S.O.L.E.C.

1994 State of the Lakes Ecosystem Conference
Background Paper

Aquatic Community Health of the Great Lakes

August 1995

Environment Canada
United States Environmental Protection Agency
EPA 905-R-95-012

State of the Great Lakes Ecosystem Conference

Background Paper

AQUATIC COMMUNITY HEALTH OF THE GREAT LAKES

Joseph F. Koonce
Department of Biology
Case Western Reserve University
Cleveland, Ohio

August, 1995

Table of Contents

Acknowledgments

Many individuals have contributed to this paper. I have drawn heavily from the work of the Lake Ontario Pelagic Health Indicators Committee. Members of this committee and others who contributed material for the paper are:

John Eaton	U.S. Environmental Protection Agency
Uwe Borgmann	Great Lakes Laboratory for Fisheries and Aquatic Sciences
J. H. Leach	Ontario Ministry of Natural Resources
W. J. Christie	Ontario Ministry of Natural Resources (retired)
R. M. Dermott	Department of Fisheries and Oceans
E. L. Mills	Cornell University
C. J. Edwards	U.S. Department of Agriculture
R. A. Ryder	Ontario Ministry of Natural Resources
Michael L. Jones	Ontario Ministry of Natural Resources
William. W. Taylor	Michigan State University
S. R. Kerr	Bedford Institute of Oceanography
Terry Marshall	Ontario Ministry of Natural Resources
Glen A. Fox	Canadian Wildlife Service
Chip Weseloh	Canadian Wildlife Service
Joseph H. Elrod	U.S. Fish and Wildlife Service
Ora Johnannson	Great Lakes Laboratory for Fisheries and Aquatic Sciences
David Best	U.S. Fish and Wildlife Service
C. P. Schneider	NY Department of Environmental Conservation

The financial support of the USEPA, U.S. Fish and Wildlife Service, and the Great Lakes Fishery Commission for a leave-of-absence appointment as Ecosystem Partnership Coordinator for the Great Lakes Fishery Commission made this work possible. Thanks are also due to C. K. Minns, J. R. M Kelso, and J. W. Owens for particularly strong and helpful criticisms of the manuscript.

NOTICE TO READER

These Working Papers are intended to provide a concise overview of the status of conditions in the Great Lakes. The information they present has been selected as representative of the much greater volume of data. They therefore do not present all research or monitoring information available. The Papers were prepared with input from many individuals representing diverse sectors of society.

The Papers will provide the basis for discussions at SOLEC. Readers are encouraged to provide specific information and references for use in preparing the final post-conference versions of the Papers. Together with the information provided by SOLEC discussants, the Papers will be incorporated into the SOLEC Proceedings, which will provide key information required by managers to make better environmental decisions.

EXECUTIVE SUMMARY

By setting a goal of restoring the chemical, physical, and biological integrity of the Great Lakes, Canada and the United States have implicitly invoked an historical benchmark for assessing recovery. Relative to this standard, the Great Lakes ecosystems are extremely unhealthy. The catastrophic loss of biological diversity and subsequent establishment of non-indigenous populations is the most striking indication of degradation of the Great Lakes.

At least 18 historically important fish species have become depleted or have been extirpated from one or more of the lakes. Amplifying this loss of species diversity is the loss of genetic diversity of surviving species. Prior to 1950, Canadian waters of Lake Superior supported about 200 distinctive stocks of lake trout, including some 20 river spawning stocks. Many of these stocks are now extirpated, including all of the river spawners. The loss of genetic diversity of lake trout from the other lakes is even more alarming, with complete extirpation of lake trout from lakes Michigan, Erie, and Ontario and only one or two remnant stocks in Lake Huron.

Accompanying this loss of diversity was a series of invasions and introductions of exotic species. Since the 1880s, some 139 non-indigenous species have become established in the Great Lakes. Non-indigenous fish species that have established substantial populations include sea lamprey, alewife, smelt, gizzard shad, white perch, carp, brown trout, rainbow trout, Chinook salmon, coho salmon, and pink salmon. Other major invasions include the spread of purple loosestrife into Great Lakes wetlands, and the population explosions of zebra and quagga mussels in Lake St. Clair and Lake Erie. Together, the non-indigenous species have had a dramatic and cumulative effect on the structure of the aquatic communities of the Great Lakes, and their persistence poses substantial problems for the restoration and maintenance of native species associations.

Changes in the biological diversity of the Great Lakes are caused by a host of chemical, physical, and biological stresses. Major stresses include:

- large-scale degradation of tributary and nearshore habitat for fish and wildlife;
- imbalances in aquatic communities due to population explosions of invading species such as sea lamprey, alewife, white perch, and zebra and quagga mussels;
- reproductive failure of lake trout;
- alterations of fish communities and loss of biodiversity associated with over-fishing and fish stocking practices; and
- impacts of persistent toxic chemicals on fish and wildlife.

Biological stresses have caused a greater decline in health of the Great Lakes than physical and chemical stresses. Historically, over-fishing and introduction of exotic species have had devastating effects. Loss and degradation of aquatic habitat, however, are also important sources of stress. In many cases, the effects of habitat loss are obscured by restructuring of aquatic communities and through compensations by managers. In Lakes Ontario and Michigan and to a lesser extent in Lakes

Huron and Superior, stocking of salmonid predators compensates for the effects of degraded habitat. Without these stocking programs, there is insufficient reproduction of non-indigenous salmonids and lake trout to sustain existing populations. In Lake Erie and other lakes with reproducing predators, fish communities have lost tributary spawning stocks, and the species composition of the fish community reflects less dependence upon nearshore and tributary habitat for spawning and nursery areas.

Persistent, toxic contaminants are also affecting fish and wildlife populations in the Great Lakes. Observed effects include alteration of biochemical function, pathological abnormalities, tumors, and developmental abnormalities. Contaminants are suspected of playing a role in recruitment failures of lake trout, but the effects of exposure to contaminants are less clear for fish than for wildlife. Eleven species of wildlife in the Great Lakes show evidence of contaminant impacts. Three species (bald eagles, cormorants, and herring gulls) provide the best evidence both of the severity of historical impacts and of recent improvements due to reductions in loadings. However, the reproductive success of breeding eagle pairs eating Great Lakes fish remains lower than those nesting inland, and occasional, local incidence of deformities indicate continuing contaminant problems in some areas. Despite these encouraging trends, exposures to persistent, toxic chemicals remain high enough to continue producing effects on fish and fish-eating wildlife.

Although the health of the Great Lakes remains degraded by historical standards, many indicators show signs of improvement. The extent of changes in the Great Lakes, however, poses a serious challenge to obtaining consensus on specific objectives for the restoration of chemical, physical, and biological integrity. Scientifically, it is possible to identify alternative configurations of aquatic communities that are consistent with fundamental ecological principles and the goals of the Great Lakes Water Quality Agreement. With the possible exception of Lake Superior, the degradation of historical community structure caused by various biological, physical, and chemical stresses coupled with the establishment of large numbers of non-indigenous species means that a return to pre-settlement conditions may not be possible. The question of how closely restored aquatic communities should resemble historical conditions is more an issue of social preference than a technical or scientific issue. Ultimately, the people living around the Great Lakes must decide what their objectives are for ecosystem restoration and maintenance. Only with such specific objectives will it be possible to decide on the current health of the Great Lakes and to establish priorities for dealing with stresses responsible for impairment of that health.

1.0 Introduction

This paper summarizes current understanding of the health of the aquatic communities of the Great Lakes. The range of communities includes aquatic species and terrestrial species (fish-eating birds, mammals, and reptiles) that rely on aquatic food webs of the lakes or on habitat with associated wetlands and other near-shore environments. The need for this summary comes from the adoption of an ecosystem approach to management of the Great Lakes. More holistic than a pollutant-by-pollutant approach to improvement of water quality associated with earlier laws and agreements, the Great Lakes Water Quality Agreement of 1978 committed Canada and the U.S. to a long-term goal of "...restoring and maintaining the chemical, physical, and biological integrity of the waters of the Great Lakes basin ecosystem." Relying on an analogy to human health, the restoration of integrity has become synonymous with returning the ecosystems of the Great Lakes to a healthy state. Implicit in this goal is the recognition that abuses due to the past 200 years of human activity in the Great Lakes basin have reduced the health of the Great Lakes. The challenge is to balance ecosystem restoration and maintenance with human development. The necessity of this balance is the fundamental premise of "ecologically sustainable economic development" advocated by the Brundtland Commission (World Commission on Economic Development, 1987).

Evaluation of the health of the aquatic community of the Great Lakes is complicated. Impairments to health of individual fish and wildlife are possible to detect through a variety of indicators (e.g. tumor incidence, incidence of developmental anomalies, and incidence of disease and parasitism), but the specific causes of health impairments and their population-level effects are often ambiguous. For example, levels of mixed function oxidase enzymes are influenced by exposure to a wide range of anthropogenic and natural substances, and such indicators of exposure may or may not indicate an illness condition.

Assessing the health of populations and communities is even more complicated for at least three reasons. First, because different causal factors may produce similar effects on populations, identification of factors responsible for particular population impairments (elevated mortality or morbidity rates or decreased reproductive rates) is difficult. Second, populations and communities are adaptive. Healthy communities share common functional integrity: ability to self-regulate in the presence of internal or external stresses and ability to evolve toward increasing complexity and integration. Thus, many different, "healthy" states may be functionally equivalent. Third, the Great Lakes are unique and very disturbed ecosystems. Many of the original communities no longer exist, and introduced species have established viable if not dominant populations. Without undisturbed communities to serve as reference benchmarks, the determination of the wellness of an ecosystem requires a value judgment.

1.1 Concepts of Ecosystem Health

The concept of ecosystem health is often more symbolic than functional. As with human health, maintenance and restoration of ecosystem health admits both curative and preventative approaches. The curative approach finds what is wrong and fixes it while the preventative approach attempts to minimize the risk of illness. Considering human health, the dichotomy of the two approaches yields the current dilemma with technological approaches to medicine--elimination of illness does not necessarily produce wellness. For humans, wellness is a harmony of mind and body, and extensions of the health analogy to ecosystems falters because we lack a definition of wellness (cf. Minns, in press). In the context of ecosystem management, we can address the causality problem by associating stresses (e.g. pollution loading, habitat destruction, and overexploitation) with impairments of beneficial uses. Without a wellness concept, however, what constitutes an overall assessment of ecosystem health is a value judgement.

To add objectivity to the concept of ecosystem health requires consideration of the adaptive potential of ecological communities. Holling (1992) argues that a small set of processes structure ecosystems. Within constraints of habitat characteristics and climate variability, ecological communities display cycles that are characteristic of various ecosystem types. The structure of climax communities of terrestrial ecosystems, as with their analogs in the aquatic communities of the Great Lakes (cf. Loftus and Regier 1972), exists in balance with patterns of disturbance. The result is a predictable set of patterns of ecosystem dynamics in which community composition changes through a series of recognizable states before returning to a climax state (i.e. persistent state). Climax states and succession transients are thus common elements to all natural ecosystems, and a concept of ecosystem health must include reference to the feedback mechanisms that govern natural cycles and persistence of climax states. As Rapport (1990) states, ecosystem health depends upon the integrity of the homeostatic mechanisms, and "integrity refers to the capability of the system to remain intact, to self-regulate in the face of internal or external stresses, and to evolve toward increasing complexity and integration."

Natural, undisturbed ecosystems would seem to be good benchmarks for integrity or wellness. Ryder and Kerr (1990) argue that ecological communities evolve toward co-adapted or "harmonic" assemblages of species and that the status of the native species associations in ecosystems is an indication of their integrity. However, chronological colonization and invasion patterns are accidental, and multiple native species associations could evolve given slightly different compositions of colonizing species.

This issue becomes especially important when ecosystem restoration is the main challenge as in the Great Lakes. The original ecological communities no longer exist, and many exotic species have established viable and at times dominant populations. Justification of preference for specific community composition may be aided by historical analysis (e.g. Ryder 1990), yet alternate composition, with similar ecological function, is certainly possible. At some level, the decision about which ecological community to pursue in restoration becomes a social preference. Scientific notions may contribute to the decision, but ultimately people must decide what their objectives are for ecosystem restoration and maintenance. Hence, what constitutes "ecosystem wellness" is, in part, a value judgment.

