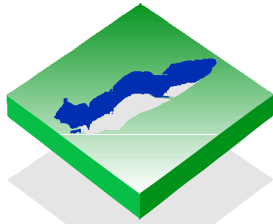LAKE ERIE

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Impairment Assessment of Beneficial Uses: Degradation of Phytoplankton and Zooplankton Populations

Ora Johannsson and Scott Millard
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Degradation of Phytoplankton and Zooplankton Populations

Prepared for the Lake Erie LaMP

Preliminary Beneficial Use Impairment Assessment

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NOTE TO THE READER:

This technical report was prepared as one component of Stage 1, or "Problem Definition," for the Lake Erie LaMP. This report provides detailed technical and background information that provides the basis for the impairment conclusions recorded in the Lake Erie LaMP *Status Report*.

This document has been extensively reviewed by the government agencies that are partnering to produce the LaMP, outside experts, and the Lake Erie LaMP Public Forum, a group of approximately of 80 citizen volunteers.   This review was designed to answer two questions:

- Is the document technically sound and defensible?
- Do the reviewers agree with the document conclusions and format?

In its present form, this report has been revised to address the comments received during that review process, and there is majority agreement with the impairment conclusions presented.

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13.1 Listing Guidelines

According to the Listing Guidelines, plankton are impaired when phytoplankton or zooplankton community structure significantly diverges from unimpacted control sites of comparable physical and chemical characteristics. In addition, this use will be considered impaired when relevant, field-validated, phytoplankton or zooplankton bioassays (e.g. *Ceriodaphnia*; algal fractionation bioassays) with appropriate quality assurance/quality controls confirm toxicity in ambient waters.

13.2 Introduction

Ecologists have grappled with the concepts of biological integrity, ecosystem health, and biodiversity in trying to define the normal condition of ecosystems. These concepts encompass different components of the ecosystem, with biological integrity being the most inclusive. The following definitions come from Callicott (1995).

Biological integrity is 'the capability ' of the ecosystem' of supporting and maintaining a balanced, integrated, adaptive community of organisms having a species composition, diversity and functional organization comparable to that of natural habitat in the region' (Karr and Dudley 1981). If a system has biological integrity it will also be healthy (Karr 1995, Callicott 1995), but the reverse is not true (Callicott 1995).

 Ecosystem health is defined more on the basis of function.   Ecosystems may be said to be *healthy* when 'linked processes and functions occur normally (that is as they have occurred historically) or change normally (that is at the rates that they have changed historically)' (Callicott 1995). According to Callicott, linked ecological processes include the capture of solar energy through photosynthesis and dissipation as it passes up trophic pyramids, biomass production accumulation and decomposition, the exchange of carbon, nitrogen and oxygen between biota and the environment etc. These concepts examine the movement of matter and energy through the trophic food-web thereby incorporating both system and community level properties. The ideas presented in Callicott have clarified how we might assess the functional health of ecosystems.

Biodiversity 'means the variety of components (or elements) at every level of biotic community organization'.

According to the Great Lakes Water Quality Agreement (IJC 1987), Beneficial Use Impairments (BUA) result from the loss of chemical, physical or biological integrity. This means that we must look at both characteristics of plankton community composition and the functional relationships between the plankton community and other trophic levels in evaluating the plankton BUA.

Plankton are small organisms, both plants (phyto) and animals (zoo), which live in the water column or attached to substrates in aquatic and marine environments. They possess limited or no ability to swim against currents, but move with the water. Phytoplankton form the base of the pelagic food web in lakes. Much of the energy captured by phytoplankton is consumed by zooplankton which in turn are eaten by larger organisms such as larger zooplankton, benthos and fish (Appendix I).

In keeping with the definitions of ecosystem health and biological integrity, we understand the beneficial use of plankton communities to be the conversion of solar energy to chemical energy (biomass), the incorporation of nutrients into biomass and the conveyance of these materials to normal, diverse fish and wildlife communities and ultimately to human populations by a plankton community that is balanced and adaptive to change.   We have defined impairment of this beneficial use as a decrease in the ability of these communities to perform these functions as a result of stresses within the ecosystem caused by anthropogenic activities. Anthropogenic stresses on plankton populations can result from the addition of nutrients and toxicants to aquatic environments, fish harvesting and stocking practices, introduction of exotic species, and habitat alterations which could include changes in ultraviolet light conditions and increased temperature associated with global climate warming.

13.2.1 How Do We Judge Impairment?

The Listing/Delisting Criteria accepted by the International Joint Commission (IJC) (1989), state that plankton are impaired 'when phytoplankton or zooplankton community structure significantly diverges from unimpacted control sites of comparable physical and chemical characteristics.' Unimpacted sites are considered to be the best measure of properly functioning ecosystems and deviation from those conditions is a measure of dysfunction.   Finding such unimpacted sites is difficult. The Remedial Action Plan (RAP) assessments assumed that the open-lake plankton were unimpacted for their comparisons. However, we know that plankton populations in the open lake have been

subjected to many of the same stresses as the more heavily impacted Areas of Concern although impacts have often been diluted in the open lake. Other physically and chemically comparable sites, such as Lake Ontario, have also been heavily impacted by human intervention.

Historical information and paleolimnological studies of historical plankton communities have the potential to provide surrogates of unimpacted or control communities but these comparisons are problematic. With the exception of diatoms, plankton communities are not well preserved in the sediments. Open-lake historical plankton data for comparison to current data are not available for periods before anthropogenic impacts occurred in the lake. In addition, community structure has been permanently changed with the addition of exotics, making absolute comparisons of species composition/relative abundance with historical information invalid. It should be noted, however, that the impairment of plankton communities by exotic species occurs through the disruption or alteration of ecosystem function in terms of energy transfer and is not based on their presence alone.

Can we use lakes of origin to assess possible impairments caused by exotics which strongly alter community composition and function? That would require comparing ecological function in the original lake(s) with that in the invaded lake and would assume that ecological function in the original lake(s) was not impaired. It has been suggested that we could use European lakes containing dreissenids (zebra and quagga mussels) as a benchmark for assessing impairment in Lake Erie now that dreissenids have invaded. Although dreissenids came from Europe, their relative importance and distribution in European lakes differs from that in Lake Erie. Dreissenids are not found at depth in European lakes but are most abundant in the 2 m - 4 m range. In Lake Erie, they cover much of the soft sediments at all depths as well as the shallows. Their impact will be quite different. European lakes are also unlikely to provide reference conditions because they have been subjected to anthropogenic activities for centuries. European lakes can not provide the necessary reference conditions for assessing impairments in the Lake Erie.

It has been suggested that we cannot define impairment until a consensus has been reached on a set of 'Ecosystem Goals and Objectives' for Lake Erie. This approach defines ecosystem health in terms of human priorities rather than ecological principles. We agree with Minns (1995) who argues that ecological, social and economic concerns are inter-related in a pyramid-like manner where a

healthy, intact ecosystem is the foundation needed to build a sound society and sustainable economy. Consequently, impairment should not be dependent on use/harvest goals and objectives which are farther up the pyramid, but on the functionality of the ecosystem or base of the pyramid.

The productive capacity and structural features of the Lake Erie ecosystem will place limits on ecosystem objectives. Objectives must fit within realistic, ecosystem-based boundaries. For example, in response to the effects of eutrophication, phosphorus loadings to the Great Lakes have been controlled and reduced as a result of the Great Lakes Water Quality Agreement. This action has restored the lake to a condition of lower ecosystem productivity from levels observed in the late 1960's and early 1970's (Millard et al. 1996). Ecosystem goals and objectives should fall within the productivity constraints of the lower nutrient conditions, conditions that existed prior to the period of most intense human influence.

How then do we know when the plankton are impaired if we are not to use man-made goals and objectives and do not have sufficient historical data or directly comparable unimpacted sites? According to Karr and Dudley (1981) and Munawar et al. (1992), impairment should be assessed using a suite of indicators which considers both the functional efficiency and structural integrity of the system. Measures of functional efficiency might include rates of photosynthesis, grazing, or production compared with some potential rate, or ratio of production between trophic levels. Measures of structural integrity might include evaluations of species and size composition with respect to other systems, species richness, species eveness, indicator species, and edibility. This approach along with the conceptual model of Minns (1995) provides sound general ecological guidelines for judging impairment.

13.2.2 Change versus Impairment

When does change constitute impairment? How can normal annual variability in response to natural factors such as climate be differentiated from changes caused by anthropogenic activities? These questions need to be answered in order to establish whether the structure or function of plankton communities is impaired. Assessing impairment is difficult when comparative data at unimpacted sites are not available and when historical data have been collected sporadically and with inconsistent methodology. Long-term biomonitoring programs are most likely to provide the requisite tools, such

as predictive relationships and measures of inter-annual variability, with which to assess impairment. These relationships should be developed for large-scale properties of plankton communities, such as production, biomass, size structure and functional integration. They should incorporate data collected across ecosystems and gradients in lake morphometry, productivity, predation and climate to determine an expected level of ecosystem performance against which to judge change. Development of these tools has begun but further development and testing are required. Presently, subjective evaluations are made in assessing trends in historical data, in assessing the importance of changes in community structure and in applying predictive relationships. Although more critical methods may be developed in the future, we have to live with these uncertainties in the present report.

13.3 Status of Phytoplankton and Zooplankton Populations in Areas of Concern on Lake Erie

Detroit River: not impaired (1996 Detroit River Remedial Action Plan Report submitted to International Joint Commission (IJC))

Raisin River: not impaired (Stage 1 Remedial Action Plan Report to the IJC, Date)

Maumee River: impaired (Stage 1 Remedial Action Plan (RAP) Report to the IJC, Date)

Black River: 'unknown due to lack of data' (Stage 1 RAP Report to the IJC, April 1994)

A 1993-1994 study of periphytic diatoms in this river system revealed stressed algal communities with high abundances of pollution tolerant species with variable diversity (Sgro and Johansen, 1995). The algal community structure of these systems was evaluated and characterized on the basis of relative abundance, global autecologies and diversity. The authors concluded that the periphyton communities in this river estuary were impaired.

Cuyahoga River: 'unknown due to lack of data' (Stage 1 RAP Report to the IJC, DATE)

A 1993-1994 study of periphytic diatoms in this river system revealed stressed algal communities with high abundances of pollution tolerant species with variable diversity (Sgro and Johansen, 1995). The algal community structure of these systems was evaluated and

characterized on the basis of relative abundance, global autecologies and diversity. The author concluded that the periphyton communities in this river estuary were impaired.

Ashtabula River: 'unknown due to lack of data' (Stage 1 RAP Report to the IJC, DATE)

A 1993-1994 study of the periphytic diatoms of this river system indicated that the Ashtabula system supported an algal community representative of a cleaner water flora than that of the Black, Cuyahoga or Old Woman systems (Sgro and Johansen, 1995). The algal community structure of these systems was evaluated and characterized on the basis of relative abundance, global autecologies and diversity.

