a significant role

in the system-wide changes that have been postulated (Tables 10 and 11) in Lake Ontario over the last 15 years. The scenarios described above, however, do demonstrate that consideration has to be given to multiple dynamics within an ecosystem like Lake Ontario when management decisions are made. Without integrative synthesis of our knowledge in these areas there is serious danger that various strategies could be working in opposition to one another regarding desired results

Two points are important to consider. First, excess abatement of nutrients by "shutting off the tap" while also stocking millions of fish species, could result in no food for fish and destruction of a multi-million dollar economic infrastructure around the Great Lakes. Likewise, nutrient abatement beyond some presently unknown threshold at costs of millions of dollars, may be absolutely unnecessary when manipulation of biological communities in the Great Lakes are by themselves having a positive effect on improving water quality through changes in nutrient cycling dynamics.

Nutrient/contaminant Cycle Linkages

Are there unknown linkages between nutrient cycles and contaminant cycles in our lakes that translate into a more productive, nutrient-rich ecosystem resulting in less toxic transfer to humans? For example, Lake Erie is more productive than Lake Ontario (Beeton 1969; Glooschenko, *et al.* 1974; Robertson and Scavia 1984), and yet there appears to be less body burden of comparable toxic contaminants in Lake Erie game fish than in Lake Ontario game fish.

Whittle and Fitzsimons (1983) used a number of species representative of several trophic levels to demonstrate Niagara River influences on Lake Ontario. We can use this same data to examine differences in body burdens of these species between Lakes Erie and Ontario. Whittle and Fitzsimons (1983) found that concentrations of PCBs,

DDT and p, p'-DDE were significantly higher in coho salmon, rainbow smelt (*Osmerus mordax*) and the benthic amphipod (*Pontoporeia affinis*) from the western basin of Lake Ontario than the eastern basin of Lake Erie (Figure 38). Similarly, in a comparison of several organic contaminants in herring gull eggs at an eastern Lake Erie colony (Port Colborne) and a western Lake Ontario colony (Mugg's Island, Toronto Harbor), organic levels were markedly higher in Lake Ontario with increases ranging from 17 percent for dieldrin to 495 percent for mirex (Stevens 1987). Samples of net plankton (i.e., phytoplankton < 153 μ m), in contrast, exhibited no significant difference in any organic contaminant except PCBs. The mean PCB level in surface plankton from Lake Ontario was 75 percent greater than the Lake Erie mean value (Whittle and Fitzsimons 1983).

Changing the food web structure through manipulation is also hypothesized as a means of altering the flow of toxic chemicals within the food web that ultimately leads to humans. By changing the flow of materials (e.g., nutrients and carbon) in Lake Ontario, it is suggested that flow of toxic contaminants from various sources (pools) within the ecosystem will significantly be altered. For example, modifications away from food web dominance by alewife populations, low in feeding efficiency and high in lipids, could reduce biomagnification of toxic substances in top predators sought by fisherman. These changes might be accomplished through the stocking of alternative forage fish that could replace the dominant alewife populations in the Lake Ontario system.

Another indirect impact on the recycling of toxic contaminants within the Lake comes from the actual harvest of the large predators in the system (e.g., salmon and trout). It is widely recognized that these fish concentrate many of the toxic chemicals from the various biotic and abiotic sources in the system. The harvest of these fish through the extensive recreational fishery, serves as a means of

removing much of the toxic contamination from the system and breaking the continuous recycle of these materials, which increases their residence time in the Lake. This strategy represents a mechanism for toxic removal as long as the fish that are harvested are not used for human subsistence.

Specific Toxic Transfer Routes

Could we have predicted that non-migrating bass in a pristine river system, such as the Salmon River on Lake Ontario's south shore, would eventually show large enough toxic body burdens to require consumption restrictions? Terrestrial mammal monitoring and familiarity with the life history of these fish suggests that we should have been able to predict this kind of phenomenon with the proper insight and holistic view of these Great Lakes systems. Consider the following. In addition to the lake-wide contamination of sediment and biota with mirex, evidence also exists for upstream dispersal of mirex into previously uncontaminated tributaries (Thomas, *et al.* 1987; Lewis and Makarewicz 1988). Anadromous migration, spawning and subsequent carcass decomposition of salmonids may result in the incorporation of mirex into stream food chains and ultimately into terrestrial biota as well.