The notion of ecosystem health is also hierarchical. The integrity of an ecosystem is a complex function of the health of its constituent populations, the biological diversity of its ecological communities, and the balance between ecological energetics and nutrient cycling as constrained by physical habitat. At some levels in such a hierarchy, illness is much easier to detect. Evaluation of the health of fish and wildlife populations, for example, admits a direct extension of notions of human health in which density, growth, incidence of disease, morbidity, and mortality statistics are accepted measures of healthiness. The health of an individual organism, in turn, is judged relative to normal biochemical and physiological functions. Indications of impaired health derive from biochemical, cellular, physiological, or behavioral characteristics, which can be observed and, to some degree, be associated with known causes. Impaired health of an individual may manifest itself in its population through effects on reproduction or mortality, and the proportion of unhealthy individuals in a population may influence the entire ecological community by altering the balance of competition and predator-prey relations that provide its dynamic structure.

1.2 Great Lakes Aquatic Ecosystem Objectives

The ecosystem approach, which was advocated with the 1978 Great Lakes Water Quality Agreement, requires ecosystem objectives. With the adoption of the 1987 Protocols, specific objectives were set forth in the Supplement to Annex 1:

> *Lake Ecosystem Objectives.* *Consistent with the purpose of this Agreement to maintain the chemical, physical and biological integrity of the [waters] of the Great Lakes Basin Ecosystem, the Parties, in consultation with State and Provincial Governments, agree to develop the following ecosystem objectives for the boundary waters of the Great Lakes System, or portions thereof, and for Lake Michigan:*
>
> > *(a) Lake Superior*
> > *The Lake should be maintained as a balanced and stable oligotrophic ecosystem with lake trout as the top aquatic predator of a cold-water community and the Pontoporeia hoyi as a key organism in the food chain; and*
> > *(b) Other Great Lakes*
> > *Ecosystem Objectives shall be developed as the state of knowledge permits for the rest of the boundary waters of the Great Lakes System, or portions thereof, and for Lake Michigan.*

The first effort of the Parties to draft ecosystem objectives for the other Great Lakes grew out of the activities of the Ecosystem Objectives Working Group (EOWG) for Lake Ontario (Bertram and Reynoldson 1992). Five ecosystem objectives have emerged from this effort:

The waters of Lake Ontario shall support diverse healthy, reproducing and self-sustaining communities in dynamic equilibrium, with an emphasis on native species.

The perpetuation of a healthy, diverse and self-sustaining wildlife community that utilizes the lake for habitat and/or food shall be ensured by attaining and sustaining the waters, coastal wetlands and upland habitats of the Lake Ontario basin in sufficient quality and quantity.

The waters, plants and animals of Lake Ontario shall be free from contaminants and organisms resulting from human activities at levels that affect human health or aesthetic factors such as tainting, odor and turbidity.

Lake Ontario offshore and nearshore zones and surrounding tributary, wetland and upland habitats shall be sufficient quality and quantity to support ecosystem objectives for health, productivity and distribution of plants and animals in and adjacent to Lake Ontario.

Human activities and decisions shall embrace environmental ethics and a commitment to responsible stewardship.

These objectives have been incorporated into the draft Lakewide Management Plan for Lake Michigan. The Lake Superior Binational Program, which was created by the parties for a demonstration of the zero discharge objective for toxic contaminants, has also used the framework of these objectives to propose extensions of the ecosystem objectives adopted for Lake Superior in the 1987 Protocols.

1.3 Indicators

1.3.1 Fish and Wildlife Health Indicators

Indicators of individual fish and wildlife health have developed from concern with disease and abnormalities in physiology and behavior. Living organisms respond to environmental stresses through a variety of physiological and behavioral mechanisms. Beitinger and McCauley (1990) review the notion of a general adaptation syndrome at a physiological level that includes a primary response in the endocrine system and a secondary response involving blood and tissue alterations. Impaired health occurs when these adaptations are not sufficient to permit normal function. Assessments of fish and wildlife health in the Great Lakes have employed a range of specific indicators of these physiological responses to stress. A partial list would include:

Indicator	Associated Stress
Induction of Mixed Function Oxidase Enzymes (MFO), e.g. P450 1A1.	Induction indicates exposure to hydrophobic planar chemicals such as PAHs, PCDD, and PCBs
Inhibition of Amino Levulinic Acid Dehydratase (ALA-D)	Inhibition indicates exposure to inorganic lead compounds
Hepatic Porphryia	Elevated levels of highly carboxylated porphyrins (HCPs) is indicative of exposure to organochlorines (PCBs, HCB, and TCDD)
Hepatic Vitamin A (Retinol)	Reduction in levels indicates unsatisfactory nutritional status and/or effects of exposure to chemicals such as TCDD
Thyroid Related Abnormalities	Changes indicate altered metabolic status due to changes in habitat and/or exposure to goitergins
Tumor Incidence	Indicates toxic exposure to PAHs or other carcinogens, but also may be due to viral and bacterial agents.
Fin Ray Asymmetry	Indicates poor environmental quality
Congenital Malformations	Increased incidence indicates excessive exposure to developmental toxins and/or maternal health status
Disease Incidence	Increased incidence of Bacterial Kidney Disease (BKD) and other bacterial and viral diseases in fish indicate nutritional or chemical stress
Parasite Incidence	Increased incidence indicates pollution or stress condition

These indicators represent responses of fish and wildlife to various stresses in the environment, but their diagnostic specificity varies as effects move from biochemical to population levels. Some biochemical indicators, such as induction of MFOs, are non-specific and indicate only exposure to some types of organochlorines, which may come from anthropogenic or natural sources. These exposures may or may not result in illness. Translation of the exposure indicators to health assessment is not always straightforward (cf. Munkittrick 1993). Nevertheless, these indicators together give indications of the quality of the environment with respect to factors causing stress on biochemical and physiological processes.

1.3.2 Community Health Indicators

Like individual health indicators, the purpose of developing community health indicators is to detect and diagnose pathology. Indicators of the health of an ecological community, however, are imbedded in a hierarchical set of ecological interactions and in a poorly coordinated hierarchy of ecosystem management jurisdictions and initiatives (cf. Evans, Warren, and Cairns, 1990). Without an integrating framework, indicators of community health tend to focus on those parts of an ecosystem most valued by their proponents. As Koonce (1990) has argued, this lack of an integrating framework creates obstacles for the use of indicators to characterize trends for the entire Great Lakes basin or to guide management actions to correct the pathologies. A pathology from one perspective, after all, may be a beneficial condition to another. Gilbertson (1993), for example, argues that the requirement for supplemental stocking of salmonids to work around the failure of lake trout reproduction in Lake Ontario is symptomatic of a pathology, but many recreational fishers prefer to catch non-native Chinook salmon and view emphasis on lake trout rehabilitation as undesirable if in doing so the Chinook fishery declines. Ideally, community health indicators should follow from the objectives for ecosystem management, but as discussed below, ecosystem objectives are often not specific enough to provide a basis either for deriving quantitative end points consistent with the objective, or for guiding the selection of an appropriate set of indicators with which to monitor trends in ecosystem health and to specify corrective action.

Attempts to develop sets of indicators have arisen in parallel with government mandates for ecosystem management. Within the International Joint Commission (IJC), the Science Advisory Board created an Aquatic Ecosystem Objectives Committee (AEOC) to develop ecosystem objectives and indicators for the Great Lakes. These efforts led to proposed indicators based on indicator species for oligotrophic portions of the Great Lakes (Ryder and Edwards 1985) and for mesotrophic areas (Edwards and Ryder 1990). Following the 1987 revisions to the Great Lakes Water Quality Agreement, Canada and the U.S. established a Binational Objectives Development Committee, which subsequently formed the Ecosystem Objective Work Group (EOWG) to continue development of ecosystem objectives and indicators. Various national initiatives have also complemented the binational efforts. Noteworthy is the Environmental Monitoring and Assessment Program (EMAP) of the Environmental Protection Agency. The primary goal of the Great Lakes EMAP strategy under development (Hedtke *et al.*, 1992) is to estimate current status and trends of indicators for the ecological condition of each of the Great Lakes. As a result of these various initiatives, formulation of indicators of aquatic community health of the Great Lakes is only just beginning, and the indicators summarized here are thus far less robust than those for fish and wildlife health.

Community health indicators fall into three categories: indicator or integrator species, ecosystem function indicators, and composite indices of ecosystem integrity.

An example of the first category is the use of lake trout (*Salvelinus namaycush*) and *Pontoporeia* for oligotrophic ecosystems (Ryder and Edwards 1985) and walleye (*Stizostedion vitreum*) and burrowing mayfly (*Hexagenia limbata*) for mesotrophic waters (Edwards and Ryder 1990). These species satisfy fundamental criteria for using species as surrogates of community health (Edwards and Ryder 1990): a strong integrator of the biological food web at one or more trophic levels; abundant and widely distributed within the system; and perceived to have value for human use to make sampling easier.

An example of indicators of ecosystem function is the proposed use of biomass size spectra (Sheldon *et al.* 1972) as measures of ecosystem health (Kerr and Dickie 1984). Table 1 lists this and other candidate indicators of ecosystem function that have been evaluated by the Lake Ontario Pelagic Community Health Indicator Committee.

Finally, there are a wide variety of examples of composite indices (Karr 1981; Steedman 1988; Rankin 1989; Yoder 1991; and Minns *et al.* in press). As Rapport (1990) notes, these indices are based on a number of variables, but usually cover biotic diversity, indicator species, community composition, productivity, and health of organisms. The Dichotomous Key, designed to assess the health of the oligotrophic aquatic ecosystems (Marshall *et al.* 1987), is in fact an example of an aggregate index using lake trout as a surrogate for the biological integrity of oligotrophic portions of the Great Lakes.

Table 1. Indicators proposed by the Lake Ontario Pelagic Community Health Indicator Committee for ecosystem structure and functional energy flow (after Christie 1993).

Indicator	Historical Data	Methodology for Collection	Status of Assessment	Interpretive Status
Biomass or production size spectrum	None - some current data in Sprules group	Traditional net sampling, various techniques for each organism targeted ,little calibration.	Currently developing new sampling methodologies	Has utility in displaying the entire structure of the ecosystem.
Yield of piscivores	Long-term commercial statistics. Some recent creel census data.	Requisite reports from commercial fishermen, spot surveys of anglers and charter boats.	Inadequate bridging between old and new data series. Need better institutional assessment data and more comprehensive creel, and charter data.	Presently used to measure fisherman satisfaction. Convergence on predicted yield estimate can measure ecosystem health.
Ratio of Piscivore to prey biomass	Comparable to above; more data available for the predator species than for the prey, especially nearshore.	Traditional fishery tools; gillnets, trawls, trapnets, seines.	Inadequate assessment of small nearshore species especially. No bridging between inshore, offshore programs. Biased estimates of relative biomass. Currently developing new sampling methodologies based on sonar.	Should measure approach to steady state conditions, and deviations therefrom. Rigorous attention to sampling routine should allow early-warning use of variance, and trend data.
Fraction of yield as native fish.	Lake trout, rainbow trout data back to the 1950s, Chinook salmon more recent.	Fin clips used in past, nasal insert tags currently used in all larger fish released. Otolith, scale, and fin ray abnormalities used for fish smaller at release, and for F_2 and later recoveries.	Methods of differentiating genetic origins of naturally produced fish still developmental.	Data presently analyzed in the form needed.
Zooplankton size distribution.	Some data available from 1972. Continuous at the CCIW BioIndex stations.	Standard techniques used. Recently extended by new computerized count-measure procedures.	Currently applied in part-spectrum applications. All collections extant for series comparisons. Inshore data not consistently collected.	Not expressly used in present lake reporting, and especially useful when compared with nearshore data, and placed in the context of other indicators.
Total P levels <= 10 mg/l	Monthly surveillance (1976-1981); biannual survey (1982-present)	Discreet depth samples at 1 meter	Adequate methods currently being used. Consistent comparisons with nearshore conditions desirable.	Analysis ongoing and reliable. Good when used in conjunction with other indicators; provides information on the baseline productivity of the lake, and linkage to future biological problems related to return to excess P loads.
Fish species diversity.	Standard gillnet, trawl, trapnet, seine collections. Gillnet data continuous since 1957 and 1958 in Bay of Quinte and Kingston Basin. Trawl data continuous since 1972 in all areas. Broken series for the others.	Conventional net sampling. Programs need broadening to include shoreline and small species, integration to allow comparison within and between series.	Analysis needs to focus on evenness component of diversity. Statistical analysis of variance in each zone should measure improving health, and the reverse.	Conservative property. Is robust when developed from comparable collection techniques.