Presque Isle Bay: unknown (Stage 1 RAP Report to the IJC, 1992)

A study conducted in October 1992 and May 1994 found that 'the composition of the phyto- and zoo-plankton communities and the densities observed were not judged to be unusual based upon comparisons to recent studies at other locations (both inshore and offshore) in Lake Erie, except for the apparent low abundance of Calanoid copepods (Presque Isle Bay RAP Sediment Study 1994).

Buffalo River: not impaired (Stage 1 RAP Report to the IJC, DATE)

Conclusions were based on a 1979 study which found similar communities of zooplankton in Lake Erie, Buffalo Harbour and Buffalo River. The assessment considered abundance and species composition relative to other areas of similar trophy. Chlorophyll a levels were moderate and phytoplankton growth was thought to be light limited due to excessive silt and turbidity. This may be an impairment although it was not considered as such.

Wheatley Harbour: not a concern (Dr. D. Hubert, OMEE, London, Ont., pers. comm.)

The water flushes in and out of Wheatley Harbor so rapidly that there are no associated zooplankton and phytoplankton populations. Environmental conditions are not unnaturally stressful: oxygen concentrations in the harbour are no longer a concern and sediment resuspension is a natural phenomenon.

3.4 Assessment of Beneficial Use Impairment of Phytoplankton

13.4.1 Introduction

Phytoplankton communities in Lake Erie have been subjected to changes in many factors over the last two decades that can alter species composition and biomass such as reduced phosphorus (P) loading, changes in food-chain structure (e.g. the resurgence of walleye and decline in yellow perch), the invasion of exotic species (zebra mussels), fluctuations in water levels, and climatic variability

When do changes in the phytoplankton community constitute impairment? Answering this question is a difficult challenge for the Lake Erie LaMP. Comparison of current to historic phytoplankton communities within a lake, as well as a comparison to communities in other lakes where the status is assumed to be unimpaired, may be the best approaches. Historic comparisons within the lake must use a realistic time frame when setting management goals. For example, although lower productivity of both phytoplankton in the west basin and *Cladophora* on a lakewide basis have been continuing goals of P management, a return to pre-settlement, pristine, conditions was not a goal.

Answering the following questions can help determine whether phytoplankton communities in Lake Erie are impaired. For example, does reduced primary productive capacity of the lake resulting from P control constitute impairment if fisheries productivity is reduced? Have there been significant changes in total phytoplankton biomass or key species and have these changes impacted other trophic levels? Is phytoplankton production currently lower than the potential set by P loading? Have zebra mussels significantly altered the structure or function of the Lake Erie phytoplankton community? We can provide an answer to some of these questions but the necessary inter-trophic level relationships and detailed ecosystem model that are necessary to fully assess the impact of changes in structure and function of the phytoplankton community on higher trophic levels are lacking. Moreover, building models that can attribute cause and effect between phytoplankton and fish is an arduous task because confounding factors increase drastically as connections between non-adjacent trophic levels are sought. The assessment of

impairment should focus on trophic efficiency (qualitative) rather than the absolute level of production (quantitative).

Phytoplankton response to toxicity bioassays (Munawar & Legner 1993) has been designated as a listing/delisting criteria to confirm toxicity of waters to phytoplankton populations. In our assessment of impairment we have chosen other strategies to assess impairment that should ultimately encompass the response to contaminants. Munawar et al. (1992) and Munawar and Legner (1993) discuss a variety of tests that can be used to assess the health of phytoplankton communities. Photosynthetic bioassays that test the short-term (hours) response to metal additions are just one of many techniques discussed in these reviews which also includes the assessment of species composition and biomass.   These tests provide limited information on cumulative damage to populations prior to experiments and fail to confirm toxicity of ambient waters, revealing only the susceptibility to contaminants such as metals that are added in the bioassay. Consequently, it is difficult to draw conclusions regarding the potential for longer-term impacts.

In general, concentrations of organochlorine contaminants in the open water of Lake Erie do not exceed regulations for the protection of fish and wildlife (except dieldrin and possibly PCBs) on a lakewide basis (S. L'Italien, Environment Canada, Burlington, personal communication). Consequently, acute toxicity of contaminants to phytoplankton is probably restricted geographically and confined to "end of pipe" situations.

Phytoplankton can respond to contaminants and other stressors more rapidly than higher trophic levels because of shorter generation times. As a result, a significant contaminant problem resulting in sub-optimal growth for any species should lead to less sensitive species that can grow optimally under that regime of contaminant exposure. Consequently, our assessment of impairment will focus on species composition and biomass as well as photosynthetic carbon input integrated over a longer time frame (seasonally). A significant contaminant impact on phytoplankton should cause low chlorophyll:phosphorus ratios or shifts in species composition. Schindler (1987) points out that in long-term studies at the Experimental Lakes Area, changes in

phytoplankton species composition has been one of the most sensitive indicators of ecosystem stress.

Our assessment of impairment is neither an exhaustive literature review nor a complete analysis of phytoplankton growth ecology in Lake Erie. We have focused on key studies of structure and function of phytoplankton communities. We will assess impairment against indicators of both structural integrity and functional efficiency in healthy ecosystems such as species composition, as well as seasonal photosynthesis and mean biomass compared to potentials set by phosphorus levels.

Literature cited and other studies relevant to phytoplankton in Lake Erie have been listed at the end. There are key periods in the recent history of the lake, each with milestone studies such as Verduin (1964) and Munawar and Munawar (1976) in the pre-P control period of the 1960's and 1970 respectively, and Makarewicz (1993a) in the post-P control but pre-zebra mussel invasion era. In recent years following zebra mussel invasion, the studies of Nicholls and Hopkins (1993), Graham et al. (1996) and Dahl et al. (1995) show evidence of the combined effects of P control and zebra mussel invasion .

There is no single study that can provide a lakewide, seasonal and long-term assessment of phytoplankton communities in Lake Erie. In general, long-term changes are probably best assessed by the water intake study of Nicholls and Hopkins (1993) although data from this study are more spatially limited than those of Munawar and Munawar (1976) and Makarewicz (1993a) which combine both spatial and temporal intensity in sampling. Comparisons with Nicholls and Hopkins are also complicated because these authors report phytoplankton biomass in areal standard units (A.S.U.) rather than cell volume. Seasonal trends are best revealed by the studies of Gladish and Munawar (1980), (1996) and Dahl et al. (1995) because of the high frequency in sampling although these studies were restricted spatially and in their duration (two years).

Publication of studies showing the impact of zebra mussels on lower trophic levels in Saginaw Bay, Lake Huron (Nalepa & Fahnenstiel 1995) and ongoing studies in the eutrophic Bay

of Quinte (OMEE 1997 and 1998 in prep.) where extensive, pre-invasion, data have been collected, will prove invaluable in assessing impacts in Lake Erie.

13.4.2 Changes in Total Phytoplankton Biomass

Significant declines in phytoplankton biomass in the Western Basin have been noted since the implementation of P control (Tables 13.1-13.3). Reductions have been less noticeable in the Central and East basins; however, studies in these areas have also been fewer. Nicholls et al. (1983) showed a localized effect at the mouth of the Grand River in the east basin on phytoplankton biomass and suggested that further reductions in loading at the time (1979) would reduce biomass and improve water clarity. Nicholls and Hopkins (1993) point out interesting lakewide trends in phytoplankton biomass using a long-term data set from water-intake monitoring sites. They noted considerable declines in phytoplankton biomass in the west basin up to the mid-1980's commensurate with reduced P loading but similar declines were not evident in the rest of the lake. However, phytoplankton biomass declined significantly in all three basins following the invasion of zebra mussels. The low chlorophyll levels shown by Nicholls and Hopkins may have been biased because samples were from nearshore intakes where filtration by mussels inside the intake structures themselves may have added to in-lake effects.

Long-term assessment of phytoplankton communities has been hindered by the lack of long-term biomonitoring including the offshore. Lack of consistency between key studies in sampling location, frequency, seasonal duration, methodology and the manner in which data have been reported creates problems in assessing the health of phytoplankton communities. The water intake study of the Ontario Ministry of the Environment and Energy stands as one of the best long-term records of phytoplankton communities in the lake.

Recent studies have noted very low mid-summer chlorophyll levels (<2 ug$^.$L^{-1}) at some offshore locations in the west and west-central basins (Dahl et al. 1995, Graham et al. 1996). However, late summer to fall chlorophyll levels increased to 8-10 ug$^.$L^{-1} in 1993 (Dahl et al. 1995) compared to 2-5 ug$^.$L^{-1} in the 1988-90 period (Leach 1993). Very low seasonal mean total phytoplankton biomass (0.3 g$^.$m^{-3}) and chlorophyll (1 ug$^.$L^{-1}) typical of oligotrophic lakes (Vollenweider 1968) have recently been measured in the east basin (Dahl et al. 1995). Phytoplankton and phosphorus levels have always been much lower in the east basin compared to

the rest of the lake. However, there is now concern that phosphorus levels have declined in spite of stable P loading and that phytoplankton biomass is also lower than predicted by P concentration.

13.4.3 Trophic Classification

Changes in trophic status have been assessed using seasonal data on total phytoplankton biomass (Table 13.1) and the classification scheme of Munawar and Munawar (1982) (Table 13.4). In the west basin, trophic status quickly responded to reduced P supply while changes in the east basin have been gradual until zebra mussel colonization led to a rapid switch to ultra-oligotrophic conditions. However, as Nalepa and Fahnenstiel (1995) correctly point out, assessing only pelagic responses to P management is no longer valid in areas with large populations of zebra mussels. Trophic classification in areas such as the east basin of Lake Erie must include the benthos and will likely show that total ecosystem productivity has remained the same with a shift from the pelagia to the benthos.

13.4.4 Chlorophyll:Phosphorus Ratio

There is overwhelming scientific evidence showing that the primary productive capacity of most lakes is set by P supply (Schindler 1974, 1975, 1978). Seasonal mean chlorophyll as an index of phytoplankton biomass during the growing season (spring to fall) is usually positively correlated to mean P levels for the same period (Nicholls and Dillon 1978). However, factors such as climate and food-chain structure can alter this relationship in any lake and predicted chlorophyll levels. Recently, Mazumder (1994) has refined chlorophyll vs. phosphorus relationships to account for the controlling influence of grazing pressure and trophic status. Lakes with well developed communities of large grazing zooplankton such as cladocerans will achieve lower biomass levels for equivalent levels of P compared to those where the grazing community is either absent or poorly developed. Those systems with well-developed grazing communities are referred to as even-linked while those without are odd-linked. We consider that the ecological function of zebra mussel communities as grazers is similar to cladoceran zooplankton communities although at a more intense level. We have assumed that phyto- and zooplankton communities in the east and central basins of Lake Erie functioned as an even-link system during the 1993-94 period because zebra mussels maintained grazing pressure on

phytoplankton although biomass of *Daphnia* differed drastically between years (Graham et al. 1996).