Upstream dispersal of mirex was examined in tributaries of the Salmon River in eastern New York State (Low, 1983). Coho and chinook salmon migrate into the accessible tributary streams, spawn and die. Resident brown trout, populations that never enter Lake Ontario, feed almost exclusively on salmon eggs during the fall spawning period (Johnson and Ringler, 1979a). Moreover, the carcasses of the salmon that wash ashore along the stream edges are colonized by blowfly larvae (Diptera: Calliphoridae), which are the major decomposer organisms on the streams, and are consumed by brown trout (Johnson and Ringler, 1979b). Other blowfly larvae that metamorphose into adults potentially can disperse their body burden of mirex into terrestrial food chains.

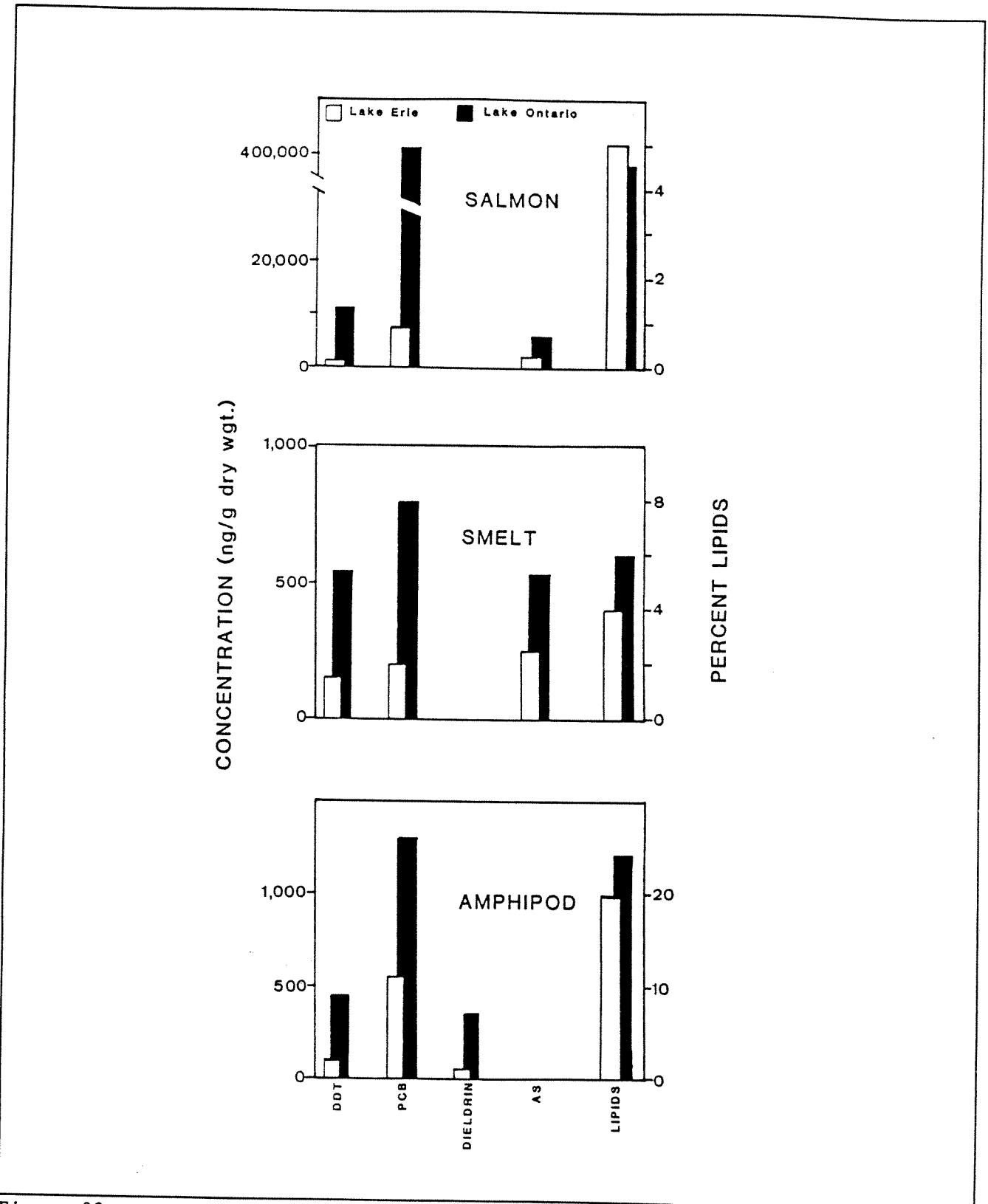


Figure 38. Comparison of toxic chemical body burdens for organisms in Lake Erie and Lake Ontario. Percent of body lipids for organisms compared between lakes is also shown. From Whittle and Fitzsimons (1983).