2.0 Status and Trends for Fish and Wildlife Health

Toxic contamination of the Great Lakes is a widely-perceived threat to fish and wildlife health. A recent compilation by the Government of Canada of scientific literature on the effects of persistent toxic chemicals (Anon. 1991b) concluded that persistent chemicals have had a significant impact on fish and wildlife species in the Great Lakes basin. Observed effects include alteration of biochemical function, pathological abnormalities, tumors, and developmental and reproductive abnormalities. A possible consequence of these effects is a decrease in fitness of populations. Contaminant body burdens in fish and wildlife also have led to alerting the public through consumption advisories of a potential human health threat. On the whole, however, the effects of toxic contamination on wildlife are much clearer than for fish populations.

Fish populations in the Great Lakes do show evidence of exposure to toxic contaminants. Induction of some mixed function oxidases (MFOs) (i.e. those which result in elevation of ethoxyresorufun o-deethylase or aryl hydrocarbon hydroxylase activity) signals AHH receptor activation, which may result in unfavorable biological responses. Surveys of MFO activity in lake trout clearly indicate elevated levels in southern Lake Michigan and western Lake Ontario (see Figure 1). Because mixed function oxidase enzymes are induced by a variety of toxic chemicals, elevated MFO activity cannot be associated with specific toxic chemicals, nor is it possible to attribute specific health effects to these elevated enzyme activities. Nevertheless, the patterns of lake trout MFO activity coincide with geographic variation in contaminant loading. White sucker (*Catostomus commersoni*) also showed similar patterns of higher MFO activity in Lake Michigan and Lake Ontario, but also showed patterns of higher activity in the nearshore than in fish sampled in off-shore environments (see Figure 2). Impairment of lake trout reproduction in Lake Michigan seems to reflect this chemical contamination (Mac 1988), and, by similarity of circumstances, chemical contaminants may be contributing to reproductive failure of lake trout in Lake Ontario. Further clarification of the effects of chemical contaminants on population health of fish may rest on resolution of methodological issues (Gilbertson *et al.* 1990, Gilbertson 1992).

Circumstantial evidence is also strong for chemically induced carcinogenesis in Great Lakes fish. Summary of observations (Anon. 1991b) indicates that proof of causation of incidence patterns of tumors is lacking. Nevertheless, the overwhelming evidence leads to the conclusion (Anon 1991b):

> *There is strong circumstantial evidence that environmental carcinogens are responsible for the occurrence of liver tumours in brown bullheads from the Black, the Buffalo and the Fox Rivers, and possibly in bullheads from several other Areas of Concern. There is no "proof" that chemical carcinogens are responsible for liver tumours in walleye and sauger from the Keweenaw Peninsula, or in white suckers from western Lake Ontario. However, the limited geographic distribution of the effects and the association with contaminated environments indicates a chemical etiology.*

Not all fish diseases, however, have a chemically dominant etiology. Recent observation of outbreaks of bacterial kidney disease (BKD) among Chinook salmon (*Oncorhynchus tshawytscha*) in Lake Michigan, and the dramatic increase in their mortality in the late 1980s (see Figure 3), have not been linked to contaminants. The Great Lakes Fish Disease Control Committee concluded that "*...the chinook mortality problem should be considered the result of an ecosystem imbalance rather than the "fault" of any one pathogen.*" Although *Renibacterium salmoninarum* is the causative agent of BKD, they believe that the disease is stress-mediated and not a simple epizootic. However, they advise implementing hatchery practices to reduce the prevalence of *Renibacterium salmoninarum.* To that end, the committee has proposed a set of guidelines for the control of disease agents imported into the Great Lakes basin (Hnath 1993, Horner and Eshenroder 1993). Other "diseases" have been observed to wax and wane in various fish populations. Smelt populations in Lake Erie, for example, experienced an epizootic of parasitism by the microsporidian, *Glugea hertwigi*, in the 1960s (Nepszy *et al.* 1978).

Relative to fish, effects of toxic contaminants on wildlife species are more extensively documented. By 1991, various studies had identified contaminant-associated effects on 11 species of wildlife in the Great Lakes (Anon. 1991b). Affected species include shoreline mink (*Mustela vison*), otter (*Lutra canadensis*), double-crested cormorant (*Phalocrocorax auritus*), black-crowned night-heron (*Nycticorax nycticorax*), bald eagle (*Haliaeetus leucocephalus*), herring gull (*Larus argentatus*), ring-billed gull (*Larus delawarensis*), Caspian tern (*Sterna caspia*), common tern (*Sterna hirundo*), Forster's tern (*Sterna forsteri*), and snapping turtle (*Chelydra serpentina*). Of these, 9 species showed historical evidence of reproductive impairment due to contaminants (see Table 1, Anon. 1991b, p. 563). Temporal and spatial trends in samples of cormorants, bald eagles, and herring gulls provide important evidence for the magnitude of the effects of contaminants on wildlife health and recent improvements.

Cormorants began to nest in the Great Lakes earlier in this century. Estimates of abundance in the 1940s and 1950s indicated about 1000 pairs, but these numbers declined substantially through the 1970s (Scharf and Shugart 1981, Price and Weseloh 1986, and Weseloh *et al.* in press). Productivity studies clearly implicated reproductive failure evident in the early 1970s (Figure 4), resulting from DDE-induced egg shell thinning, as the cause of these declines. Since 1979 cormorant populations have increased substantially throughout the Great Lakes (Weseloh *et al.* in press), but prevalence of bill defects and other developmental anomalies throughout the 1980s suggest that sufficient amounts of PCBs and other toxic contaminants occurred in fish to influence the embryo development of these and other colonial, fish-eating bird species, particularly in Green Bay (Fox *et al.* 1991, Gilbertson *et al.* 1991).

Bald eagles have shown drastic declines throughout their North American range. Wiemeyer *et al.* (1984) suggested that toxic contaminants have contributed to these declines with DDT causing eggshell thinning and reproductive impairment. Restrictions on the manufacture and use of DDT, PCB, and other organic compounds seemed to reverse these trends, and within the conterminous U.S. the Fish and Wildlife Service reported that bald eagles had recovered from a low of 400 pairs nationwide in 1964 to 2700 pairs in 1989 (Anon. 1991b). Great Lakes populations have followed this recovery trend, but reproductive success of breeding pairs nesting on shorelines of the Great Lakes or on tributaries with adfluvial fish populations from the Great Lakes are lower than those nesting inland (Best *et al.* in press). Between 1966 and 1992, seven bald eaglets were found with abnormal bills, 16 per 10,000 banded (Bowerman *et al.* in press), and the U.S. Fish and Wildlife Service reported that four eaglets with deformities were found on Great Lakes shorelines in 1993 (Best personal communication, East Lansing Fact Sheet, July 8, 1993).

More than any other wildlife species, the herring gull has become an indicator of contaminant trends in the Great Lakes (Mineau *et al.* 1984). As year-round residents, adult herring gulls offer a monitoring opportunity to detect regional variability in contaminant stress that is not complicated by migratory patterns characteristic of other fish-eating bird species (Weseloh *et al.* 1990). Since 1974, the

Table 2. Temporal and geographic variations of productivity of Great Lakes Herring Gulls, 1972-1985 (after Table 10, Anon. 1991b, p. 601), expressed as 21 day-old chicks per pair.

	1972	1973	1974	1975	1976	1977	1978	1979	1980	1981	1982	1983	1984	1985
Lake Ontario														
Snake I.	0.21					1.01	0.86	1.60	1.49	1.73		1.34		
Scotch Bonnet I.	0.12	0.06		0.15		1.10	1.01			2.13				
Brother's I.	0.10													
Presq'ile Pk.	0.06													
Black Ant I.	0.08													
Mugg's I.						1.52	1.47	1.56		1.40			1.17	
Lake Erie														
Big Chicken I.	0.45													
Port Colborne			0.48	0.65	0.79		1.45			1.60				
Middle I.							1.70	1.63	1.62	2.10		2.17	0.95	1.00
Lake Huron														
Chantry I.				1.48		1.12	1.40	2.17	2.17	2.16		1.84		
Double I.							1.57	2.17	2.25	2.23		1.25	2.33	
Lake Superior														
Agawa Rk.				1.32	1.55		1.66	0.88	0.40	0.37	0.14	0.37	0.85	1.30
Granite I.							1.12	1.70	1.40	0.46	1.39		1.39	

Canadian Wildlife Service has maintained a long-term monitoring program for toxic chemicals through a network of 13 sites throughout the Great Lakes. In general, organochlorine residues in herring gull eggs have declined from higher levels in the early 1970s (Anon. 1991b, p. 332). As is the case with cormorants, temporal and geographic variation of productivity reflect these trends (Table 2). Reproductive success was low in the early 1970s and has improved since.

Although the etiology of these changes has not been rigorously determined, egg exchange experiments indicate both intrinsic and extrinsic factors were involved, and biochemical markers provide substantial indication that biochemical abnormalities are strongly associated with diets contaminated by polyhalogenated aromatic hydrocarbons (Fox *et al.* 1988). Gilbertson *et al.* (1991) have proposed mechanisms to account for these reproductive effects. According to Fox (1993), "*studies of impairments to health using such biomarkers as induction of mixed function oxidases, alterations in heme biosynthesis, retinol homeostasis, thyroid function and DNA integrity and various manifestations of reproductive and developmental toxicity in these birds suggests that the severity* *varies with time and location and generally decreased between the early 1970s and late 1980s. However, these studies confirm the continued presence of sufficient amounts of PCBs and related persistent halogenated aromatic hydrocarbons in forage fish to cause physiological impairments in these birds over much of the Great Lakes basin.*" Fox (1993) also argues "*these injuries are most prevalent and severe in, but not confined to, hotspots such as Saginaw Bay, Green Bay, Hamilton Harbour, and the Detroit River.*"

3.0 Status and Trends for Community Health

Objectives for restoration of the physical, chemical, and biological integrity of the ecosystems of the Great Lakes have not defined explicit interim goals. Realizing that pre-Columbian states of the Great Lakes ecosystems represented one definition of a "healthy" ecosystem, one interim goal for restoration could be re-establishment, to the maximum possible extent, of natural communities. Alternatively, an interim goal could be the restoration of a functional equivalent of historical communities. Although this issue (i.e. development of indicators and end points for ecosystem objectives) is under active consideration, the historical benchmark remains an important reference point with which to judge the extent of degradation of Great Lakes ecosystems and the prospects for various levels of restoration.

Any assessment of the status and trends of ecosystem health must begin with the catastrophic loss of biological diversity and subsequent establishment of non-indigenous populations. Fish play a major role in structuring aquatic ecosystems, as tress do in many terrestrial ecosystems (Steele 1985). Summaries of the changes in the fish species composition of the Great Lakes (Lawrie and Rahrer 1973, Wells and McLain 1973, Berst and Spangler 1973, Hartman 1973, and Christie 1973) reveal substantial alteration of the fish communities. Table 3 lists the species that have either disappeared from the lakes or have been severely depleted, but these losses belie a much more fundamental loss of genetic diversity among surviving indigenous species. Goodier (1981), for example, showed evidence that Canadian waters of Lake Superior supported about 200 spawning stocks, including 20 river spawning stocks, of lake trout prior to 1950.