Chlorophyll concentrations for the three basins have been predicted from P concentration using three separate relationships from the literature (Table 13.5). In all instances, with the exception of nearshore locations in the east basin, observed values were either close to or higher than predicted values. Nearshore values in the east basin in 1993 and 1994 were well below predicted levels in both 1993 and 1994 although biomass of large cladocerans increased from an almost non-existent level in 1993 to 3.19 g·L^{-1} in 1994. Spatial chlorophyll data from the east basin in the spring also shows that levels at many locations in the east basin are well below those expected based on P levels compared to the rest of Lake Erie and to Lake Ontario (S. Millard, Burlington unpublished data). The west basin was treated as an even-link system but observed values were higher than predicted suggesting that grazing by mussels and cladocerans was not as extensive as in the other two basins. These findings differ from those of Nicholls and Hopkins (1993) who found that observed chlorophyll levels in the west basin were substantially lower than those predicted from P loadings. This apparent discrepancy may be due to the nearshore sampling in the later study, different years of sampling or the inherent variability of phytoplankton populations in the west basin (Wallen and Botek 1994, S. Millard, DFO, Burlington, unpublished data).

13.4.5 Changes in Species Composition

The analysis of changes in species composition of phytoplankton communities has been based on several key studies (Verduin 1964; Munawar and Munawar 1976; Gladish and Munawar 1980; Makarewicz 1993a; Nicholls and Hopkins 1993; Dahl et al. 1995).

13.4.5.1 West Basin.

A common theme in several studies was the increase in nuisance and eutrophic indicator species with advancing eutrophication through the 1960's and early 1970's. For example, Verduin (1964) concluded that prior to 1950 the west basin was dominated by the mesotrophic species *Asterionella formosa* as well as *Tabellaria fenestrata*, and *Melosira ambigua* which

subsequently disappeared with accelerated P enrichment up to the late 1960's. P control has reversed some of these trends with a gradual reduction in the abundance of eutrophic indicator species. The variable biomass of *A. formosa* and another mesotrophic species *Rhizosolenia eriensis* also indicates the effect of P control on trophic status and phytoplankton species composition. Makarewicz (1993a) found that from 1983-87 these two species had densities comparable to those in the 1930s and '40s indicating a return to mesotrophic conditions. Makarewicz (1993a) found that eutrophic species of the genus *Stephanodiscus* (*S. binderanus, S. niagarae, S. tenuis*) were still present although levels were much reduced (70-98%) compared to pre-P control. Makarewicz also found that biomass of *Aphanizomenon flos-aquae*, and *Rhodomonas minuta* was similarly reduced with P control and species composition was similar between 1970 and 1983-87 for the common species. More recently Dahl et al. (1995) did not find eutrophic species such as *Fragilaria capucina* and *S. tenuis* in the western basin. This study found that diatoms were still the dominant group however, biomass declined 68% compared to the 1983-87 period studied by Makarewicz. This finding agrees with that of Holland (1993) who found a startling decline in diatom biomass following zebra mussel colonization in Hatchery Bay in the west basin. In addition, Dahl et al. (1995) found a 60% increase in Chrysophycean biomass which they suggested may have been a response to reduced competition from diatoms. It is difficult to attribute the long-term fluctuations in biomass of some species to a causative factor. For example, *S. alpinus* was abundant in the 1983-87 study of Makarewicz, but was not present in 1970, was dominant in the 1960's and present in low numbers in the 1930's & 40's.

Zebra mussels may have a less selective effect on species composition than changes in nutrient regimes. For example, Nicholls and Hopkins (1993) showed that there was equal decline in all phytoplankton groups including large colony-forming diatoms following the invasion of zebra mussels. Holland (1993) also found that zebra mussel filtering had a striking impact on diatom biomass in a bay in the west basin where biomass declined about 80% compared to years just prior to zebra mussel invasion.

Factors other than P control or zebra mussel colonization have caused significant changes in species composition of phytoplankton in the west basin. Nicholls et al. (1980) found significant declines in total biomass related to P control but noted high summer biomass of *Fragilaria*

crotonensis in some years related to water level fluctuations. These same authors also concluded that fluctuations in biomass of dinoflagellates were not related to changes in nutrient regime or directly to lake level but were influenced by sampling at a water intake.

There have been anecdotal reports of dense summer blooms of *Microcystis aeruginosa* in the west and central basin since 1994 leading scientists to speculate that these blooms are linked to zebra mussel colonization. This species has always been a component of the phytoplankton community in the west basin although researchers differ in their opinions on its historical quantitative importance. In a lakewide survey in 1970, Munawar and Munawar (1976) reported *Aphanizomenon flos-aquae* and *Anabaena spiroides* as the dominant blue-green species. However, data from the water intake at Union show that *M. aeruginosa* has always been a dominant blue-green alga (K. Nicholls OMEE, pers. comm.). It is unclear whether these recent observations represent a significant increase in biomass of this species.

Microcystis aeruginosa is one of a group of blue-green algae that form gas vacuoles and float to the surface during calm weather to form surface scums, a phenomenon referred to in the early literature as "flowering of the waters" (Reynolds and Walsby 1975). The term bloom was originally used only in reference to surface accumulations of blue-green algae but is now (perhaps mistakenly) used to describe any dense assemblage of phytoplankton. Blooms of gas-vacuolate blue-greens are the result of redistribution of biomass in the water column and not necessarily an actual increase in biomass (Reynolds 1971). Moreover, these surface scums are easily dispersed through the water column by moderate winds.

The ecological significance of these blooms in Lake Erie is presently unknown. This species was already important in the phytoplankton of Saginaw Bay prior to zebra mussel colonization. However, *Microcystis* has become dominant in the phytoplankton of Saginaw Bay since zebra mussels invaded because it is less impacted by filtration compared to Chlorophytes and diatoms (Fanslow et al. 1995) although absolute biomass has also increased (T. Johengen, pers. comm.). An alternate hypothesis presented by Heath et al. (1995) is that zebra mussels may have increased nutrient recycling and given a competitive advantage to those algae such as *Microcystis* that achieve maximal growth rates in high-nutrient environments.

Recent findings suggest that the dominance of phytoplankton communities by Microcystis in zebra mussel infested parts of the Great Lakes may be predictable. This species has become relatively abundant in recent years (1996/97) in the Bay of Quinte, Lake Ontario where zebra mussels have colonized since 1994 (K. Nicholls OMEE pers. comm.). Summer peaks in chlorophyll now coincide with peaks in biomass of Microcystis.

13.4.5.2 Central Basin

Evaluating change in the phytoplankton community of the central basin is more difficult than for either of the other basins because there have been fewer studies. Conditions in this part of the lake were never eutrophic as in the west basin suggesting that a strong response to P control should not have been anticipated. Annual variability in other factors such as mixing, temperature and grazing may be as important as trends in phosphorus loading in affecting long-term trends in species composition in the central basin.

Makarewicz (1993a) found that *Melosira granulata*, a dominant diatom in the central basin in 1969 (Hohn 1969), was not common in 1970 and was rare from 1983-86 but increased in numbers in 1987. Similarly, *Tabellaria fenestrata* and *Fragilaria crotonensis* which were common in 1960 (Hohn 1969) became rare by 1983-87 (Makarewicz 1993a). *Stephanodiscus tenuis* had a similar pattern as in the west basin, dominant in 1970 but not common from 1983-87. Makarewicz (1993a) found that dominance fluctuated between several species from 1983-87 in the central basin such as *S. niagarae* (1983: 44.4%), *Asterionella formosa, Rhodomonas minuta* and *Ceratium hirundinella* (1984: 6.3, 6.8 and 9.2% respectively), *S. hantzschii* (1985: 36.1%), *Tabellaria flocculosa and C. hirundinella* (1986: 8.6 and 10.0% respectively) and *S. niagarae* (1987: 29.8%). The recent biomonitoring study in 1993 (Dahl et al. 1995) showed that community structure has changed little since 1987 although there was an increase in the eutrophic indicator *M. granulata*. However, several species of *Cyclotella*, indicators of oligotrophic conditions were also present.

13.4.5.3 East Basin

Prior to P control (1970), diatoms were relatively less abundant and dinoflagellates (*Gymnodinium helveticum* and *G. uberrimum*) more important in the total biomass of the east

basin compared to the rest of the lake (Munawar & Munawar 1976). Phytoflagellates showed an increasing relative importance from west to east in 1970. Dominance of various species varied considerably from year to year in the 1983-87 period studied by Makarewicz although he concluded that species composition had not changed much since 1970 with the exception of a significant reduction in biomass of *S. tenuis*. Species such as *Stephanodiscus niagarae*, *Fragilaria crotonensis*, *Tabellaria flocculosa, Ceratium hirundinella, Peridinium sp*., *Cryptomonas erosa*, and *Cosmarium sp*. were dominant at various times in his study. More recently, Dahl et al. (1995) have similarly concluded that species composition in the east basin has not changed drastically since the 1983-87 study of Makarewicz with a few notable exceptions. For example, the relative contribution of *Rhodomonas* to total biomass has increased although the biomass of this genus has declined . One of the most significant changes may be the appearance and at times significant biomass (>10%) of *Dinobryon* which was not seen in the 1983-87 study. Dahl et al (1995) found peaks in diatom biomass higher than those in the Makarewicz study, however these were short-lived and diatom biomass was low throughout the rest of the stratified period. Other notable changes in 1993 were the disappearance of *S. niagarae* and *C. hirundinella* since 1983-87. The absence of these two eutrophic species as well as *Melosira binderana*, an increase in smaller phytoplankton species and the presence of *Dinobryon* are all indicators of a shift to a more oligotrophic environment.

13.4.6 Photosynthesis

Phytoplankton photosynthesis (PP) is the main pathway for energy input to the pelagia of Lake Erie. A decline in this process could have a negative impact on higher trophic levels dependent on planktonic energy at any life stage. The areal rate (g C m^{-2}) of PP is a complex process that integrates carbon fixation by phytoplankton over the water column through the interaction of water clarity, biomass, physiological properties of the phytoplankton , vertical mixing and solar irradiance. Compensatory interactions between these determinant variables can buffer against changes in areal rates of PP to some degree. However, volume-based rates of photosynthesis at optimal light levels in the upper part of the water column decline with chlorophyll. The impact on areal PP is only partially offset by deeper photosynthesis as the water column becomes more transparent. Several recent studies have shown that reduced chlorophyll led to significant declines in areal PP in spite of improved water clarity (Millard and Sager 1994,

Fahnenstiel et al. 1995). Species succession is a longer-term adaptive strategy for phytoplankton communities to maximize areal photosynthesis as growth conditions shift in amplitude and frequency from the optimum for any population. Compensatory responses in photosynthetic parameters can also offset potential changes in PP. For example, chlorophyll concentrations drop to seasonal minima in mid Lake Ontario during the summer but the potential impact on areal PP is partially offset by an increase in photosynthetic capacity (P^b_m) because of higher temperatures and a shift in species composition to phytoflagellates (Millard et al. 1996).