Elevated mirex concentrations (mean of 34 ng/L) were found in brown trout in streams with access to salmonid migration (Low, 1983). Relatively high levels of mirex were observed in salmon eggs (49 ng/L) and blowfly larvae (150 ng/L). The level of contamination of salmon eggs, and their availability, apparently provided the most important pathway of mirex loading into resident fishes (Thomas, *et al.* 1987). The salmon egg-resident fish pathway and the blowfly larvae-terrestrial pathway appear to be avenues of toxic contaminant fate and distribution worthy of investigation in other regions and with other lipophilic toxic contaminants. Indeed, Merna (1986) noted similar upstream dispersal of chlorinated hydrocarbons by migrating salmon in a tributary to Lake Michigan.

Scavenging of spent carcasses by vertebrates, such as raccoons, has also been observed (Fisher 1981), thereby providing another mechanism of mirex dispersal into the terrestrial environment. Furthermore, Foley, *et al.* (1988) have verified the probability of this mechanism occurring through the monitoring of mink and otter. These investigators have observed a strong correlation between mink body burdens for PCBs and body burdens of this same contaminant in Lake Ontario fish that potentially serve as a food source for the mink and otter.

Toxic Contaminant Persistence in Lake Ontario

Recognizing the long-standing abuse of the Lake Ontario ecosystem as a disposal ground for harmful discharges, including domestic sewage and industrial waste, the various governing jurisdictions now acknowledge the need to develop a clearer understanding of the relative importance of input to the system, and other factors that influence the fates of materials (*e.g.*, nutrients and toxics) in the Lake, in order to establish priorities in allocating resources to control the most significant sources of pollution to the system. Beyond

this objective the jurisdictions desire to develop zero discharge limits for all harmful substances entering Lake Ontario (*i.e.* Great Lakes Water Quality Agreement Amendment 1987). The question to consider here is if the elimination of discharge of toxic materials were implemented today, how long would the Lake Ontario ecosystem continue to exhibit the residue of the toxic materials that have already entered the ecosystem? In other words, how persistent are these toxic contaminants in the Lake?

To investigate this question a mass balance modelling approach is the best means to develop an accounting for materials in a system such as Lake Ontario over time. Mass balance modelling will provide an accounting of material entering and leaving the lake by various pathways, a systematic process for predicting the effects of proposed remedial actions and an indication of the persistence of the materials following implementation of control processes. The disadvantage of a mass balance approach for tracing the movement of materials through Lake Ontario is one of data availability. Knowledge is required on input sources and rates, residence times, partitioning factors, and ultimate fate paths of the materials in question. Unfortunately there are very few toxic contaminants for which all of these data are available.

A possible exception may be mirex in the Lake Ontario ecosystem. For this particular chemical there is a relatively good estimate of the amount that entered Lake Ontario over a specific time period (Holdrinet, *et al.* 1978). Measurements have been made of concentrations in various components of the Lake (*e.g.*, sediments and biota) that serve as pools for this contaminant in the ecosystem (*e.g.*, Whittle and Fitzsimons 1983; Eadie, *et al.* 1983). In addition, projected loss of mirex to Lake Ontario via outflow of the St. Lawrence River has been estimated (Halfon 1984). Probably most importantly, there are presently no known new sources of mirex to the Lake Ontario ecosystem and

therefore, using a mass balance approach in determining dynamics of transport for this chemical and predicting long-term persistence, one can assume an input event of limited and defined duration. Based upon available data, we believed it would be beneficial to develop a mass balance model for mirex in order to obtain a clearer understanding for the potential persistence of this material in the Lake Ontario environment, and to obtain a better feeling for the kind of "legacy" which society has created for other chemicals that have not yet reached zero discharge limits into the Lake.