Table 3. Summary of fish species lost or severely diminished by lake in the Great Lakes. An asterisk (*) indicates stocking programs exist to attempt re-introduction. Status codes are 1 (Depleted), 2 (Extirpated), and 3 (Extinct).

Common Name	Species Name	Superior	Huron	Michigan	Erie	Ontario
Lake sturgeon	*Acipenser fluvescens*	1	1	1	1	1
Longjaw cisco	*Coregonus alpenae*		2	2	2	
Lake herring	*C. artedii*	1		1	2	
Lake whitefish	*C. clupeaformis*				1	
Bloater	*C. hoyi*					2
Deepwater cisco	*C. johannae*		2	2		
Kiki	*C. kiki*	1	2	2		2
Blackfin cisco	*C. nigripinnis*	2	2	2		2
Shortnose cisco	*C. reighardi*		2	1		1
Shortjaw Cisco	*C. zenithicus*	1		3		
Burbot	*Lota lota*					1
Fourhorn sculpin	*Myoxocephalus*					3
Emerald shiner	*Notropis atherinoides*			2	1	
Atlantic salmon	*Salmo solar*					2*
Lake trout	*Salvelilnus namaycush*		2*	2*	2*	2*
Sauger	*Stizostedion canadense*				2	
Blue pike	*S. vitreum glaucum*				3	3

Accompanying these changes in diversity of Great Lakes fishes was a succession of invasions and intentional introductions of non-indigenous fish species. Species that have established substantial populations include: sea lamprey (*Petromyzon marinus),* alewife (*Alosa pseudoharengus*), smelt (*Osmerus mordax*), gizzard shad (*Dorosoma cepedianum*), white perch (*Morone americana*), carp (*Cyprinus carpio*),brown trout (*Salmo trutta*), Chinook salmon(*Oncorhynchus tshawytscha*), coho salmon(*O. kisutch*), pink salmon(*O.gorbuscha*), rainbow trout (*O. mykiss*). Since 1985, other species such as the ruffe (*Gymnocephalus cernuus*), the rudd (*Scardinius erythrophthalmus*), fourspine stickleback (*Apeltes quadracus*), and two species of goby (*Neogobius melanostomus* and *Proterorhinus marmoratus*) have also invaded the Great Lakes (Mills *et al.* 1993). Including these introductions, Mills *et al.* (1993) have documented 139 non-indigenous aquatic organisms (plants, invertebrates, and fish) that have become established in Great Lakes ecosystems.

The pre-Columbian species assemblages of the Great Lakes represented an adaptive complex that was an essential determinant of the wellness of Great Lakes ecosystems. The loss of so much diversity diminished the health of the Great Lakes, but recent efforts to restore fish communities raise the question of whether it is possible to establish a standard of functional equivalency to these historical fish communities. By launching an aggressive, bi-national program to control sea lamprey, which with overexploitation caused the extirpation of lake trout in Lake Michigan and Lake Huron as well as a substantial reduction in the lake trout of Lake Superior, the Canadian and U.S. governments prepared the way for an intensive stocking program to reintroduce lake trout, and introduce non-indigenous salmonid predators, to all of the Great Lakes. These efforts have certainly resulted in development of highly successful sports fisheries in the Great Lakes that surpasses historical communities in the range of species available to anglers. The stability of these fisheries, however, is not clear. Except for Lake Superior, the salmonid stocking programs are not complemented by sufficient natural reproduction to sustain current populations. The fisheries, in fact, are dependent upon the continuation of artificial propagation. Furthermore, the prey species complex that support these predators is also dominated by unstable populations of invading species like alewife and smelt. The loss of the highly adaptive coregonid complex and native lake trout stocks has thus left a void that introductions have so far failed to fill.

Indicators of ecosystem function have not been applied systematically to the Great Lakes, but some studies hint at continuing problems. Biomass size spectrum studies of Lake Michigan (Sprules *et al.* 1991) have shown promising results for the use of particle-size spectra in analyzing food web structure. Through this analysis, Sprules *et al.* (1991) found that piscivore biomass was lower than they expected. The imbalance in the food web appears to be limited availability of prey fish production to the mix of stocked piscivore species. Zooplankton size distribution, as a component of the biomass size spectrum, also indicates imbalance between planktivory and piscivory. According to the Lake Ontario Pelagic Health Indicator Committee (Christie 1993), a mean zooplankton size of 0.8 to 1.2 mm shows a healthy balance in the fish community. Over the period 1981 to 1986, the observed range of mean size of zooplankton was 0.28 to 0.67 mm (Johannsson and O'Gorman 1991), indicating excess planktivory. Emerging evidence for 1993, however, suggests that Lake Ontario may be undergoing an abrupt shift in zooplankton size with a collapse of the dominant prey fish population (E. L. Mills, Cornell University, personal communication). The recent trends in Lake Michigan and Lake Ontario may indicate that declines in productivity of both lakes associated with reduced phosphorus loading make these systems less able to sustain predator stocking levels that were successful earlier. Recent modeling studies of Lake Michigan and Lake Ontario (Stewart and Ibarra 1991; and Jones *et al.* 1993) indicate a strong possibility that excessive stocking of predators is de-stabilizing the food webs in these ecosystems.

3.1 Case Study: Lake Erie

The recent history of Lake Erie further illustrates how tenuous is the continuing effort to restore the health of the Great Lakes. As reviewed by Hartman (1973), the ecosystem integrity of Lake Erie reached its lowest point in the decade of the 1960s. The combined effects of eutrophication, over-exploitation of fishery resources, extensive habitat modification, and pollution with toxic substances had severely degraded the entire ecosystem of Lake Erie. Once-thriving commercial fisheries had all but disappeared and the populations of the last remaining native predator, the walleye, had fallen to a record-low level. Beginning in the 1970s, new fishery management strategies and pollution abatement programs contributed to a dramatic reversal. Lake Erie walleye fisheries rebounded to world-class status (Hatch *et al.* 1987), and point-source phosphorus loading has declined to target levels in the 1972 Great Lake Water Quality Agreement (Dolan 1993). These reductions were accompanied by a dramatic decrease in the abundance of nuisance and eutrophic species of phytoplankton (Makarewicz

1993a) and an associated decline in zooplankton biomass (Makarewicz 1993b). Surveys of the benthic macroinvertebrate communities further illustrate the improvement in the most degraded sediment areas of Western Lake Erie. Compared with surveys conducted in 1969 and 1979, Farara and Burt (1993) found that there was a marked decline in the abundance of pollution tolerant oligochaetes and that overall the macroinvertebrate community of Western Lake Erie has shifted to more pollution intolerant and facultative taxa.

The invasion of zebra mussels into Lake Erie has affected this recovery trend. Leach (1993) reported that associated with zebra mussel increases was a 77% increase in water transparency between 1988 and 1991, a 60% decrease in chlorophyll a, and a 65% decline in number of zooplankters. Although Leach (1993) has observed an increase in the amphipod *Gammarus* in nearshore benthic communities dominated by zebra mussels, Dermott (1993) has observed an inverse relation to abundance of *Diporeia* and the Quagga mussel, which appears to be a second *Dreissena* species. These abrupt changes in water quality and associated plankton and benthic communities make predictions about future status of the Lake Erie ecosystem highly uncertain. Despite the recovery of walleye, however, the causes of current trends of change in the structure and function of the Lake Erie ecosystem are dominated by effects of non-indigenous species. The extent of the changes in community structure of the Western and Central basins is so great that the historical species composition is unlikely to serve as an achievable benchmark with which to assess ecosystem health.

3.2 Oligotrophic Waters

The offshore, oligotrophic portions of the Great Lakes also seem to show variable recovery. The lake trout surrogate indicator (Edwards and Ryder 1985) is the only indicator of aquatic community health that has been systematically applied to the oligotrophic areas of the Great Lakes. As documented in Edwards *et al.* (1990), this indicator is a composite index, which is derived from a wide range of conditions necessary to sustain healthy lake trout stocks. The rationale for the use of lake trout as a surrogate for ecosystem health is based on the notion that lake trout niche characteristics and historical dominance in the Great Lakes provide the best basis to detect changes in overall ecosystem health. The index is based on scores from a Dichotomous Key of questions about lake trout or their habitat (Marshall *et al.* 1987). A score of 100 indicates pristine conditions. For the period 1982-85, Edwards *et al.* (1990) indicate that Lake Superior had the highest score (i.e. was the least degraded) followed by Lake Huron , Lake Ontario, Lake Michigan, and Lake Erie (Figure 5). The Dichotomous Key further allows dissection of the indicator score into components associated with various stress categories. In all cases except Lake Erie, contaminants are an important cause of lower indicator values (Figure 6).

Marshall *et al.* (1992) reported on historical and expected future trends in the lake trout indicator for the period 1950 to 1995. The overall value of the indicator showed a decline through the mid-1960s with a projected recovery by 1995 approaching 1950 levels (Figure 7). Ryder (1990) argues that this recovery pattern indicates that recovery to near pristine conditions is a reasonable goal. Dissection of the score into stress categories, however, indicates that contaminant problems are not improving as rapidly as other stresses (Figure 8). In an independent effort, Powers (1989) applied the Dichotomous Key to explore trends in the ecosystem health of Lake Superior and Lake Ontario. Her conclusions were similar to the findings of Marshall *et al.* (1992) for Lake Superior, but she found that Lake Ontario's trends indicated substantial and continuing imbalance.

Powers (1989) explored the possible effects of various fishery management schemes on the future health of the Lake Ontario. In 1973, the indicator showed a degraded state, and ecosystem health appeared to decline through 1983 in spite of a rather substantial recovery of recreational fishing (Figure 9). Future projections showed a recovery to the 1973 level as rehabilitation of lake trout approached the goals set in the Lake Trout Rehabilitation Plan for Lake Ontario (Schneider et al., 1985). Other aspects of the Lake Ontario system health profile (Figure 9), however, are more troubling. In spite of achieving some of the interim goals for lake trout rehabilitation by 1988, the system health of Lake Ontario resists exceeding the degraded condition in 1973. Over the period 1973 to 1988, the lake trout population and other salmonid populations have increased markedly due to intensive stocking efforts. From the perspective of fish management agencies and the recreational fishing industry, these changes represent successful restoration of an extremely degraded fish community. The indicator, however, implies that this rehabilitation effort did not increase system health. Closer analysis of the stress categories (Figure 10) reveals that toxic contamination has contributed significantly to the decrease in system health. Further recovery of system health in Lake Ontario seems to be hindered by fundamental shifts in the fish community (Environmental Biotic stresses), future levels of exploitation (Exploitation stress), and continuing toxic contamination. Although some of these stress continue to improve, an indicator based on historical benchmarks for lake trout, in contrast to one based on the degraded state in 1960s, does not show any indication of improvement of system health despite a massive investment of resources in the rehabilitation of Lake Ontario.

Composite indices other than the Dichotomous Key of Marshall *et al.* (1987) have also been applied to portions of the Great Lakes. The Ohio Environmental Protection Agency, for example, has attempted to characterize the state of the estuarine fish communities in Ohio waters of Lake Erie (Thoma, unpublished report, OEPA). Using an Index of Biotic Integrity (IBI), the Ohio EPA found that only one of fourteen estuaries sampled met minimal integrity and health criteria (Figure 11). Factors responsible for the degraded state of the estuarine communities include extensive habitat modification, point source discharges, and diffuse, non-point sources effects preclude most sampled sites from attaining minimal goals. However, the most serious degradation is the modification of wetlands in the estuaries (Thoma, unpublished report).

4.0 Management Implications

The Great Lakes today do not meet current ecosystem objectives. In recent years, various indicators show improving conditions in all lakes. All of the lakes have some extremely degraded areas associated with local pollution sources. Apart from its areas of concern, Lake Superior is clearly in the best state of recovery, and even considering continuing concern about levels of toxic contaminants in fish and wildlife, ultimate achievement of the objectives seems a reasonable goal. The governments of Canada and the U.S., in fact, have selected Lake Superior for a demonstration program for zero discharge of toxic contaminants as part of their responsibilities under the Great Lakes Water Quality Agreement. All of the other Great Lakes, however, have some significant problems that will impede future recovery. These include: large-scale degradation of tributary and nearshore habitat for fish and wildlife; inadequate reproduction of many native predatory fish; imbalance of aquatic communities associated with population explosions of invading species like sea lamprey, white perch, and zebra mussels; expectations of production from fish communities through stocking and exploitation levels that are not consistent with the productive capacity of the ecosystems; and contaminant levels in fish and wildlife that are sufficient to continue producing effects on health of humans, fish, and fish-eating wildlife.