Both global and regional relationships exist between phytoplankton productivity and P supply across a range of lakes (Schindler 1978, Vollenweider et al. (1974). Vollenweider et al. (1974) quantified the relationship between annual areal PP vs. P loading for the Laurentian Great Lakes.   More recently, Millard et al. (1996) developed a similar relationship for Lake Ontario and the Bay of Quinte relating seasonal (May 1-Oct 31) areal PP to seasonal mean P (TP) concentration (Y=419.8 TP/20.8 + TP, r^2=0.81, n=27). Additional data from Lake Erie had little effect on parameterization of the curve (Y=397.5 TP/19.4 + TP) (Graham et al. 1996, Millard et al. 1998 in prep.). Comparing carbon input over longer time frames (e.g. annual or growing season) to the potential set by P loading may be a valuable tool for assessing limitations to productive capacity of ecosystems. Measured areal PP from the Lake Erie biomonitoring project for 1993 (Dahl et al. 1995) and 1994 (Graham et al. 1996 and S. Millard unpublished data) has been compared to predicted values using the above equation (Table 13.6).

Significant PP rates occurred down to the bottom because of the high transparencies and shallow depths at both east basin nearshore stations. Profiles at these stations were truncated at the bottom. This means that photosynthesis profiles could have theoretically extended beyond the bottom resulting in higher areal than given in table 6. In addition, non-algal sources of light extinction lowered transparency at E1 thus reducing areal PP compared to E3 where chlorophyll and photosynthetic parameters were similar but the water column was more transparent. Low transparency and shallow depth, reduced observed PP at these nearshore stations to below the potential set phosphorus. Corrections for both effects were applied to the areal PP values used for E1 and E3 in table 13.6. In spite of these corrections, predicted levels of areal PP were still

consistently greater than observed, indicating that photosynthetic input of energy in this part of lake is below the potential set by P.

13.4.7 Conclusions

1. Eutrophic conditions in the west basin peaked in the 1960's and have been eliminated by P management. West basin phytoplankton populations can no longer be considered impaired by eutrophication. P enrichment in the lake as a whole is restricted to local (e.g. nearshore at Cleveland, S. Millard DFO unpublished data) and RAP areas (Wheatley Harbour) and the mouths of some rivers (Grand R., Maumee R.). Declines in phytoplankton productivity due to the P control program **do not constitute impairment** according to our definition of ecosystem health.

2. Trophic classification based on phytoplankton biomass now ranges from mesotrophic to ultra-oligotrophic from the west to east basin. Phytoplankton productivity in the west basin in 1993 appeared to be in equilibrium with current P supply and may have been less impacted by zebra mussels over the whole basin than suspected. However, this relationship may need to be reevaluated because zebra mussels are more abundant because of colonization of soft sediments. The trophic status of the pelagia in the east basin has always been lower than the rest of the lake but is now ultra-oligotrophic due to the combined effects of P reduction and zebra mussel grazing. Trophic status of the east basin may not have changed if the benthos is included.

3. Phytoplankton biomass and seasonal photosynthesis are below the potential set by P loading in the nearshore of the east basin due to zebra mussels. Reduced energy input into the phytoplankton may have an impact on higher trophic levels dependent on energy transfer through the pelagic food chain. Offshore levels of both of these variables in the east basin are close to values predicted by phosphorus levels.

4. Phytoplankton productivity in the east basin may be further reduced if zebra mussel colonization results in a change in the P loading vs. concentration relationship due to sequestering of P in zebra mussel tissue or changes in recycling patterns.

5. Diatom biomass is now very low and some species typical of the east basin have been lost. This represents a significant disruption of an energetic pathway and may be a key factor in the decline of the benthic amphipod *Diporeia* (R. Dermott, DFO Burlington, personal communication).

6. Lack of a long-term biomonitoring program in Lake Erie has hampered assessment of lower trophic level productivity. Conclusions on impairment have been reached using simple predicative relationships developed from recent biomonitoring programs on both Lakes Erie and Ontario. There have been inconsistencies between studies on the impact of zebra mussels in the west and central basins that may be related to location of study sites. Zebra mussels may be having as big an impact in the very nearshore regions of the west basin as in the east, however, extensive colonization of the deeper areas of the east basin has had notable impacts on phytoplankton biomass in the offshore during the spring. Lower numbers of quagga mussels in the west basin and less colonized area may explain why impacts in the west basin appear to be less.

13.4.8 Impairments

Nearshore phytoplankton populations in the East basin are considered impaired on a seasonal basis as well as in the offshore in the spring due to zebra and quagga mussel grazing. Impairment is on total standing crop, photosynthesis, loss of key species and trophic transfer to *Diporeia*.

13.5 Assessment of Beneficial Use Impairment of Zooplankton

13.5.1 Historical Trends in Zooplankton:

The first basin-wide surveys on Lake Erie were conducted by the U.S. Fish and Wildlife Service between 1928 and 1930 (Wright 1955, Fish 1960). These surveys were undertaken in response to serious declines in the highly prized whitefish and cisco fisheries. In the east and central basins, zooplankton were sampled with 1-m, 112-μ (# 12) mesh nets trawled just below the surface and just off the bottom of the lake once a month from June to September. In the west basin, zooplankton were sampled twice a month at 16 sites with a 112-μ mesh closing net in 1928, and in 1929 and 1930 with a series of 45-L trap samples filtered through 112-μ mesh. Samples were taken from May to November. The next seasonal, whole lake, zooplankton survey was conducted by the Canadian Department of Fisheries in 1970 at the height of the eutrophication problems (Watson 1976). Samples were collected monthly from April to December at 30 sites using a 50-cm diameter, 64-μ mesh net hauled vertically to the surface from 2 m off the bottom or from 50 m depth, when sounding depth was greater than 50 m. In 1983, the U.S. Environmental Protection Agency started a monitoring program on Lake Erie to determine whether the reductions in phosphorus loading to the lake were having the expected effects. Zooplankton samples were collected from 21 sites along the east-west, mid-lake transect twice in late April and again twice in August. A 50-cm diameter, 62-μ mesh net was towed vertically from 20 meters depth or from 1 m above the bottom at shallower sites. To date, data from 1983 to 1987 have been reported (Makarewicz 1993b). In 1993, the Canadian Department of Fisheries and Oceans studied the seasonal dynamics of phytoplankton, zooplankton and benthos at eight sites, three in each of the east and west basins and two in the west-central basin of Lake Erie (Fig.13.1) (Dahl et al. 1995, Graham et al. 1996). The eastern basin stations were visited again in 1994. Zooplankton were collected with a 110-μ mesh, 50-cm diameter net pulled vertically through the water column from 2 m above the bottom. This study was initiated in response to changes in water quality and declines in the perch and smelt fisheries coincident with the arrival of zebra mussels in the lake. Seasonally and spatially more restricted studies have been carried out by a number of investigators; however, the above studies capture the basic changes in

the zooplankton community during the period from 1928 until 1993. The slight differences in mesh-size amongst the nets used in these historical studies do not affect the data on species composition and biomass. The 112-u mesh nets would have captured fewer cyclopoid nauplii. However, depending on the study, nauplii were either not included in the sample enumeration or were treated as a separate group. Counts of other stages and species should not be affected.

Members of the zooplankton community have not changed since the 1928-30 period except through the addition of exotics: *Eurytemora affinis* (1961), *Eubosmina coregoni* (196?), *Bythotrephes cederstroemi* (1985) and veligers of *Dreissena* spp. (1988). However, the relative abundance of species has changed with changes in physical and nutrient conditions and with planktivore abundance to the extent that some dominant, cold-water members of the 1928-1929 east basin community are now rare.

13.5.1.1 Eastern and Central Basins

The dominant zooplankton species in the central and eastern basins in 1928-1929 were the large *Daphnia* (*D. pulex* and *D. galeata mendotae*) and large calanoids (*Leptodiaptomus ashlandi, L. sicilis, Limnocalanus macrurus* and *Epischura lacustris*). *Mysis relicta*, the opossum shrimp, was also abundant and restricted to the hypolimnion of the eastern basin during the stratified period.

By 1970 the zooplankton communities in the eastern and central basins had changed considerably. Cyclopoids had replaced calanoids as the most abundant copepods: the dominant species was *Diacyclops thomasi*. Amongst the calanoids, *Skistodiaptomus oregonensis* and *Leptodiaptomus minutus* had replaced *L. ashlandi* and *L. sicilis* as dominants. *Limnocalanus* and *Epischura* had virtually disappeared from the central and eastern basins. The dominant cladocerans were *Bosmina*, *Daphnia retrocurva* (more abundant in west basin), *D. longiremis* (more abundant in eastern regions), and *D. g. mendotae* (found lakewide after August). *Daphnia pulex* was not observed. *Eubosmina coregoni* was first reported in the lake in the late 1960's and was secondarily important in the central and eastern basins by 1967 (Gannon 1981).

Bythotrephes cederstroemi entered the lake in 1985 and was found in high abundance in the central and eastern basin in August of 1986 and 1987 (Bur et al. 1986, Makarewicz 1993b). By the mid-1980's, calanoids had again become more important than cyclopoids both numerically and in terms of biomass in the eastern basin. However, the biomass of cyclopoids still exceeded that of calanoids in the central basin. *Limnocalanus*, *L. sicilis* and *Epischura* were not considered common species or mentioned by Makarewicz (1993b). This may be due to the fact that these species were relatively rare and/or to the shorter sampling depth which would have missed hypolimnetic species during the stratified period.

In 1993 the zooplankton communities in the eastern and west-central basins differed considerably. In the west-central basin *Bosmina* was the dominant cladoceran; however, *Daphnia retrocurva* and *Eubosmina* were sometimes important. Although *Leptodora* was observed on occasion, *Bythotrephes* was the only abundant predatory cladoceran. The dominant calanoids were *Epischura*, *Leptodiaptomus sicilis*, *L. minutus*, and *L. ashlandi* and the dominant cyclopoid was *Diacyclops thomasi*. Here, cyclopoids were more important than calanoids both numerically and in terms of biomass. The biomass of veliger larvae was important on occasion. In the eastern basin, *Bosmina* was the only important herbivorous cladoceran, while the predatory cladocerans, *Leptodora* and *Bythotrephes*, were abundant on occasion. Calanoid composition was diverse. *Epischura* had become the dominant calanoid both inshore and offshore while *Eurytemora*, *L. minutus*, and *S. oregonensis* were sometimes abundant offshore. *Limnocalanus*, *L. sicilis* and *L. siciloides* were observed but were never important components of the community. *Diacyclops* was the dominant cyclopoid. Calanoids maintained numerical and biomass superiority over cyclopoids. The veliger larvae of zebra mussels contributed a significant amount to zooplankton biomass from June until October. In 1994, zooplankton community structure in the eastern basin changed, becoming more similar to that observed in the west-central basin in 1993. *Bosmina* was still the dominant cladoceran, but *D. retrocurva* and *Eubosmina* were abundant for short periods of time. *Epischura* and *L. minutus* remained abundant both inshore and offshore. In the offshore, *S. oregonensis* and *L. ashlandi* replaced *Eurytemora* in importance. Calanoid abundance no longer exceeded cyclopoid abundance although their biomasses were similar. *Diacyclops* remained the dominant cyclopoid. The switch in relative abundance of calanoids and

cyclopoids is thought to be related to changes in predation pressure from different year classes of smelt (Johannsson and Graham (submitted)).