The following initial conditions and assumptions were set for the Lake Ontario mirex mass balance model. Holdrinet, *et al.* (1978) estimated that 688 kg of mirex were dumped into the Niagara River from the late 1960s through 1976. Another source of

mirex was identified and described from the Oswego River in New York State (Scudato and DelPrete 1982). Sediment concentrations for mirex offshore of the Oswego River (10 ng/g) were observed to be similar to mirex sediment concentration offshore of the Niagara River (Scudato and DelPrete 1982; Thomas 1983). Therefore for purposes of these calculations, it was assumed that a load of mirex was discharged into the Oswego River similar in mass to the amount discharged into the Niagara River. Based upon this assumption, it was estimated that a total loading of 1.3×10^6 grams of mirex was input to the Lake Ontario system over a time span of 10 years (1966-1976).

Through the measurement of particulate material leaving Lake Ontario via the St. Lawrence River, Halfon (1984) calculated that approximately 8 percent of the total mirex load leaves the system by this pathway annually (Figure 39). By 1983 it appeared as though two thirds of the mirex in Lake Ontario was buried in sediments at least 6 cm deep (Eadie, *et al.* 1983), deep enough to be inaccessible to burrowing benthic invertebrates. Using the correlation of sediment depth to bacterial calculations in time through carbon-dating (Eadie, *et al.* 1983), it was estimated that 6.6 percent of the total mirex entering Lake Ontario was lost to sediment burial on an annual basis (Figure 39).

The other major loss of mirex from Lake Ontario is through the food web (Flint, *et al.* 1987) and is ultimately removed from the system by harvest of fish. Mean concentrations of mirex in depositional basins of Lake Ontario equal approximately 0.02 ug/g (Thomas 1983). Uptake of mirex by the benthos based upon body burden measures for the amphipod *Pontoporeia hoyi* (Whittle and Fitzsimons 1983; Flint, *et al.* 1987), equal approximately 0.2 ug/g. Lake trout body burden concentrations of mirex have varied from 2.5 ug/g (d.w.) in 1977 to 1.0 ug/g (d.w.) in the early 1980s (Whittle and Fitzsimons 1983). We can use the range of

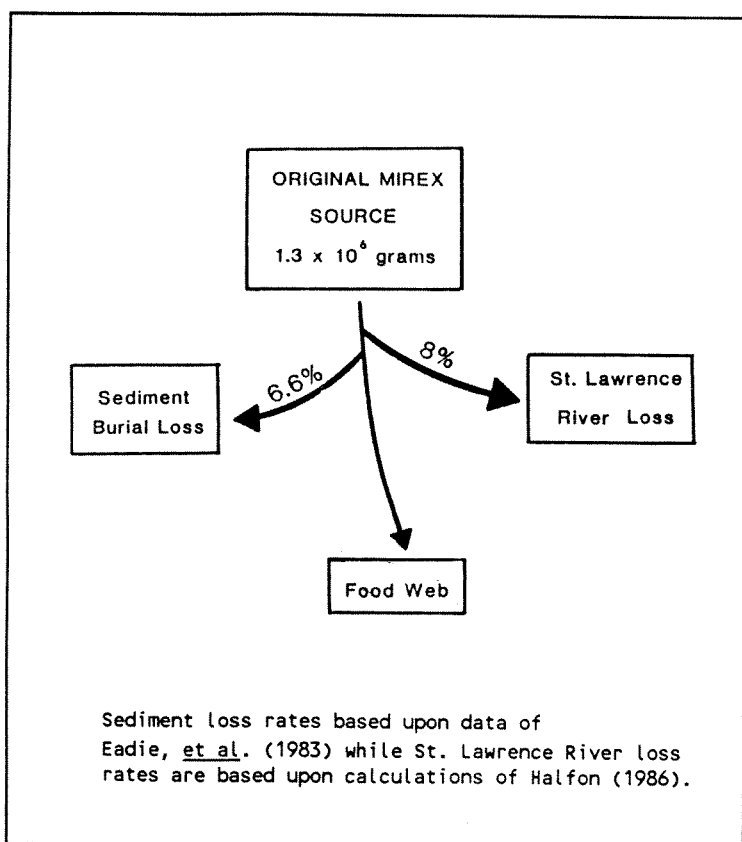


Figure 39. Estimated rates of transfer of mirex to various Lake Ontario pools that serve as sinks or losses of this toxic chemical.