4.1 Evaluation of Stresses

Chemical pollution of the Great Lakes has decreased. Phosphorus loading targets have been attained for Lake Erie and Lake Ontario, and there is continuing improvement in the regulation of non-point sources of nutrient and sediment loading throughout the Great Lakes basin. Although trends are also encouraging, declining levels of toxic contaminants in fish and wildlife have leveled off (cf. companion paper on the state of toxic contaminants in the Great Lakes). Concern with this continuing contamination led the National Wildlife Federation and the Canadian Institute for Environmental Law and Policy to call for more active efforts of governments to adopt a uniform system of consumption advisories for fish and to move more aggressively to promote a program for zero discharge of toxic contaminants (Anon. 1991c).

The 1987 Protocols to the GLWQA created an initiative for Lakewide Management Plans (LaMPs) to address the need for a more coordinated approach to management of critical pollutants. Management plans for toxic chemicals have been the first focus of these efforts in Lake Ontario and Lake Michigan. These efforts promise continued downward trends in chemical pollution, if progress is made on reduction of atmospheric input, on suppression of the resuspension of contaminated sediments, and on control of input from non-point sources. Future progress in restoration of the ecosystems of the Great Lakes will then depend upon reducing physical and biological stresses.

The physical integrity of the ecosystems in the Great Lakes basin has been degraded by a wide range of historical human activities. The assessment of Ohio's estuarine fish communities (Thoma, unpublished report) is typical of other areas in the Great Lakes. Thoma lists several types of habitat modifications that contribute to degradation: wetland filling, marina construction, shipping channel construction and maintenance, and bank alterations with either rip rap or vertical bulkheads. Throughout the Great Lakes, natural shorelines, wetlands, and tributaries have disappeared or have been altered. Impoundments and siltation have eliminated spawning habitat for adfluvial fish species, and nearshore fish communities and nursery areas for off shore fish species have been seriously impaired. The magnitude of these effects has been well documented for some Areas of Concern (e.g. Ohio EPA, 1992). However, the overall effects of these habitat modifications on the health of open-water fish communities are not readily documented. In Lakes Ontario and Michigan and to lesser extents in Huron and Superior, stocking of top predators obscures the effects of degraded habitat. In Lake Erie, Lake St. Clair, and mesotrophic portions of the other Great Lakes (e.g. Green Bay, Bay of Quinte, and Saginaw Bay) the fish communities may have already compensated for these effects by restructuring and elimination of tributary dependent stocks. A major challenge to aquatic resource managers will be the inventory and classification of this habitat (cf. Busch and Sly 1992) to support planning for preservation and remediation of critical habitat.

Although physical and chemical stresses have contributed to the decline in the integrity of Great Lakes' ecosystems, stresses associated with biological factors have, in fact, caused much more severe degradation, particularly in lake ecosystems. The primary stresses are over-exploitation of biological resources and introduction of exotic organisms. Sustainable exploitation of renewable, natural resources is a challenge to managers. Ludwig *et al.* (1993) argue that technical and social factors combine in such a way that the challenge may never be fully met. Certainly, the history of the Great Lakes offers dramatic examples of the effects of over-fishing and mismanagement. Christie (1972) documents the major role of over-fishing in destabilizing the fish community of Lake Ontario, and similar findings are available for Lake Erie (Nepszy 1977), Lake Michigan (Wells and McLain 1973), Lake Huron (Berst and Spangler 1973), and Lake Superior (Lawrie and Rahrer 1973). The interaction of exploitation and the deliberate and accidental introduction of non-indigenous species has proven to be extremely disruptive. The invasion of sea lamprey into the upper Great Lakes resulted in the demise of lake trout in Lake Michigan and Lake Huron and the loss of a number of lake trout stocks in Lake Superior before an international program for the control of sea lamprey was begun in the 1950's (Smith and Tibbles 1980). The extent of the disruption of the food web by sea lamprey and more recently by zebra mussels and the spiny water flea have led to recommendations for more stringent controls on introductions (IJC and GLFC 1990). Mills *et al.* (1993) document 139 non-indigenous species that have become established since the 1880s. Although few of these species have had the disruptive impact of purple loosestrife, sea lamprey or zebra mussels, they have a cumulative effect on the structure of aquatic communities of the Great Lakes, and their persistence raises substantial problems for the rehabilitation and maintenance of native species associations.

4.2 Management Challenge

Various indicators clearly show that the present state of the health of aquatic communities of the Great Lakes does not satisfy the ecosystem objectives adopted by Canada and the United States. Although some of these indicators show signs of improvement, managers will find an emerging problem in obtaining agreement on quantitative specification of endpoints for the indicators that will specify attainment of ecosystem objectives. The goal of the GLWQA is to restore and maintain the integrity of the ecosystems of the Great Lakes. Until now, there has been an assumption that specification of ecosystem integrity is largely a scientific or technical issue. The extent of historical disruption of aquatic communities and the establishment of large numbers of non-indigenous species, however, may preclude the use of native associations (i.e. pre-settlement ecosystems) as benchmarks for ecosystem integrity.

At best, scientific analysis will allow specification of alternative configurations of the structure of aquatic communities in the Great Lakes that are consistent with fundamental ecological principles. The ultimate selection of a restored state is thus a matter of social preference. Because social preference for state of the Great Lakes ecosystems embodies an implicit set of uses, the specification of quantitative end points for the indicators is embroiled in the determination of acceptable ways of using the resources of the Great Lakes. Ecosystem objectives do not address the issue of how to balance the various uses of these resources, and managers may find future progress toward attaining the goals of the GLWQA impeded by the lack of consensus on the desired state of aquatic ecosystems.

One role of State-of-the-Lakes reporting is to define the condition of the ecosystems of the Great Lakes relative to the desired state and to identify and prioritize management initiatives necessary to improve and/or to maintain it. As such, the State of the Lakes Report is a vital part of a strategic management process. However, management of the Great Lakes is deficient as a strategic planning process. As Naisbitt (1980) stated, strategic planning requires a strategic vision with explicit milestones. As discussed above, the goals and specific objectives in the GLWQA do not serve as a strategic vision nor does it provide milestones. The challenge of ecosystem management in the Great Lakes, therefore, is as much a challenge to institutional structure as to individual management agencies.

5.0 Literature Cited

Anonymous. 1991 a. Toxic Chemicals in the Great Lakes and Associated Effects. Volume I: Contaminant Levels and Trends. Environment Canada, Department of Fisheries and Oceans, and Health and Welfare Canada. 488 p.

Anonymous. 1991 b. Toxic Chemicals in the Great Lakes and Associated Effects. Volume II: Effects. Environment Canada, Department of Fisheries and Oceans, and Health and Welfare Canada. pp 495-755.

Anonymous. 1991 c. A prescription for healthy Great Lakes. Report of the Program for Zero Discharge. National Wildlife Federation and the Canadian Institute for Environmental Law and Policy. 65 p.

Beitinger, T. L. and R. W. McCauley. 1990. Whole-animal physiological processes for the assessment of stress in fishes. J. Great Lakes Res. 16:542-575.

Berst, A. H. and G. R. Spangler. 1973. Lake Huron--The ecology of the fish community and man's effects on it. Great Lakes Fish. Comm., Tech. Rep. 21. 41 pp.

Bertram, P. E. and T. B. Reynoldson. 1992. Developing ecosystem objectives for the Great Lakes: policy, progress and public participation. J. Aquat. Ecosys. Health 1:89-95.

Best, D. A., W. W. Bowerman, T. J. Kubiak, S. R. Winterstein, S. Postupalsky, M. C. Shieldcastle, and J. P. Giesy. (in press). Reproductive impairment of bald eagles along the Great Lakes shorelines of Michigan and Ohio. IV World Conf. Birds of Prey and Owls.

Bowerman, W. W., T. J. Kubiak, J. B. Holt, D. L. Evans, R. G. Eckstein, C. R. Sindelar, D. A. Best, and K. D. Kozie. In press. Observed abnormalities in mandibles of nestling bald eagles (*Haliaeetus leucocephalus*). Bull. Environ. Contam. Toxicol.

Busch, W. N. and P. G. Sly 1992. The Development of an Aquatic Habitat Classification System for Lakes. CRC Press, Ann Arbor. 225 p.

Christie. W. J. 1972. Lake Ontario: effects of exploitation, introductions, and eutrophication on the salmonid community. J. Fish. Res. Bd. Canada 29: 913-929.

Christie, W. J. 1973. A review of the changes in the fish species composition of Lake Ontario. Great Lakes Fish. Comm. Tech. Rep 23, 65p.

Christie, W. J. [Ed.]. 1993. Lake Ontario: an ecosystem in transition. Report of the Lake Ontario Pelagic Community Health Indicator Committee. Ecosystem Objectives Working Group.

Dermott, R. M. In press. Distribution and ecological impact of "Quagga" mussels in the lower Great Lakes. In Proceedings of Third International Zebra Mussel Conference. Electric Power Research Institute Report No. TR-102077.

Dolan, D. M. 1993. Point source loadings of phosphorus to Lake Erie: 1986-1990. J. Great Lakes Res. 19:212-223.

Edwards, C. J. and R. A. Ryder. 1990. Biological surrogates of mesotrophic ecosystem health in the Laurentian Great Lakes. International Joint Commission. Windsor, Ontario. 69 p.

Edwards, C. J., R. A. Ryder, and T. R. Marshall. 1990. Using lake trout as a surrogate of ecosystem health for oligotrophic waters of the Great Lakes. J. Great Lakes Res. 16:591-608.

Evans, D. O., G. J. Warren, and V. W. Cairns. 1990. Assessment and management of fish community health in the Great Lakes: synthesis and recommendations. J. Great Lakes Res. 16:639-669.

Farara, D. G. and A. J. Burt. 1993. Environmental assessment of Western Lake Erie Sediments and Benthic Communities--1991. Report prepared for the Ontario Min. of Env. and Energy, Water Res. Branch, Great Lakes Section by Beak Consultants Ltd., Brampton, Ontario. 193 pp.

Fox, G. A., S. W. Kennedy, R. J. Norstrom, and D. C. Wigfield. 1988. Porphyria in herring gulls: a biochemical response to chemical contamination of Great Lakes food chains. Env. Toxic. Chem. 7:831-839.

Fox, G. A., B. Collins, E. Hayakawa, D. V. Weseloh, J. P. Ludwig, T. J. Kubiak, and T. C. Erdman. 1991. Reproductive outcomes of colonial fish-eating birds: a biomarker for developmental toxicants in Great Lakes food chains. II. Spatial variation in the occurrence and prevalence of bill defects in young double-crested cormorants in the Great Lakes, 1979-1987. J. Great Lakes Res. 17:158-167.

Fox, G. A. 1993. What have biomarkers told us about the effects of contaminants on the health of fish-eating birds in the Great Lakes? The theory and a literature review. J. Great Lakes Res. 19: 722-736.

Gilbertson, M. 1992. PCB and dioxin research and implications for fisheries research and resource management. Can. J. Fish. Aquat. Sci. 49: 1078-1079.

Gilbertson, M. 1993. Show cause: Response to Munkittrick (1993). Can. J. Fish. Aquat. Sci. 50: 1570-1573.

Gilbertson, M., G. A. Fox, M. Henry, and J. P. Ludwig. 1990. Commentary: New strategies for Great Lakes toxicology. J. Great Lakes Res. 16: 625-627.

Gilbertson, M., T. Kubiak, J. Ludwig, and G. Fox. 1991. Great lakes embryo mortality, edema, and deformities syndrome (GLEMEDS) in colonial fish-eating birds: similarity to chick-edema disease. J. Toxic. Env. Health, 33:455-520.