13.5.1.2 Western Basin

The dominant zooplankters in the western basin in 1928-1930 were Skistodiaptomus *oregonensis*, *Acanthocyclops vernalis*, and *Daphnia retrocurva*. In 1928 and 1929 diaptomids (calanoids) were much more common than cyclopoids during the July-September period; however, in 1930 the opposite pattern occurred. Temperatures were higher in 1930 and it was hypothesized that this may have allowed for the summer peak in cyclopoids. *Daphnia* increased in abundance in late May, peaked in June and declined sharply in July (1928) or tailed off through July and August (1929, 1930). A pattern observed repeatedly in later studies.

In 1970, cyclopoids were the dominant form of copepod in the spring as well as in the summer: *Diacyclops thomasi*, *Acanthocyclops vernalis* and *Mesocyclops edax* were the most common species. Cladoceran abundance rose in early June and remained high until early October: as in the past, the dominant species early in the season were *Bosmina* and *Daphnia retrocurva*, the summer cladoceran populations were composed of *D. g. mendotae* and *Eubosmina*. *Eubosmina* was first observed in the lake in the late 1960's. Amongst the calanoids, *Leptodiaptomus minutus* and *Skistodiaptomus oregonensis* were the most important species. *Eurytemora affinis* was first observed in the lake in 1961 (Engel 1962 in Balcer et al. 1984) and was more abundant in the west basin than elsewhere in the lake (Watson 1976).

In the 1983 to 1987 period, cyclopoids continued to be more important than calanoids to the zooplankton community. The dominant cyclopoids remained *Acanthocyclops vernalis*, *Diacyclops thomasi* and *Mesocyclops edax*. The dominant calanoids, however, were *Leptodiaptomus ashlandi* and *L. siciloides*. Amongst the cladocerans, *Daphnia retrocurva* and *Eubosmina* were the most important species in terms of biomass and abundance, although several other cladocerans were also common: *D. g. mendotae*, *Chydorus sphaericus, Bosmina* and *Diaphanosoma.* Of the predatory cladocerans, *Leptodora* made up a significant amount of the cladoceran biomass. *Bythotrephes* was not observed in the U.S. EPA study. *Bythotrephes* was found in the western basin by Berg and Garton (1988) in night-time samples collected near the

surface in 1987. *Bythotrephes* appears to prefer cooler temperatures (Garton et al. 1990) and may not have been present in April or August when samples were collected.

In 1993, seasonal cyclopoid and calanoid abundances were very similar, cladocerans were by far the most abundant organisms and the most important in terms of biomass. The dominant calanoids were *Eurytemora* and *Epischura* although *Leptodiaptomus ashlandi* and *L. sicilis* were occasionally abundant. *Diacyclops* was the only important cyclopoid. Amongst the cladocerans, *Daphnia retrocurva* and *Bosmina* were dominant. *Leptodora* was present. *Bythotrephes* was not seen. It is possible that *Bythotrephes* may retreat to the bottom during the day and therefore be missed during daytime sampling. Of a total of eight nighttime tows in the west basin between July 1992 and September 1994, only one sample contained *Bythotrephes*: abundance was 0.6 individuals.m^{-3} (W.G. Sprules and E. Jin, Erindale College, University of Toronto, Mississauga, unpubl. data).

13.5.2 Potential Stressors

Five categories of stressors are known to impact the structure and function of zooplankton communities. They are alterations in the physical habitat, changes in food resources, excessive predation, immigration of exotics, and contaminant poisoning.

The *physical habitat* refers to conditions such as oxygen levels, temperature, pH, suspended sediment, light levels and for nearshore and benthic zooplankton, substrate characteristics.

Changes in *food resources* result from changes in nutrient loading to the system or alteration in energy pathways through the addition of exotics, damage to the phytoplankton community, or changes in planktivory. Different zooplankton species are adapted for living under high and low food resource conditions. It is the change in condition which alters the zooplankton community structure. If the change in nutrient conditions, phytoplankton community or level of planktivory is uncharacteristic for the system then alterations in the zooplankton community would be an impairment. In systems where zooplankton species are not controlled adequately by predators, zooplankton populations grow until they are food limited. There is a bottleneck in

energy transfer up the food chain and little energy is passed on to other organisms, again resulting in an impairment. In a balanced system, zooplankton populations would be controlled by both predators and food resources.

Energy is passed up the food chain through *predation*, but predators can affect the population structure and production of their prey. Predatory fish (planktivores) tend to remove the largest zooplankton while invertebrate predators tend to remove the smaller zooplankton. If predation by fish is severe, larger species of zooplankton become rare or disappear and the size of the remaining species is depressed. The probability of an individual reaching a large size and producing many young is lessened, thus production is depressed. If invertebrate predation is severe, few young zooplankters survive to become adults and reproduce. Again production is depressed. Mills et al. (1987) have suggested that a mean size of 0.8 mm for an individual in the zooplankton community (caught with a 153-µ mesh net) indicates a appropriate level of planktivory. They found that when zooplankton mean size is <0.8 mm the ratio of piscivores to planktivores, that is fish-eating to plankton-eating fish, was <0.2 and the fish community was not balanced, but dominated by planktivores. More energy is passed up the food chain when zooplankton are large (Sprules 1980). The relationships examined by Mills et al. (1987) between zooplankton mean size and the ratio of piscivores:planktivores was based on a summer (July or August) sample from each lake. Because young-of-the-year (YOY) fish feed on zooplankton, a strong year class can impact the zooplankton community especially later in the summer. To better evaluate the state of the fishery, Mills et al. (1987) suggested assessing size in spring and summer. If zooplankton were large in either period, it suggested a balanced ratio of predator:panfish. If zooplankton were large in both periods, it suggested that the system was weighed towards predators and that the crop of YOY fish was not as large as it could be.

Exotics can disrupt or change the functioning of the system and the passage of energy up the food chain. If the exotic exerts excessive predation on the zooplankton community, decimates the zooplankton community by removing its food resource, or is a zooplankter which is not eaten and therefore diverts energy from going up the food chain, then the exotic causes an impairment of the zooplankton community.

Contaminants can cause impairment by eliminating species or reducing the productive capacity of the populations; that is the ability of the zooplankton populations to grow. No assessment has been done on possible contaminant impacts on Lake Erie zooplankton.

13.5.3 Methods of Assessment

Zooplankton species composition in Lake Erie has changed since the 1920's both in terms of relative abundance and with the addition of exotics. Although many of the same species are present, the community can not 'return' to some previous state. We must assess the structure and function of the present community against the definition of biological integrity presented in the introduction.   Such a community should be *balanced, integrated, and adaptive* with a *species composition, diversity and function* comparable to that of the natural region.

A balanced community is structured by both predators and food supply. Under such conditions, the mean size of an individual in the community should be around 0.8 mm (Mills et al. 1987), and the community should be comprised of both herbivores and predators, and both cladocerans and copepods in a normal ratio. The members of the community should be integrated into the food web efficiently so that species occur in their normal abundance in their historical habitats and so that energy is not accumulated in species for which there is no predator. Protocols for assessing biological integrity have not been developed. At this point in time we can assess the balance in the community by comparing the mean size of an individual to the criterion developed by Mills et al. (1987). We can compare the ratio of cladoceran to copepod abundance and predator to herbivore biomass with historical ratios and ratios for other lakes. To some extent, we can examine the integrity of the system by looking for changes in species richness and relative abundance composition. We can examine the functioning of the system by comparing rate processes such as production per unit of biomass (P/B) to that in other lakes and to empirical relationships.

13.5.4 Community Integrity:

Species richness has increased since the 1920's with the addition of three exotic species of zooplankton and *Dreissena* veliger larvae. However, species relative abundance has changed considerably during the same time.

The cold water species, *Limnocalanus macrurus* and *Leptodiaptomus sicilis,* were characteristic of the eastern basin in 1928-1929. During the winter and spring they were found throughout the lake, however, the warmer temperature and at times low oxygen conditions in the central and western basins during the summer were unfavourable (Gannon and Beeton 1971). The hypolimnion in the east basin provided a refuge for them during the stratified period. By 1970, the distribution of *L. macrurus* was essentially restricted to the west basin; nor were they found in the east basin in 1967-68 or 1974 (Davies 1968,1969, Cap 1980). This skewed distribution towards the west basin was also observed in 1983-1987 and 1993. Populations in the west might be maintained by continued immigration from Lake Huron (Gannon and Beeton 1971). By 1993, *Limnocalanus* was occasionally observed in the east basin.   By 1970, *L. sicilis* was also rare through out the lake. In 1993 it was an important component of the zooplankton community in the west and west-central basins on occasion, but not in the east basin.

Two factors are thought to be responsible for the loss of *Limnocalanus* from the eastern basin: low oxygen levels and increased planktivory (Gannon and Beeton 1971). Levels of 5.5 $mgO_2.L^{-1}$ observed in the eastern basin in September 1961 are believed to be near the tolerance limit of *Limnocalanus* eggs and nauplii (Strom 1946 in Gannon and Beeton 1971). Increased planktivory would affect both *Limnocalanus* and *Leptodiaptomus sicilis*.   The loss of these two large, cold-water, species coincided with the immigration and expansion of smelt in the lake (Gannon and Beeton 1971). Smelt are a cold-water, planktivorous fish. As smelt age they move to deeper, colder water and can be found in the metalimnion and hypolimnion in the Great Lakes (Scott and Crossman 1973). For this reason they are more abundant in the eastern and east-central regions of Lake Erie than in the west. Their habitat overlaps completely with that of *Limnocalanus* and *L. sicilis*. Smelt were first reported from Lake Erie in 1935 and became abundant during the 1940's (Van Oosten 1937, Christie 1974). In 1993, oxygen conditions in the eastern basin were favourable, but smelt were still abundant.

Mysis relicta, the opossum shrimp, was reported through out the colder waters of the eastern basin in 1928-30 (Fish 1960), present in low numbers in 1971 (Carpenter et al. 1974) and not observed in the plankton between the late 1970's and 1997. One mysid was found in a smelt gut in a March 1994 survey (Witzel, OMNR Port Dover, unpubl. data) and a few mysids were captured in the eastern basin in September 1997. It is possible that planktivory and the loss of food sources, such a *Limnocalanus* and *Leptodiaptomus sicilis,* are implicated in their decline.