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fish body burden amount (based upon lake trout measures) coupled with known harvest of fish from Lake Ontario to estimate the amount of mirex removed from the Lake by fishery harvests annually. Based upon the 1977 mirex body burden measures for lake trout (2.5 ug/g) and an annual average fishery harvest of 4.086×10^9 g (wet wgt.), the annual mirex removal rate by fish in 1977 equalled 2.0×10^3 g. In 1982 the mirex body burden was estimated to be 1.0 ug/g (dry wgt.) for lake trout and combined with an annual harvest rate of 4.086×10^9 g, it was estimated that 0.82×10^3 g of mirex were removed annually in the 1980s through fish harvest.

Using the initial mirex source amount of 1.3×10^6 g and the annual rates of mirex transfer to various system compartments, as well as losses to the Lake, a simulation mass balance model was developed. The model was used to project the rate of decay for mirex in Lake Ontario, assuming no new sources, and the amount of time for total decay to occur.

The results of this simulation are presented in Figure 40. An exponential decay of mirex is exhibited for loss due to sediment burial and loss via St. Lawrence River outflow. The fishery harvest loss rate is displayed as a constant over time with a slight change between the 1970 and 1980 periods due to changes in body burden measures between 1977 and 1982 (Whittle and Fitzsimons 1983). Based upon this simulation (Figure 40), in 1970 it was assumed that Lake Ontario contained a total of 1.3×10^6 g of mirex. In 1987 the model indicated that a total of 7.6×10^4 g of mirex remained in Lake Ontario and that the rate of removal by sediment burial and St. Lawrence outflow had decreased by more than 90% (Figure 40). In the year 2010 simulation results indicated that after losses of mirex to the food web (0.08×10^4 g), sediment burial (0.01×10^4 g) and river outflow (0.01×10^4 g), approximately 0.08×10^4 g of mirex would remain in the Lake Ontario ecosystem. This amount is equal to what is taken up by fisheries annually and thus we could assume, based