Goodier, J. L. 1981. Native lake trout (*Salvelinus namaycush*) stocks in Canadian waters of Lake Superior prior to 1955. Can. J. Fish. Aquat. Sci. 38: 1724-1737.

Hatch, R. W., S. J. Nepszy, K. M. Muth, and C. T. Baker. 1987. Dynamics of the recovery of the Western Lake Erie walleye (*Stizostedion vitreum vitreum*) stock. Can. J. Fish. Aquat. Sci. 44: 15-22.

Hartman, W. L. 1973. Effects of exploitation, environmental changes, and new species on the fish habitats and resources of Lake Erie. Great Lakes Fish. Comm. Tech. Rep. 22, 43p.

Hedtke, S., A. Pilli, D. Dolan, G. McRae, B. Goodno, R. Kreis, G. Warren, D. Swackhamer, and M. Henry. 1992. EMAP-Great Lakes monitoring and research strategy. Environmental Research Lab., U.S. Environmental Protection Agency. Duluth, Minn. June, 1992.

Hnath, J. G. [ed]. 1993. Great Lakes fish disease control policy and model program. Great Lakes Fish. Comm. Spec. Publ. 93-1: 1-38.

Holling, C. S. 1992. Cross-scale morphology, geometry, and dynamics of ecosystems. Ecol. Mon. 62: 447-502.

Horner, R. W., and R. L. Eshenroder [eds.]. 1993. Protocol to minimize the risk of introducing emergency disease agents with importation of salmonid fishes from enzootic areas. Great Lakes Fish. Comm. Spec. Publ. 93-1: 39-54.

International Joint Commission and Great Lakes Fishery Commission. 1990. Exotic species and the shipping industry: the Great Lakes-St. Lawrence ecosystem at risk. A special report to the Governments of the United States and Canada. 74 p.

Johannsson, O. E., and R. O'Gorman. 1991. Roles of predation, food, and temperature in structuring the eplimnetic zooplankton populations in Lake Ontario, 1981-1986. Trans. Am. Fish. Soc. 120: 193-208.

Jones, M. L., J. F. Koonce, and R. O'Gorman. 1993. Sustainability of hatchery-dependent fisheries in Lake Ontario: the conflict between predator demand and prey supply. Trans. Amer. Fish. Soc. 122: 1002-1018.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. Fisheries 6:21-27.

Kerr, S. R. and L. M. Dickie. 1984. Measuring the health of aquatic ecosystem. In Cairns, V. W., P. V. Hodson, and J. O. Nriagu [Eds.]. Contaminant Effects on Fisheries. John Wiley and Sons. New York. pp: 279-284.

Koonce, J. F. 1990. Commentary on fish community health: monitoring and assessment in large lakes. J. Great Lakes Res. 16:631-63.

Lawrie, A. H. and J. F. Rahrer. 1973. Lake Superior--A case study of the lake and its fisheries. Great Lakes Fish. Comm., Tech Rep. 19. 69 pp.

Leach, J. H. 1993. Population dynamics and ecological impacts of zebra mussels in Western Lake Erie. Abstract. Third International Zebra Mussel Conference. Toronto, Ontario, February 23-26, 1993.

Loftus, K. H. and H. A. Regier [eds]. 1972. Proceedings of the 1971 symposium on salmonid communities in oligotrophic lakes. J. Fish. Res. Bd. Canada 29:611-986.

Ludwig, D., R. Hilborn, and C. J. Walters. 1993. Uncertainty, resource exploitation and conservation: Lesions from history. Science 260(2):17, 36.

Mac, M. J. 1988. Toxic substances and survival of Lake Michigan salmonids: field and laboratory approaches. In: Evans, M. S. [ed.] Toxic contaminants and ecosystem health: a Great Lakes Focus. John Wiley and Sons, Inc. New York. pp: 389-401.

Makarewicz, J. C. 1993a. Phytoplankton biomass and species composition in Lake Erie, 1970 to 1987. J. Great Lakes Res. 19:258-274.
Makarewicz, J. C. 1993b. A lakewide comparison of zooplankton biomass and its species composition in Lake Erie, 1983-87. J. Great Lakes Res. 19:275-290.
Marshall, T. R., R. A. Ryder, C. J. Edwards, and G. R. Spangler. 1987. Using the lake trout as an indicator of ecosystem health--application of the Dichotomous Key. Great Lakes Fish. Comm. Tech. Rep. 49. 35 p.
Marshall, T. R., R. A. Ryder, and C. J. Edwards. 1992. Assessing ecosystem quality through a biological indicator: a temporal analysis of Lake Superior using the Lake Trout Dichotomous Key. In Vander Wal, J. and P. D. Watts [Eds.]. Making a Great Lake Superior. Lakehead University. Centre for Northern Studies, Occasional Paper #9. pp: 41-65.
Mills, E. L., J. H. Leach, J. T. Carlton, and C. L. Secor. 1993. Exotic species in the Great Lakes: a history of biotic crises and anthropogenic introductions. J. Great Lakes Res. 19: 1-54.
Minns, C. K. (in press). Approaches to assessing and managing cumulative ecosystem change, with the Bay of Quinte as a case study. Submitted to J. Aquat. Ecosystem Health.
Minns, C. K., V. W. Cairns, R. G. Randall, and J. E. Moore. in press. An index of biotic integrity (IBI) for fish assemblages in the littoral zone of Great Lakes' areas of concern. Can. J. Fish. Aquat. Sci. (in press).
Mineau, P., G. A. Fox, R. J. Norstrom, D. V. Weseloh, D. J. Hallet, and J. A. Ellenton. 1984. Using the herring gull to monitor levels and effects of organochlorine contamination in the Canadian Great Lakes. In: Nriagu, J. O. and M. S. Simmons, [eds.]. Toxic contaminants in the Great Lakes. J. Wiley and Sons, New York. pp 426-452.
Munkittrick, K. R. 1993. Ecoepidemiology: Cause and effect or just be-cause. Can. J. Fish. Aquat. Sci., 50: 1568-1570.
Nepszy, S., J. Budd, and A. O. Dechtiar. 1978. Mortality of young-of-the-year rainbow smelt (*Osmerus mordax*) in Lake Erie associated with the occurrence of *Glugea hertwigii*. J. Wild. Diseases 14:233-239.
Naisbitt, John. 1980. Megatrends: Ten New Directions Transforming Our Lives. Warner Books, New York.
Ohio EPA. 1992. Biological community status of the lower Ashtabula River and harbor with the Area of Concern (AoC). Ohio EPA Tech. Rep. EAS/1992-6-2. 22 p. + tables.
Powers, Melanie S. 1989. An Examination of the Dichotomous Key and the Reliability of its Ecosystem Indicator Measure of System Health, the Lake Trout (*Salvelinus namaycush*), for Lakes Ontario and Superior. M.S. Thesis, Case Western Reserve University. Cleveland, Ohio. 137 p.
Price, I. and D. V. Weseloh. 1986. Increased numbers and productivity of Double-crested Cormorants, *Phalacrocorax auritus*, on Lake Ontario. Can. Field-Nat. 100: 474-482.
Rankin, E. T. 1989. The qualitative habitat evaluation index [QHEI]: rationale, methods, and application. Ohio EPA. Division of Water Quality Plan. and Assess. 72 p.
Rapport, D. J. 1990. Challenges in the detection and diagnosis of pathological change in aquatic ecosystems. J. Great Lakes Res. 16:609-618.
Ryder, R. A. 1990. Ecosystem health, a human perception: definition, detection, and the dichotomous key. J. Great Lakes Res. 16:619-624.
Ryder, R. A. and C. J. Edwards [Eds.]. 1985. A conceptual approach for the application of biological indicators of ecosystem quality in the Great Lakes basin. Report to Great Lakes Science Advisory Board. International Joint Commission. Windsor, Ontario. 169 p.
Ryder, R. A. and S. R. Kerr. 1990. Harmonic communities in aquatic ecosystems: a management perspective. In Van Densen, L. T., B. Teinmetz, and R. H. Hughes [Eds.]. Management of Freshwater Fisheries. Proc. Sym. EIFAC, Göteborg, Sweden. Pudoc, Wageningen. pp: 594-623.
Scharf, W. C. And G. W. Shugart. 1981. Recent increases in Double-crested Cormorants in the United States Great Lakes. American Birds 35: 910-911.
Schneider, C. P., D. P. Kolenosky, and D. B. Goldthwaite. 1983. A joint plan for the rehabilitation of lake trout in Lake Ontario. Great Lakes Fish Comm. Spec. Publ. 50 pp.
Sheldon, R.W., A. Prakash, and W.H. Sutcliffe, Jr. 1972. The size distribution of particles in the ocean. Limnology and Oceanography 17: 327-340.

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

Sprules, W. G., S. B. Brandt, D. J. Steward, M. Munawar, E. H. Jin, and J. Love. 1991. Biomass size spectrum the Lake Michigan pelagic food web. Can. J. Fish. Aquat. Sci. 48:105-115.

Steedman, R. J. 1988. Modification and assessment of an index of biotic integrity to quantify stream quality in southern Ontario. Can. J. Fish. Aquat. Sci. 44(Suppl. 2):95-103.

Steele, J. H. 1985. A comparison of terrestrial and marine systems. Nature 313: 355-358.

Stewart, D. J. and M. Ibarra. 1991. Predation and production by salmonine fishes in Lake Michigan. Can. J. Fish. Aquat. Sci. 48:909-922.

Thoma, R. F. Manuscript. A preliminary assessment of Ohio's Lake Erie estuarine fish communities. Unpublished report, Ohio EPA, Division of Water Quality Planning and Assessment. 27p.

Wells, L. and A. L. McLain. 1973. Lake Michigan: Man's effects on native fish stocks and other biota. Great Lakes Fishery Commission, Tech. Rep. 20. 55 p.

Weseloh, D. V., P. Mineau, and J. Struger. 1990. Geographic distribution of contaminants and productivity measures of herring gulls in the Great Lakes: Lake Erie and connecting channels. Sci. Total Environ. 91:141-159.

Weseloh, D. V. C., P. J. Ewins, J. Struger, P. Mineau, C. A. Bishop, S. Postupalsky, and J. P. Ludwig. In press. Double-crested Cormorants (*Phalacrocorax auritus*) of the Great Lakes: Changes in population size, breeding distribution and reproductive output, 1913-1991. Colonial Waterbirds.

Wiemeyer, S. N., T. G. Lamont, C. M. Bunck, C. R. Sindelar, F. J. Gramlich, J. D. Fraser, and M. A. Byrd. 1984. Organochlorine pesticide, polychlorobiphenyl and Mercury residues in bald eagle eggs, 1969-79, and their relationship to shell thinning and reproduction. Arch. Environ. Contam. Toxicol. 13:529-549.

World Commission on Economic Development. 1987. Our Common Future. Report of the Brundland Commission on Economy and Environment. Oxford Univ. Press. Oxford.

Yoder, C. O. 1991. The integrated biosurvey as a tool for evaluation of aquatic life use attainment and impairment in Ohio surface waters. Proceedings of a Symposium on Biological Criteria: Research and Regulation. U.S. EPA. pp: 110-122.