Larger zooplankton are preferentially consumed by fish compared with smaller zooplankton. In the eastern-basin hypolimnion, these larger species have been replaced by smaller species - *Bosmina*, veliger larvae, and *Diacyclops,* in order of total biomass. The loss of all larger species in a habitat and their replacement with smaller species could be considered an impairment.

13.5.5 *Community Balance*:

13.5.5.1 *Mean Individual Size*

Larger zooplankton species are more abundant when planktivore abundance is low. These include *Daphnia pulex*, *D. galeata mendotae*, *Mesocyclops edax*, *Limnocalanus macrurus* and *Leptodiaptomus sicilis*. The first three are warm water species which inhabit the meta- and epilimnion while the latter two are cold water species which prefer the hypolimnion and lower metalimnion.

As mentioned above, the cold water species are now rare in their dominant habitat.

The warm water species, *Daphnia pulex* and *D. galeata mendotae* have been observed as dominants in all basins of the lake. *Mesocyclops edax* tends to be a more important member of the zooplankton community in the western basin than the central or eastern basins. In the 1928-1930 period *D. pulex* was a dominant species in all three basins while *M. edax* was abundant in the western and common in the central and eastern basins. These distributions indicate that planktivorous fish were not overly abundant in this period.

In 1970, *Daphnia pulex* was not found, *D. g. mendotae* was abundant lakewide and *M. edax* was again the dominant cyclopoid in the western basin (no comment made of its abundance in central and eastern basin). In the 1983-1987 period, *D. pulex* and *D. g. mendotae* were dominant cladocerans in the central and eastern basins but only the latter was important in the western basin. *Mesocyclops* maintained its dominance over *Diacyclops thomasi*, a small cyclopoid, in the western basin. The lakewide abundance of *D. g. mendotae* and the importance of *M. edax* again suggest that planktivorous fish did not dominate the fish community in 1970 or through the mid-1980's

In 1993, *Daphnia pulex* and *D. g. mendotae* were not observed in the western or west-central basins, only the smaller *D. retrocurva* was abundant. *Daphnia g. mendotae* and *D. retrocurva* were present in low numbers in the eastern basin. *Mesocyclops edax* was important on occasion in the western, west-central and eastern basins (no central basin stations were sampled). *Diacyclops thomasi* now surpassed *M. edax* in abundance and biomass in all basins. The decrease in relative importance of these large warm water species indicate that planktivory was more severe in 1993 than during any of the previous surveys.

The New York Department of Environmental Conservation began monitoring zooplankton in the eastern basin at two sites off Dunkirk in 1985 using 153-μ mesh nets. The Canadian Department of Fisheries and Oceans examined zooplankton in Long Point Bay in 1993 and 1994 using 110-μ mesh nets. The mean individual size of zooplankton in the eastern basin confirmed that zooplankton were exposed to higher predation rates in the 1990's than in the 1980's. Sizes were smallest in 1991 and 1992, largest in 1985, and 1988 - 1990 and intermediate in 1993 - 1996 (Fig. 13.2a) (D. Einhouse and W. Culligan, NYDEC, Dunkirk, N.Y, unpubl. data; Dahl et al. 1995, Graham et al. 1996). Although zooplankton from Long Point Bay were smaller than those from Dunkirk, this is likely due to the differences in mesh sizes used for sampling: 110-μ versus 153-μ mesh. Zooplankton mean sizes in the 1990's were well below Mill's 'preferred size' and indicate that the community in the eastern basin has been heavily preyed upon by fish since 1991.

No seasonal size data exist for the central basin. Dahl et al. (1995) presented data on mean zooplankton size for the northern part of the west-central basin for 1993: samples collected with 110-u mesh net. Sizes here were consistently near 0.4 mm for the May to October period, values similar to or lower than those in Long Point Bay. Therefore, the mean size determined with a 153-μ mesh sample would likely have fallen below the 0.8 mm 'preferred size' as well

In the western basin between 1988 and 1990, zooplankton mean size was large (approximately 1.2 mm) in spring and early summer and declined to 0.9 mm in August (J. Leach, OMNR, Wheatley, unpubl. data). By 1993, zooplankton mean size had declined although the seasonal pattern of larger animals in the spring and early summer remained (Fig. 13.2b) (Dahl et al. 1995).   In the spring and early summer of 1993, the mean size ranged from 0.4 mm to 0.5 mm at a northern nearshore station and from 0.6 mm to 0.9 mm at a central location. These changes in mean size indicate that planktivory increased between the late 1980's and 1993. Mills et al. (1987) indicated that a zooplankton community of appropriate mean size in either the spring or summer indicated an appropriate ratio of piscivores to planktivores. The presence of large zooplankton in July in the western basin have been reported for many western basin sites over the years, including the 1990's (Culver, Ohio State University, pers. comm.). These observations suggest that the western basin has a fish community with an appropriate balance of piscivores to planktivores.

The small mean size of zooplankton in the eastern and likely also in the west-central basin is considered an impairment as it reflects an imbalance in the fish community between piscivores and planktivores, and because energy flow to the fish community is less efficient in these communities.

We need quantitative conversion factors to convert zooplankton mean size determined from small-mesh samples to the 153-u standard in order to interpret intermediate values more accurately. Intermediate values occurred in the nearshore of the western basin in the spring in 1993 (Fig 13.4) and the offshore of the eastern basin during the August spatial cruise in 1994.

13.5.5.2 Ratio of Cladoceran to Copepod Abundance

The relative proportion of cladocerans:cyclopoids:calanoids is influenced by trophic state, temperature, and predation. Increases in nutrient loads to a lake increase primary productivity, especially during the warmer periods when phosphorus limits phytoplankton growth. At warm temperatures, zooplankton can respond rapidly to the more abundant food supply. Total zooplankton biomass increases, particularly amongst the cladocerans and some cyclopoids which are summer species. The abundance of cladocerans and cyclopoids in western Lake Erie peaked in the late 1960's early 1970's and declined thereafter with reductions in phosphorus loads to the lake (Fig. 13.3) (Makarewicz 1993b, Dahl et al. 1995). Changes were less pronounced or undetectable in the central and eastern basins, areas for which we have less data and for which phosphorus levels did not change as dramatically. Severe planktivory may also alter the ratio depending on whether cladocerans, calanoids or cyclopoids are targeted by the planktivores. Young of the year fish tend to eat calanoids and cyclopoids while older fish take cladocerans.

In 1993, the relative densities of the groups appeared to differ from that expected. In the western basin, where the most data exist, the May to October mean cyclopoid densities were at or below 1928 - 30 densities while seasonal cladoceran densities were still greater than those in 1928 - 1930 (Fig. 13.3). Seasonal calanoid densities also appeared to be lower in 1993 than in the 1928 - 1993 period although the range in densities was small. The ratio of cladocerans:copepods in the western basin increased between 1928 - 1949 and the late 1960's and early 1970's (Fig. 13.4). The USEPA surveyed the lake between July 27th and August 25th every year between 1983 and 1987 (Makarewicz 1993b). According to August cladoceran:copepod ratios, the ratio had decreased to values comparable to the 1928-1949 period by 1986. Although phosphorus levels did not change, the ratio increased again in 1993 and 1994, more than doubling in value. The Ontario Ministry of Natural Resources has been monitoring two sites in the western basin since 1988. They saw a persistent decrease in the density of copepod naupliar densities concomitant with the arrival of zebra mussels and suggest that direct suppression by zebra mussels and food limitation may be responsible (McIsaac et al. 1995). The decrease in naupliar densities would cause the cladoceran to copepod ratio to increase.

Eastern basin, nearshore ratios in 1993 and 1994 were not dissimilar to those expected given the phosphorus concentrations. However, the pelagic food chain was more severely disrupted in this region than in other areas of the lake with much lower levels of photosynthesis and chlorophyll a than expected given the phosphorus concentrations (Graham et al. 1996), and less zooplankton production than expected given the levels of photosynthesis (Johannsson, unpubl. data). It might not be appropriate to compare these cladoceran to copepod ratios with the observed phosphorus concentrations.

The ratios of cladocerans to copepods in Lake Erie do not conform to historical patterns of change with phosphorus concentrations: cladocerans are now relatively more abundant than copepods. Similar changes in the relative abundance of cladocerans to copepods were noted in the Stage 1 Report for Presque Isle Bay (1992) in the early 1990's: the change was attributed to a decline in calanoids. Severe predation was not likely responsible as the intensity of predation varied amongst basins while the ratio changed in all basins. The changes may be linked with the behaviour of dreissenid mussels. The mussels are reported to consume copepod nauplii and to alter the phytoplankton community towards smaller species which would be more suitable for cladocerans than copepods. If ratios remain high, this hypothesis could be tested.

In conclusion, there appears to have been a restructuring of the zooplankton community between 1970 and 1993 in association with changes in planktivory, declines in phosphorus and perhaps the invasion of dreissenids. The zooplankton community is still diverse with a balance of functional feeding groups occupying the various thermal strata (except the hypolimnion of the eastern basin, as discussed above). At this point, the change in ratio of cladocerans to copepods is seen as a change in state, not an impairment. If it were shown that this change in relative composition is detrimental to energy flow in the system or survivorship of native species, the status would become impaired.

13.5.5.3 Ratio of Predatory to Total Zooplankton Biomass (P/B)

At this point the use of predator/prey ratios and production/biomass ratios have not yet been developed as indicators; however, they are potentially very useful. For this reason, preliminary information on these measures is presented for Lake Erie.

The ratio of carnivorous to herbivorous zooplankton affects the efficiency of energy transfer from phytoplankton to fish (see Sprules 1980). Carnivorous zooplankton feed on other zooplankton (cladocerans, copepods and rotifers) and thus insert another trophic level in the food chain between herbivorous zooplankton and planktivorous fish. To the extent that predatory zooplankton eat other predatory zooplankton, an additional trophic level is also inserted. Each trophic level uses approximately 80% of the food it consumes in respiration; and therefore, only 20% of the biomass consumed can be passed on to its predators (Sprules 1980). This is energy lost to fish. However, not all prey consumed by large zooplankton could also be consumed by fish: rotifers are too small for all except the smallest fish to capture. Predatory zooplankton play an important role in capturing this energy and passing it up the pelagic food chain. Size of prey also affects the efficiency of energy transfer: it is more efficient for a predator to feed on larger prey (Borgmann 1982). For these reasons, the consumption and dissipation of energy by large predatory zooplankton is to some extent balanced by their role in capturing 'small-sized zooplankton' production and by their greater value as food for planktivorous fish.

In Lake Erie the ratio of predatory zooplankton biomass to total macrozooplankton biomass (cladocerans + copepods) ranged from 0.29 to 0.41 in the eastern and west-central basins and 0.09 to 0.14 in the western basin (Table 13.7) (D. Graham, O. Johannsson and J. Dahl, DFO, Burlington, unpubl. data).   Using the same definition of predators, Yan (1986) found that a set of 13 lakes from Northern Ontario with a similar range of total phosphorus concentration (5 to 18 $\mu g.L^{-1}$) had an average ratio of 0.26. The ratio of predators to total zooplankton biomass may be on the high side in the eastern and west-central basins, which would decrease energy flow to fish in these areas.