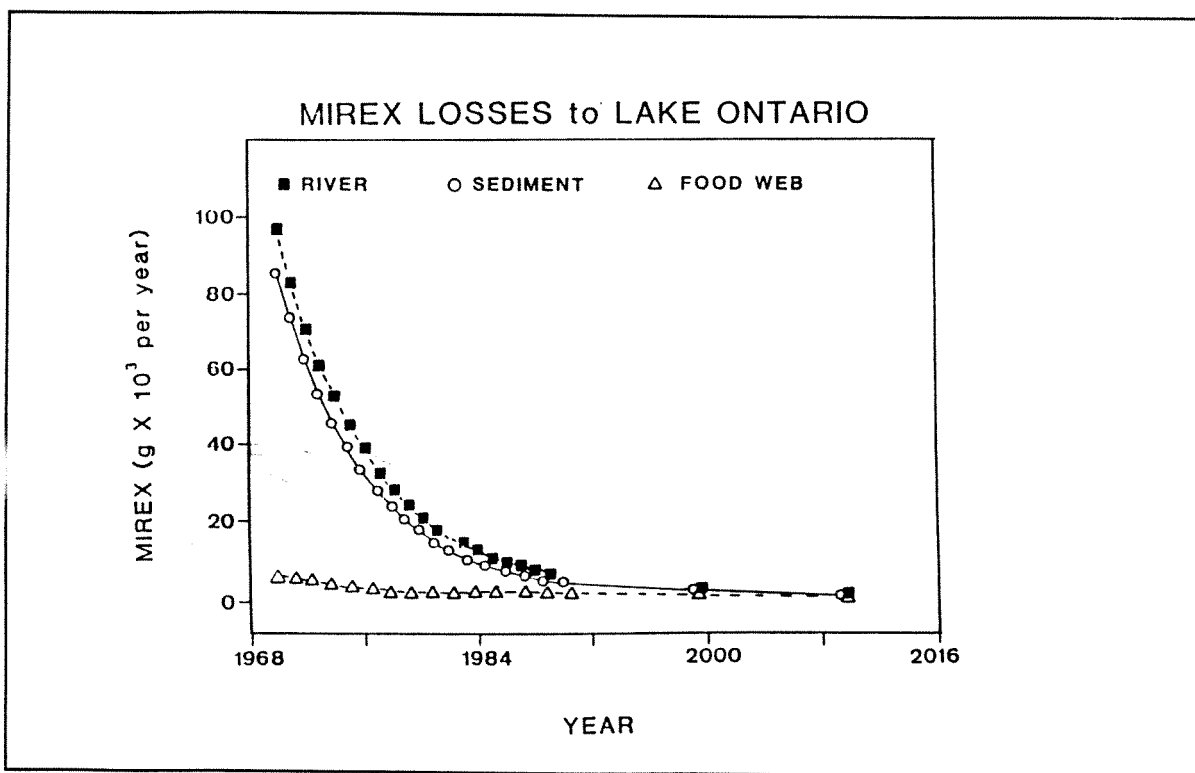


Figure 40. Mass balance model simulation results that estimate the loss of mirex from various Lake Ontario ecosystem components.

upon model results, that after the year 2010 mirex would be eliminated from the Lake Ontario ecosystem.

Therefore, based upon the simulation model described above, for a toxic contaminant for which we possess some data concerning transfer rates and some knowledge regarding total loading, it is projected that a minimum of 40 years is required for the Lake Ontario system to purge itself of this contaminant. This estimate is conservative, considering that there could be large variability in food web transport annually, varying lake levels will affect the amount of material exiting the Lake via the St. Lawrence River, and amounts of mirex initially lost through burial may be reintroduced to the system either by storm surge scouring of deep sediments or by burrowing benthic invertebrates that rework sediments deeper than 6 cm. Thus, the best available but conservative estimate for persistence of a toxic chemical in Lake Ontario, similar in characteristics to mirex, as described here, would be a minimum of 40 years.

The above phenomena, which are thought to be occurring presently in some of our Great Lakes, can only be adequately considered with an appreciation for the "ecosystem approach" which was emphasized by the 1978 Great Lakes Water Quality Agreement. By actively working this approach we will be able to make better use of our existing resources, prevent future surprises, and make progress toward development of sound solutions to the major problems facing our Great Lakes. But if we only pay "lip-service" to the ecosystem approach, we will never be able to appreciate the interconnections of the characteristics described above or those characteristics that we don't even know about yet. And we will not be able to improve upon management policy through attempts to balance social needs with environmental protection.

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CHAPTER 8

CONCLUSIONS

Although not always recognized due to some of the burdens placed on society because of various pollution problems, our Great Lakes heritage influences all ways of life and is of tremendous value to the peoples of the region, as well as to the U.S. and Canada as a whole. For example, although the Great Lakes basin represents only one sixth of the total U.S. land area, its contribution to the national gross product far exceeds this fraction. Personal income and retail sales in the Great Lakes basin represent 33% and 30%, respectively for all U.S. income and sales (Federal Reserve Bank of Chicago 1985). Manufacturing value and cash receipts from farm marketings for the Great Lakes basin each contribute 36% to national figures.

Fishery resources also represent a significant economic value to the Great Lakes region. For example, recreational sport fishing had an economic impact of \$2.9 billion in 1984 to New York State. Based upon the taxes collected from this income contrasted to the amount of money that New York put into managing fisheries in support of the sport fishing industry, a cost/return ratio to the State treasury for fishing recreation on the Great Lakes ranged from 1:7 to 1:20, which represents a very good investment to the State.

In contrast to the Great Lakes economic value to society summarized above, however, the commitment of funds for environmental problem solution and scientific investigation in support of sound policy formulation on the Great Lakes by the federal government falls short. The estimated net federal funds flow to all states in the U.S. is lowest to states surrounding the Great Lakes and the ratio of federal expenditures to estimated tax

revenues on a national scale is also lowest in the Great Lakes states (Federal Reserve Bank of Chicago 1985).

More than 30 percent of the U.S. population is located in and around the Great Lakes watershed, occupying one sixth of U.S. land area (Federal Reserve Bank of Chicago 1985). This burgeoning human population and the continuing development of complex industrial technology in the basin have inevitably led to increases both the quantity and diversity of waste materials discharged into these aquatic environments. Historically, the Great Lakes were thought to be so vast that waste disposal would have no measurable damaging impact. It is now known that these large water bodies are being seriously degraded. Lake Ontario, for example, has experienced contamination from industrial, municipal, non-point atmospheric, and landfill leachate releases. These contaminants include DDT, mirex, PCBs and other persistent toxic substances that have entered the Lake in the past two decades (e.g. Reinke *et al.* 1972; Kais 1974; Haile 1975). Since the Great Lakes provide a significant source of drinking water to much of the U.S. and Canadian populations living in these watersheds there is no greater threat to the economic viability and natural health of the Great Lakes than the introduction of the persistent toxic substances to what we now view as irreplaceable water resources.