List of Figures

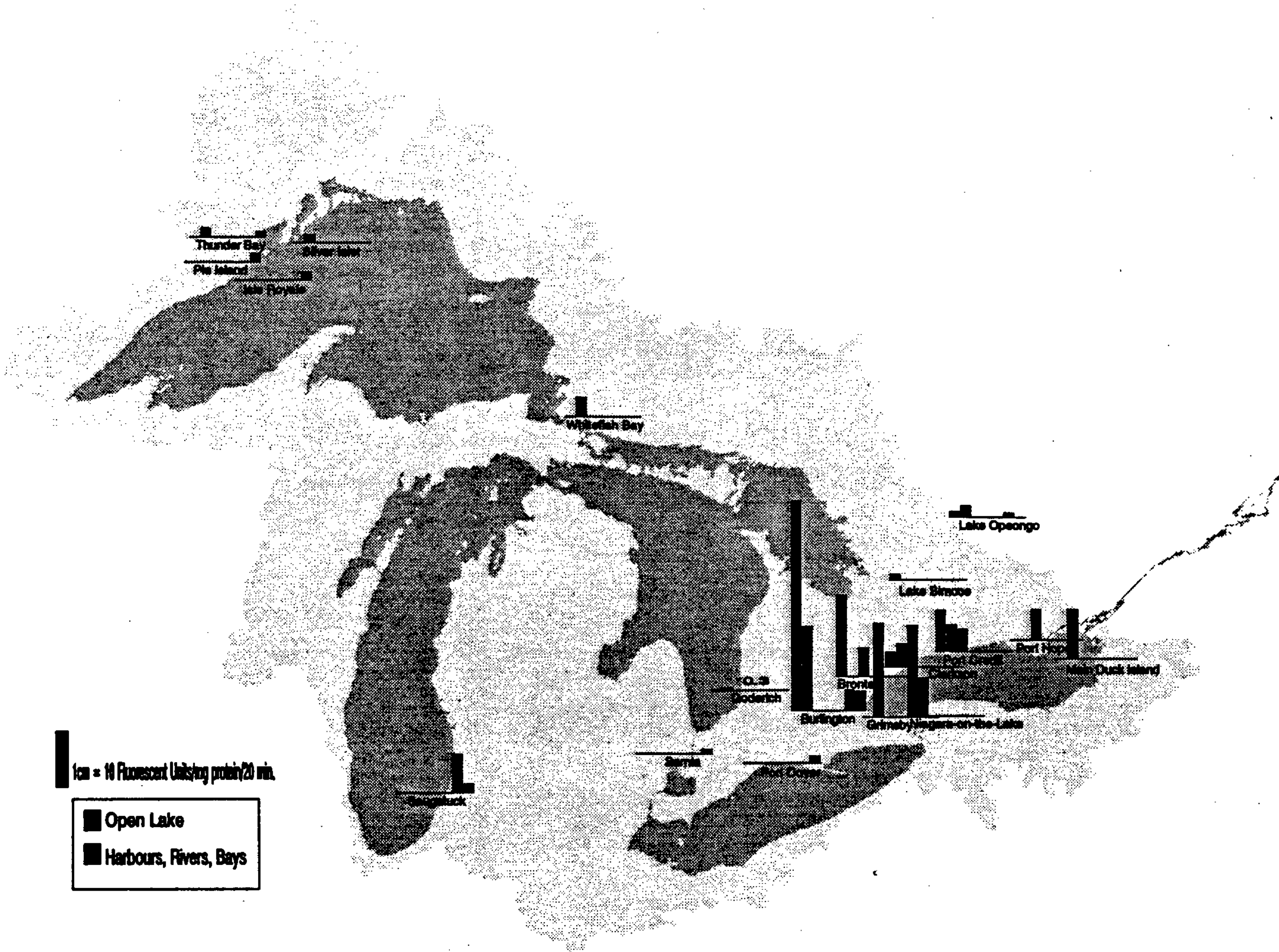

Figure 1. Patterns in observations of mixed function oxidase (MFO) activity in lake trout of the Great Lakes basin

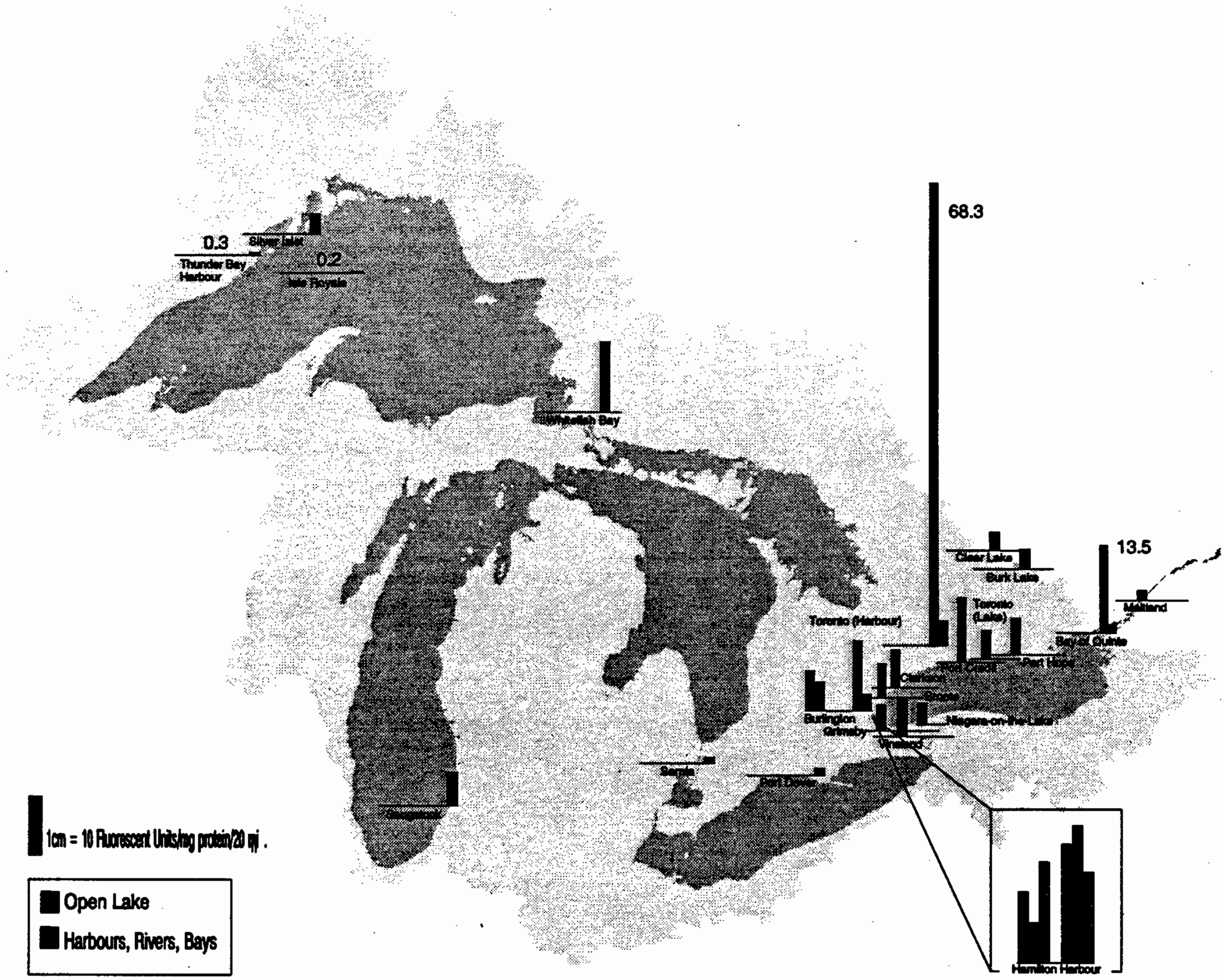

Figure 2. Patterns in observations of mixed function oxidase (MFO) activity in white sucker of the Great Lakes basin

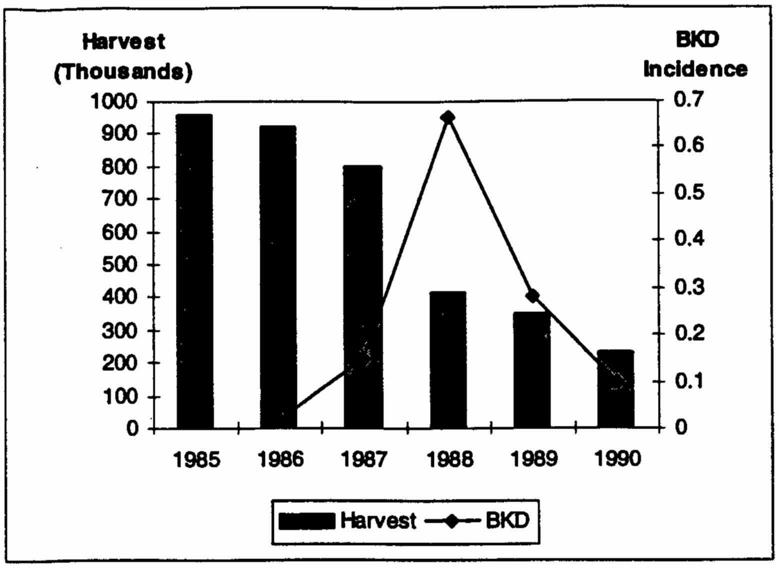

Figure 3. Harvest and BKD incidence trends for Chinook salmon in Lake Michigan

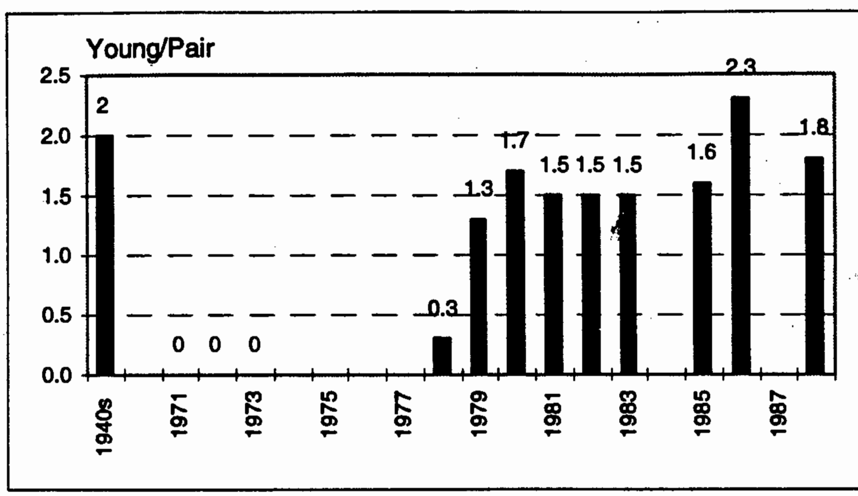

Figure 4. Trends in productivity of double-crestested cormorants in Lake Ontario

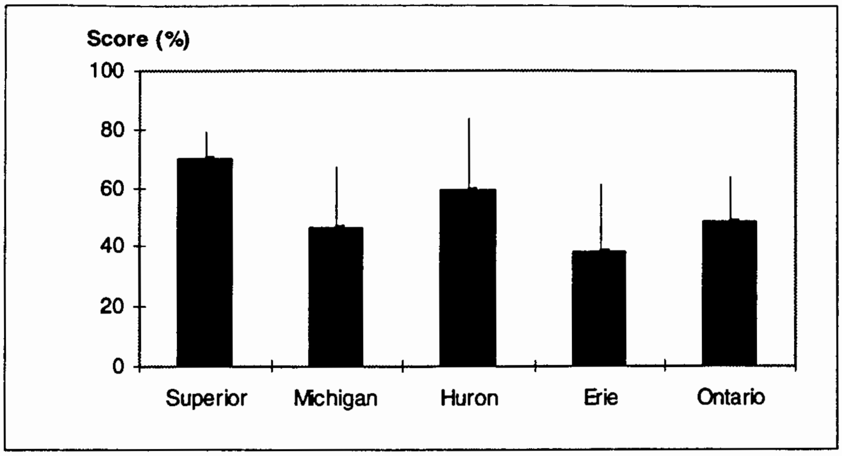

Figure 5. Scores for each Great Lake for the interval 1982-1985 from the Dichotomous Key