Most of the predatory zooplankton are not obligate carnivores, but omnivores which also consume algae and detritus. If we assume that they are predatory only 50% of the time and include rotifer biomass in the ratio, the ratios decrease to 0.15 to 0.23 in the eastern and west-central basins and 0.05 to 0.07 in the western basin. More comparative data are needed to assess the relationships between these ratios and energy transfer efficiency.

13.5.6 Community Function:

13.5.6.1 Ratio of Production/Biomass

A comparison of epilimnetic, seasonal-mean, daily P/B ratios for some of the dominant cladoceran species from the Lake Erie Biomonitoring sites in 1993 and 1994 (described above) with 13 years of epilimnetic data from the two Bioindex monitoring sites on Lake Ontario indicate that values from Lake Erie are often in the range observed for Lake Ontario (O. Johannsson, D. Graham and J. Dahl, DFO, Burlington, unpubl. data) (Table 13.8). On two occasions daily P/B ratios were higher in Lake Erie than the mean ± 1 s.d. for the species in Lake Ontario. *Bosmina*, a dominant summer cladoceran species, had low daily P/B ratios in the offshore of the west-central basin (station WC2) and in the nearshore of the eastern basin (stations E1, E3). *Daphnia* were not present in sufficient numbers in the nearshore regions of the west-central and eastern basins to determine production using the egg-ratio method: the P/B ratio in the nearshore of the western basin was low. In the offshore, the P/B ratio of *Daphnia* was high in the western basin, normal in the eastern basin and very low in the west-central basin compared with Lake Ontario values. Lower values suggest greater stress on the populations in these regions. Because production was calculated using the egg ratio method, low values indicate a low production of eggs. That can result from a reduced food supply, poor environmental conditions or more intense planktivory on the larger, egg-bearing females.

13.5.6.2 Nutrient Changes/Food Resources

Lake Erie has passed through a period of increasing and then decreasing nutrient loads, the height of eutrophication occurring in the late 1960's and early 1970's. With increasing eutrophication, the proportion of blue green algae, which is generally inedible, increases in abundance relative to other more edible algae. By 1993, blue greens constituted only 0.1% to

3.5% of algal biomass (May-October mean) (Dahl et al. 1995). Growth of zooplankton would not be limited by these quantities of inedible algae. In 1994, 1995 and 1996, *Microcystis*, a large colonial blue green became abundant in the western basin of Lake Erie and other warm shallow areas in the Great Lakes basin. Whether *Microcystis* was more abundant than normal (see section 13.4.5.1) and whether it reduced zooplankton growth or production are not known. However, the possibility of negative impacts on the zooplankton community is real and should be investigated.

13.5.6.3 Energy Sinks

If a species is not readily eaten by predators it can become an energy sink. Under some conditions, if its population is not controlled*, Bythotrephes* can disrupt zooplankton community structure and function (e.g. Harp Lake, Yan and Pawson 1997), and become an energy sink. *Bythotrephes* has a long, spined tail which protects it from predation by small fish. Most Great Lakes' fish species do not eat *Bythotrephes* until they are >70 mm in length, some >100 mm in length (Barnhisel 1991a,b; Barnhisel and Harvey 1995: Branstrator and Lehman 1996). Only young bloater, *Coregonus hoyi*, can eat *Bythotrephes* once the fish reach 30 mm in length (Branstrator and Lehman 1996). Of the older, planktivorous fish which can eat *Bythotrephes,* only adult alewife have a spatial distribution which matches that of *Bythotrephes* both horizontally and vertically, and thus have a greater potential for controlling *Bythotrephes* than other fish species: both inhabit the offshore epilimnion and metalimnion (Olson et al. 1988; Johannsson and O'Gorman 1991; Lehman 1987, 1991). It is noteworthy, that in Lake Ontario where alewife are the dominant planktivore, *Bythotrephes* has been excluded (Makarewicz and Jones 1990, Johannsson et al. 1991).

To function as an energy sink, *Bythotrephes* would have to consume a large proportion of the zooplankton production that would normally go to fish. In addition, a significant proportion of the *Bythotrephes* population would have to go uneaten. *Bythotrephes* feed selectively on small cladocerans such as *Eubosmina, Bosmina* and *Daphnia retrocurva*, and the rotifer *Asplanchna* in preference to copepod nauplii or copepodids (Vanderploeg et al. 1993).   However, when cladocerans are rare, they do take copepods (Vanderploeg et al. 1993). Shulz and Yurista (1995) also found evidence of feeding on diaptomids, *Epischura* and copepod nauplii as well as *D. g*.

mendotae in wild populations from Lake Michigan. Johannsson and Graham (DFO, unpubl. data) compared the consumption of zooplankton by *Bythotrephes* to zooplankton production, that is, the production of the macrozooplankton (not including that of *Leptodora* and *Bythotrephes)* plus the production of *Asplanchna,* in Lake Erie in 1993 and 1994. *Bythotrephes* consumed nothing in the western basin, 6% - 12% of production in the eastern basin and 31% - 64% of production in the west-central basin. It would appear that under some conditions a significant portion of zooplankton production was being consumed by *Bythotrephes* in Lake Erie. The frequency, extent, and therefore, importance of these occurrences are not known. . We need to know whether this is a widespread phenomenon and whether this energy reaches fish in spite of the relatively inedible nature of *Bythotrephes.*

If Bythotrephes really is acting as an energy sink in Lake Erie, this may be a transitory phase. The European experience suggests that fish may evolve (learn) to cope with *Bythotrephes*. In Lake Vanern Sweden, smelt (*Osmerus albula*), cisco (*Coregonus albula*), roach (*Rutilus rutilus*) and whitefish (*Coregonus spp*), contained significant proportions of *B. cederstroemi* in their diets (Nilsson 1979).

13.5.7 Nearshore Regions

As noted by Fish (1960) and Krieger and Klarer (1991), zooplankton of coastal wetlands are not an extension of the lake populations and we can not extrapolate changes in the main lake to these areas. The U.S. Fish and Wildlife Service did a nearshore (<1 m water depth) survey for zooplankton in 1929 and 1930. This type of survey has not been repeated since that time (Krieger and Klarer, 1991).

Burns (1985) reconstructed the history of Lake Erie, describing the original conditions of the Sandusky basin and subsequent alterations to the land and water, a process that was likely repeated through out the basin. The water in the streams was originally clear and the bottom substrates were sandy or rocky. With deforestation and cultivation, the silt load in the streams increased. This, together with the action of carp made the water turbid, destroying much of the submerged vegetation. Many of the near-shore zooplankton species are associated with these

vegetated areas. To the extent that these types of habitats have disappeared from the margins of the lake, the abundance of these species will also have declined.

In studies of Old Woman Creek, Krieger and Klarer (1991) found that the environment was degraded and that this had a negative effect on zooplankton populations. Deforestation and cultivation decrease water retention by the land, increasing the water flow in streams after heavy storms. Storm-water, run-off events wash zooplankton out into the lake depleting the marsh populations (Krieger and Klarer 1991). These populations take many weeks to recover if they do recover that season. Zooplankton are a key component to the diet of many small fish. The depletion of the zooplankton populations undoubtedly has ramifications through the food chain.

13.5.8 Conclusions

13.5.8.1 Possible Impairments

1.   The eastern basin has lost the dominance of its original cold-water zooplankton species, *Mysis relicta, Limnocalanus macrurus* and *Leptodiaptomus sicilis*, apparently through increased planktivory and perhaps unfavourable oxygen conditions. Present oxygen concentrations are not a problem, but planktivory is still high.

2.   The mean size of zooplankton in the eastern basin indicates that planktivory is a strong structuring force in this region.   The equally small size of zooplankton in the west-central basin, when compared with 110-μ mesh samples, suggests that planktivory may also be high in this region. Small zooplankton mean size indicates a possible impairment of transfer of biomass up the food chain

3.   The high ratio of predatory zooplankton biomass to total zooplankton biomass may indicate a high level of planktivory but could also be associated with an energy sink. In 1993, *Bythotrephes* appeared to act as an energy sink in the west-central basin. This is an impairment. Unfortunately, there is a balance between the number and type of

planktivores and the number of *Bythotrephes*. What conditions are optimal for zooplankton and energy transfer up the food chain in this new community of zooplankton still need to be determined.

4.   Changes in nearshore habitat have adversely affected zooplankton in estuaries.

13.8.2 Other Conditions

5.   Based on mean individual size and predator to total zooplankton biomass the zooplankton communities in the offshore of the west basin appear to be healthy. The mean size at the nearshore station was smaller but not as small as in the eastern basin. As these data are based on 110-μ mesh samples we can not compare them directly to Mill's et al. (1987) 'preferred size'. Judgment must be reserved on the inshore west basin community.

6.   The ratio of cladocerans to copepods has changed with the invasion of zebra mussels. The use of the calanoid/(cladoceran+cyclopoid) ratio may no longer be comparable with past ratios to follow trends in eutrophy.

7.   Species richness has increased with the invasion of exotics. As the role of the exotic is not always favourable, species richness, per se, is not a good indicator of health.

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Table 13.1 Historical changes in total phytoplankton biomass (mm$^{3\cdot}$ L^{-1}) in Lake Erie.

Study	Years	Basin			Comments
		West	Central	East	
Munawar & Munawar 1976, table 7.	1970	5.30	3.20	2.40	Seasonal and areal mean from ten spatial surveys.
Gladish & Munawar 1980	1975-76	2.50			Two nearshore stations in northern west basin.
Hopkins & Lea 1982	1969-78			0.12	Seven nearshore stations. A.S.U. Conversion 10 yr. mean, range 0.052-0.233.
Nicholls, Standen, & Hopkins 1980	1967-70 1971-76 1978	2.33 1.33 2.08			Union water intake 450 m offshore. A.S.U. conversion.
Nicholls, Taylor, & Hamdy 1983	1979			0.45-7.60	Three stations in mouth of Grand R. Six stations 2-4 km offshore.
Nicholls & Hopkins 1993	1987-88	0.26	0.15	0.07	Nearshore water intakes. Pre-zebra mussels conversion from A.S.U.
Nicholls & Hopkins 1993	1989-90	*	*	*	Post-zebra mussels conversion from A.S.U.
Makarewicz 1993a	1983-87	1.88	1.04	0.63	Offshore cruises spring-summer
Makarewicz 1993a	1983-85	1.65	1.30	0.68	Apr-Nov interpolated from fig. 7a

Table 13.1 Historical changes in total phytoplankton biomass (mm$^{3\cdot}$ L^{-1}) in Lake Erie.

		Basin			
Study	Years	West	Central	East	Comments
Dahl et al. 1995	1993	1.02	0.91	0.51	spring-summer
		0.76	0.75	0.35	May-Oct

* Conversion results in zero biomass because of very low A.S.U. values and negative intercept in the predictive equation (Nicholls 1981).