Clean, usable water resources are an essential element in the quality of life which makes the Great Lakes region an attractive place to live and do business. Toxic contamination is a threat to the quality of life. It can lead to human health problems such as changes in infant

behavior, genetic damage, disorders of the reproductive system, and cancer. Toxic contamination has also been identified as the source of tumors in fish and damage to aquatic plants and animals.

Environmental contamination raises a number of questions regarding effects to ecosystems and societal health. For example, what is an unsafe level of contamination, either to fish or humans? While there has been some progress made in addressing this question, more collective thinking is required to change public and government attitudes and behavior. The lack of change and the slowness of response regarding research needs and policy implementation may be due to the fact that the right questions have not been asked in the right ways.

Are questions being asked in a way that brings focus to our insufficient scientific understanding? Is the major issue water quality, ecosystem health, human behavior, human health, international legislation, or something else? Can new strategies evolve from discussions set in a spirit of cooperation and integration that reflect holistic assessment of Great Lakes toxic chemical problems?

Environmental protection agencies generally focus their efforts on determining the concentrations and trends of toxic chemicals in the Great Lakes. An alternative effort might focus upon the use, production, and disposal of long-lived toxic chemicals in light of ecosystemic changes and human health impacts. The latter approach is strategic, shifting the focus from concentrations to behavior, from lakes to ecosystems, and from chemicals to health.

Is the correlation of environmental abnormalities with the presence of toxic contaminants a signal that the health of the Great Lakes may be jeopardized? This concerns regulatory agencies that do not want the public exposed to a health hazard but at the same time want to promote the benefits of valuable Great Lakes resources.

However, we require more than just signals. We need a better understanding of what the specific issues are and what should be the focus of new data collection. The public needs reliable information to make judgments about utilizing and consuming resources from the Great Lakes. We need to integrate the various components of the toxic chemical issue and evaluate effects on ecosystems and humans. Uncertainty needs to be reduced in order to improve confidence in management strategies. It is both inappropriate and costly to place the burden of proof of harm from conceivably toxic chemicals on the general public, which is the exposed population. The scientific community has a vitally important role to play in seeking answers to these important questions.

At present there is not a consensus on which chemicals are harmful to humans in the Great Lakes ecosystem. There is a need to document what is known and what is unknown about exposure to the different chemicals. We also need to document what is known about the risk of exposure, independent of whether there is actually physical exposure to toxic compounds. Although there are no easy answers we must acknowledge that the world is not going to wait for a perfect state of our knowledge on the issue of toxics and human health. Recognizing that our scientific information is far from perfect, the charge we face is to:

- *synthesize the information we have right now and relate it to public policy recommendations,*
- *educate the public with the best information presently available,*
- *identify information gaps and develop new interdisciplinary research strategies, and*
- *develop long-term efforts towards scientific understanding in support of future public health policy and education.*

In addition, the relative lack of knowledge about the effects of toxic contamination can have expensive consequences for industries subject to water quality regulation. Regulators faced with the need to exercise their environmental protection responsibilities without sufficient knowledge of the effects of toxic contamination are often forced to adopt a "worse case" scenario which may, in some instances, result in over-regulation. We are just beginning to understand all the effects of introducing toxic substances into the fragile ecosystem of a Great Lake like Lake Ontario. To understand and control these contaminants will require a much more effective and comprehensive effort than is probably possible under the current mix of state, provincial and federal management programs that exist at the several levels of jurisdictions involved with Great Lakes governance.

Within the realm of environmental governance, there is an inherent conflict between natural areas and political jurisdictions in the Great Lakes region. Natural areas, such as Lake Ontario, can be defined because of the "natural" occurrences within them. Political jurisdictions exist for a wide variety of reasons, few of which ever relate usefully to natural phenomena. Thus, political jurisdictions are apt to cut across coastal zones or run along rivers, making environmental management an interjurisdictional activity. Also, by tradition, different levels of government have accepted, or been given, responsibility for different environmental components, thus further complicating the management of our Great Lakes resources. As a consequence, inefficiencies, gaps, and redundancies can occur, and, most important, often no one is responsible for integrating activities or looking holistically at science and turning it effectively into policy needs. The making of public policy and implementation of management decisions is a complex process and one that requires the best synthesis of scientific and technical knowledge

available to guarantee wise use and protection of Great Lakes resources.