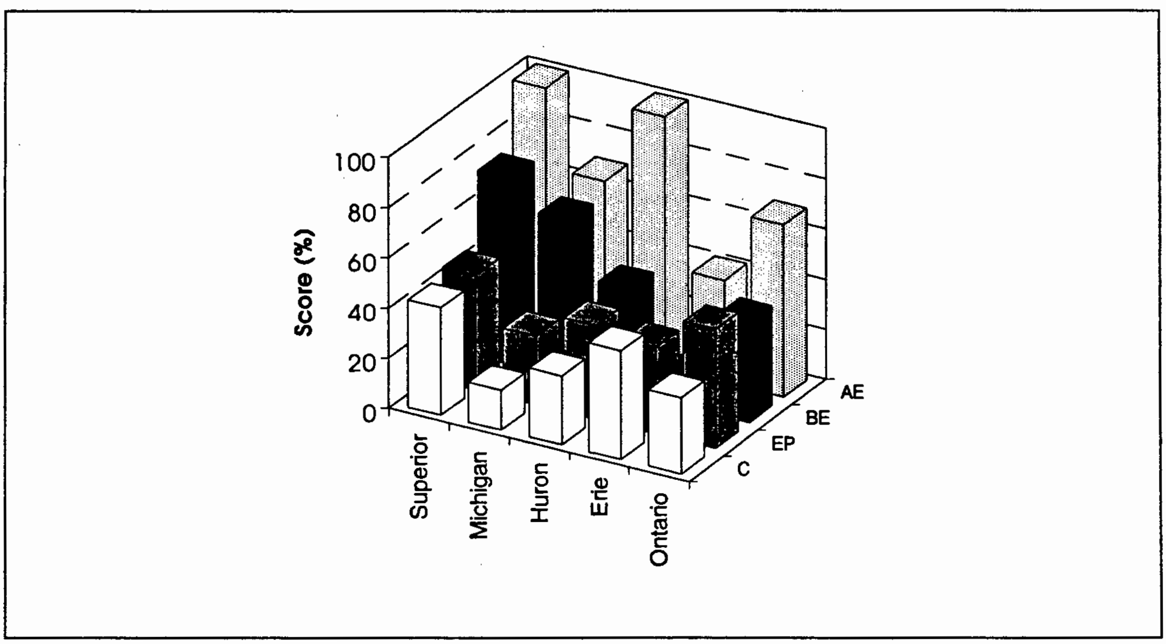

Figure 6. Contribution by stress category to Dichotomous Key scores for each Great Lake for the interval 1982-1985

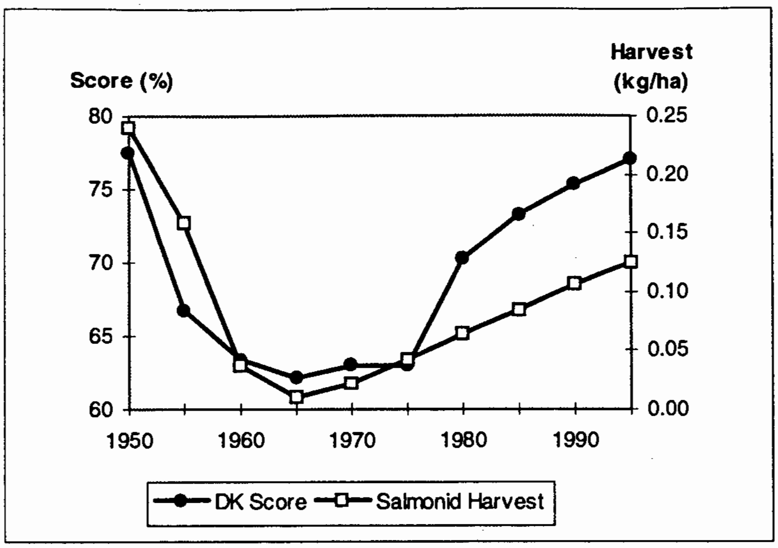

Figure 7. Comparison of annual harvest of all salmonines in Lake Superior with the score from the Dichotomous Key. Data

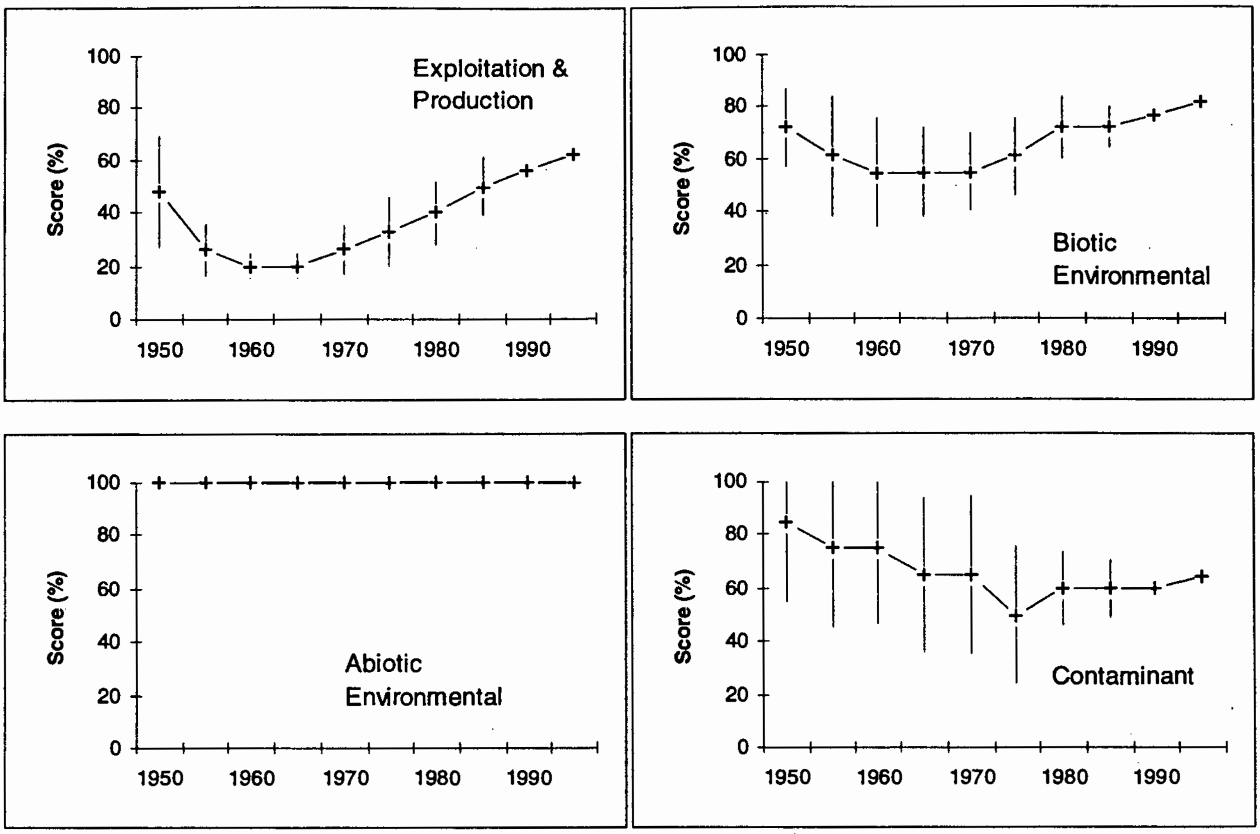

Figure 8. Contribution by stress category to Dichotomous Key scores for trends in Lake Superior

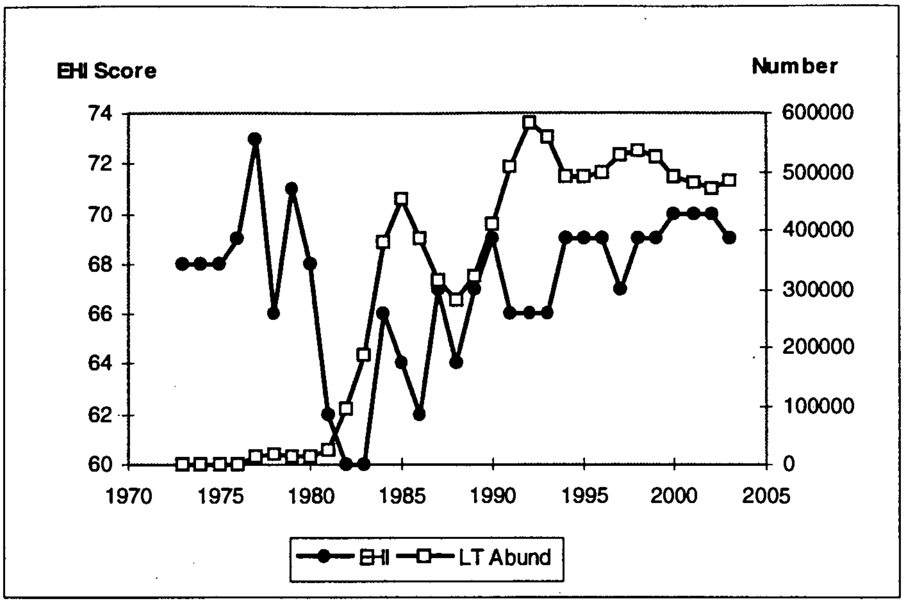

Figure 9. Estimated ecosystem health index for Lake Ontario in the period 1973 to 2002

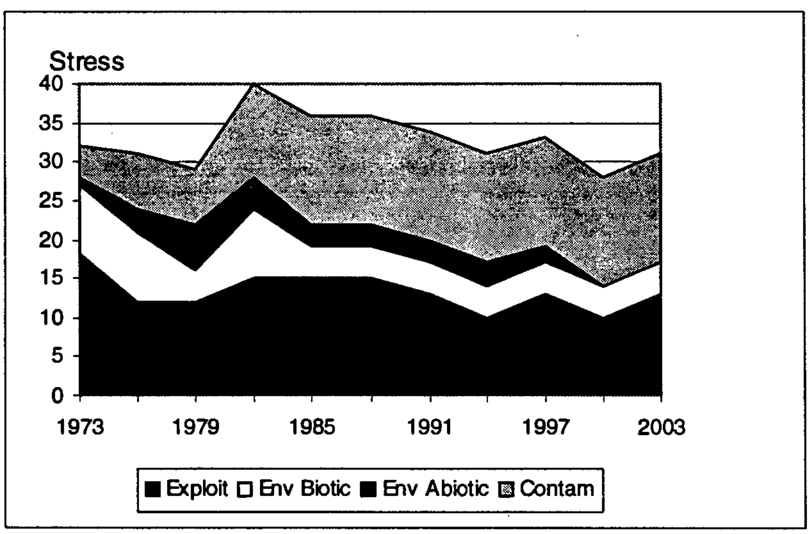

Figure 10. Contribution of various stress categories to the degradation of ecosystem health in Lake Ontario

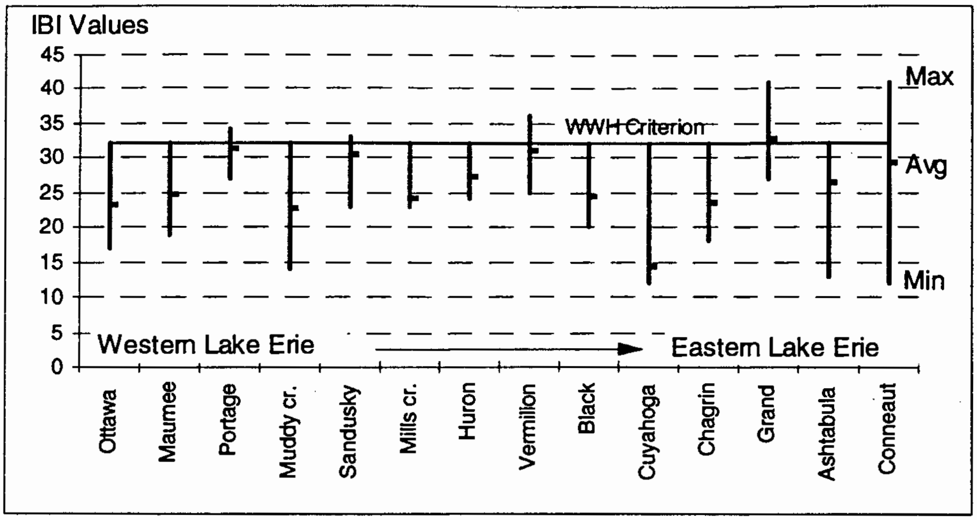

Figure 11 Minimum, maximum, and mean Index of Biotic Integrity (IBI) for 14 estuaries. (For comparison the Warm Water Habitat aquatic life use criterion value of 32 is plotted as a solid line)