Discussions of and judgments about Great Lakes management frequently revolve around what we might refer to as the trinity of ecosystem management concepts. These are that management should be 1) ecologically sound and holistic, 2) comprehensive and 3) integrated. The first of these requires that all significant elements of the system be incorporated in management. The second requires that the information be complete regarding the operation and dynamics of the system. The third concept requires that the management units relevant to ecosystem be coordinated and act together to maximize ecosystem sustainability and health (Hennessey and Robadue, 1987). Now while it is recognized that these concepts are normative goals and not necessarily operational management principles, knowledgeable observers frequently invoke such criteria in judging the adequacy of Great Lakes management. Hence, it is important to examine these concepts in light of the historical data bases that we possess on system function and the relationship of trends and patterns to man's anthropogenic influences on a system like Lake Ontario.

There can be little doubt that Lake Ontario is an extraordinarily complex ecosystem involving the dynamics of natural processes and the relationship between such processes and human activities and behavior. The natural complexity of a Great Lake, however, has features which make it particularly difficult to handle analytically (Hennessey and Robadue 1987). Among these are the following:

1. The components or subsystems are connected in a selected way. Everything is not always closely tied to everything else.
2. The impact of ecological events is not uniform. Different areas react in different ways.

3. Dramatic changes in behavior are natural to many ecosystems, and many of these changes are beyond man's means to predict.
4. Variability, not consistency, is the characteristic of ecosystems that enables them to adjust and therefore, to persist.

The dynamic and interdependence characteristics of the Lake ecosystem pose enormously difficult problems for the governance system. This is particularly the case when a number of environmentalists, planners and engineers require that the Lake's policy system be holistic, comprehensive and integrated. These management concepts are suggested presumably because it is thought that they will somehow be able to capture the complexity identified above. Unfortunately, these concepts require a management system which is simply not feasible or even desirable if it is impossible to apply these concepts in light of the constraints of the policy environment.

Because everything is connected, it is beyond our capacity to manipulate variables comprehensively. Because many components are interconnected, the whole of the environmental problem is beyond our capacity to control in one unified policy. We therefore have to find critical points of intervention which are tactically or strategically defensible points of intervention (Lindblom and Brooke 1963).

Following Lindblom, we see management as requiring points of strategic intervention. This is the only way to simplify tasks so as to make them manageable. In order to establish policies for managing Great Lakes resources, particular problems and issues must be identified, their importance estimated and approaches to intervention defined, and policies selected, implemented, and evaluated. This process can only be accomplished through the synthesis of research results that will help to set

sound public policy. But interpretation of these research results must be set in a framework that is transmittable to policy-makers. Through the assessment of ecosystem trends and water quality patterns presented here for Lake Ontario, as well as the identity of possible interconnected parts of the system that need to be considered in an integrated fashion, we have been able to identify some of the key points of system intervention that will guide future policy formulation and management decisions on this Great Lake.

Long-term changes in any or all of our Great Lakes basins, which might include anthropogenic changes or natural phenomena (e.g., global climate change from the greenhouse effect), will result in impacts to biophysical systems and these impacts will express themselves in social and economic terms. We need to not only improve our understanding of biophysical systems but also improve our understanding of how anthropogenic change and natural environmental variability are expressed in political systems and economic markets. The integration that can occur in this regard between social scientists and biophysical scientists makes their respective research more relevant, more widely used, and more generally applicable to environmental management issues.

Society now demands that governments direct their attention to protecting and rehabilitating Great Lakes resources. Significant advances have been made in our understanding of how the Great Lakes operate but until our knowledge is increased substantially and integrated into a more holistic perception of system function, there will be uncertainty as to which management strategies to apply and what results to expect from a given strategy, or combination of strategies. For Lake Ontario, an attempt has been made here through the assessment of long-term trends and synthesis of data, to improve the science upon which regulation is based and advance management practices which provide needed protection and remediation at the lowest cost to industry and society.

State of a Great Lake

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