Table 13.2. Maximum phytoplankton biomass (mm$^{3\cdot}$L^{-1}) in Lake Erie.

Study	Years	Basin West	Central	East	Comments
Munawar & Munawar 1976	1970	13.0		4.0	
Gladish & Munawar 1980	1975-76	9.9			
Hopkins & Lea 1982	1970			0.6	A.S.U. conversion
	1978			0.5	A.S.U. conversion
Makarewicz 1993a	1970-87	6.6	8.8	3.4	
Dahl et al. 1995	1993	2.7	3.6	0.6	

Table 13.3. Historical changes in chlorophyll (mg×m^{-3}) in Lake Erie.

Study	Years	West	Central	East	Comments
Glooschenko et al. 1974.	1970	8.9	4.4	3.2	Apr-Dec basin means
Glooschenko et al. 1974	1970	3.3-19.3	2.5-9.2	1.4-5.4	Range of basin means Apr-Dec
Fay & Rathke 1980	1973	13.4	5.3		EPA Report. Table 62. volume-weighted average June-Nov
	1974	14.6	4.8		
	1975	15.2	6.6		
Wallen & Botek 1984.	1978	3.8-15.1			Range of monthly means for northern and southern stations
Nicholls and Hopkins 1993	1988-89	<1.0			Union Water Intake Station
Millard unpublished	June 1993	4.7	1.7	0.8	Basin area-weighted means*
Charlton 1994.	Aug 1993	5.1	2.7	2.2	Arithmetic means

*Data from Lake Erie Trophic Transfer Study. Data from each station were weighted by a representative area based on a nearest neighbours analysis between all stations.

Table 13.3. Cont'd.

Study	Years	West	Central	East	Comments
Millard unpublished	Sept 1993	6.7	4.5	0.4	Basin area-weighted means*
Dahl et al. 1995	1993	3.5-4.5	2.7-4.0	1.1-2.1	May-Oct mean
Millard unpublished	May 1994	5.2	1.8	1.6	Basin area-weighted
	Aug 1994	5.3	3.9	2.4	means*
	Sept 1994	6.0	4.5	1.8	

Table 13.4. Changes in the trophic classification* of Lake Erie basins based on total phytoplankton biomass using the classification of Munawar and Munawar (1982)

Basin	Munawar & Munawar 1976	Makarewicz 1993a	Dahl et al. 1995
West	eutrophic	mesotrophic	oligotrophic
Central	meso-eutrophic	mesotrophic	oligotrophic
East	meso-eutrophic	oligotrophic	ultra-oligotrophic

* Classification is for pelagia only.

Table 13.5. Predictions of seasonal mean chlorophyll (ug·L^{-1}) from total phosphorus (TP ug·L^{-1}) using various equations from the literature, Smith (1982), Prepas and Trew (1983), Mazumder (1994). Other column headings are defined as follows: mean summer total phosphorus (TP$_{sum}$), total nitrogen (TN), observed mean seasonal chlorophyll (Chl$_{obs}$), observed mean summer chlorophyll (Chl$_{sum}$).

Stn	TP	TP$_{sum}$	TN	Chl$_{obs}$	Chl$_{sum}$	Smith	Prepas &Trew	Mazumder even link
1993								
E1	7.8	7.7	468	0.97	1.20	2.20	2.26	1.72
E2	9.6	7.5	457	2.00	1.97	2.33	2.20	1.96
E3	6.7	5.7	473	1.16	1.19	2.10	1.60	1.58
WC1	11.2	8.6	470	2.75	2.73	2.53	2.57	2.17
WC2	12.6	9.9	475	4.33	3.36	2.67	3.00	2.35
W1	17.2	14.7	631	4.53	4.74	3.91	4.75	2.49
W2	15.4	14.0	598	3.55	4.13	3.57	4.49	2.10
W3	18.4	19.8	649	4.53	4.27	4.12	6.68	2.76
1994								
E1	10.1	9.1	502	1.54	1.20	3.87	2.72	2.02
E2	8.1	7.8	552	2.24	1.79	2.60	2.31	1.76
E3	8.5	8.1	480	1.27	1.28	2.33	2.41	1.82

See Dahl et al. (1995) for specific station locations. E=east basin, WC=west central basin,

W=west basin

Table. 13.6. Ratio of predicted:observed seasonal areal phytoplankton photosynthesis in Lake Erie.

Basin & Stn	Ratio
1993	
West stn 1	1.2
West stn 2	1.4
West Central stn 1	1.0
West Central stn 2	0.9
East stn 2	1.3
East stn 3	1.5
1994	
East stn 1*	1.8
East stn 2	1.4
East stn 3*	1.7

*Corrections for shallow depth and non-algal effects
 on transparency were applied to observed PP.
 See text for detailed explanation

Table 13.7. Ratios of predatory to total zooplankton biomass (Maximum Ratio) or to total crustacean plus rotifer biomass (More Realistic Ratio) in the three basins of Lake Erie in 1993 and 1994. Copepod carnivores are omnivorous and assumed to be 100% predatory in the first ratio and 50% predatory in the second ratio.

Station/Year	Maximum Predator Biomass		More Realistic Predator Biomass	
	1993	1994	1993	1994
E1	0.39	0.40	0.20	0.21
E2	0.30	0.38	0.16	0.20
	0.39	0.29	0.20	0.15
WC1	0.32		0.17	
WC2	0.41		0.23	
W1	0.14		0.07	
W3	0.09		0.05	

Data from O. Johannsson, D. Graham, and J. Dahl, DFO, Canada, Burlington

Table 13.8. Ratios of seasonal-mean, daily production/biomass for dominant zooplankton species in Lakes Ontario and Erie. The mean ± 1 S.D. is given for the Lake Ontario data which comes from a 13 year monitoring program at a midlake (41) and eastern basin (81) station (O. Johannsson, DFO, Burlington, unpubl. data). Samples were collected through the epilimnion or top 20 m during unstratified conditions. Ontario samples were unmetered. Data were calculated over the April (Ontario) or May (Erie) to October period and production calculated using the egg ratio method. Data are presented for 1993 and 1994 for the eastern basin sites in Lake Erie.

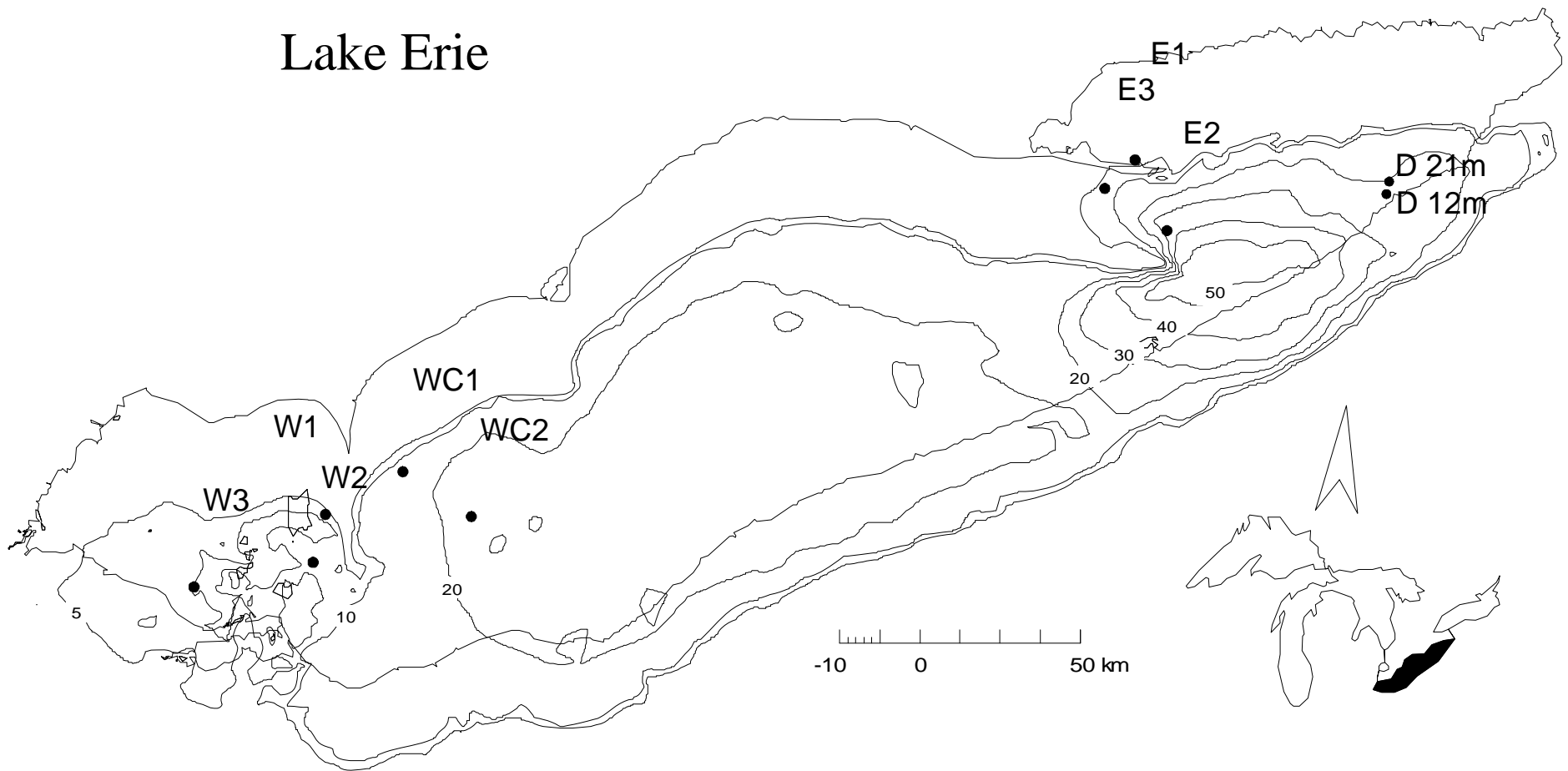| Species	Lake Ontario		Lake Erie						
	Stn 41	Stn 81	W1	W3	WC1	WC2	E1	E3	E2
Bosmina	0.080± 0.028	0.096± 0.032	0.166	0.101	0.081	0.058	0.060 --	0.119 0.050	0.089 0.135
Eubosmina	0.095± 0.047	0.117± 0.028	0.045	--	0.103	0.066	-- --	-- --	-- --
Daphnia	0.088± 0.044	0.087± 0.022	0.053	0.202	--	0.015	-- --	-- --	0.089 0.095

Fig. 13.1 Map of Lake Erie showing depth contours(m) and the location of zooplankton sampling site in the 1990s. New York Department of Environmental Conservation (1986-1996) D 21m and D 12m; Ontario Ministry of Natural Resources (1988-1990) W1, W2; Fisheries and Oceans Canada (1993-1994) W1, W3, WC1, WC2, E1, E2, E3.
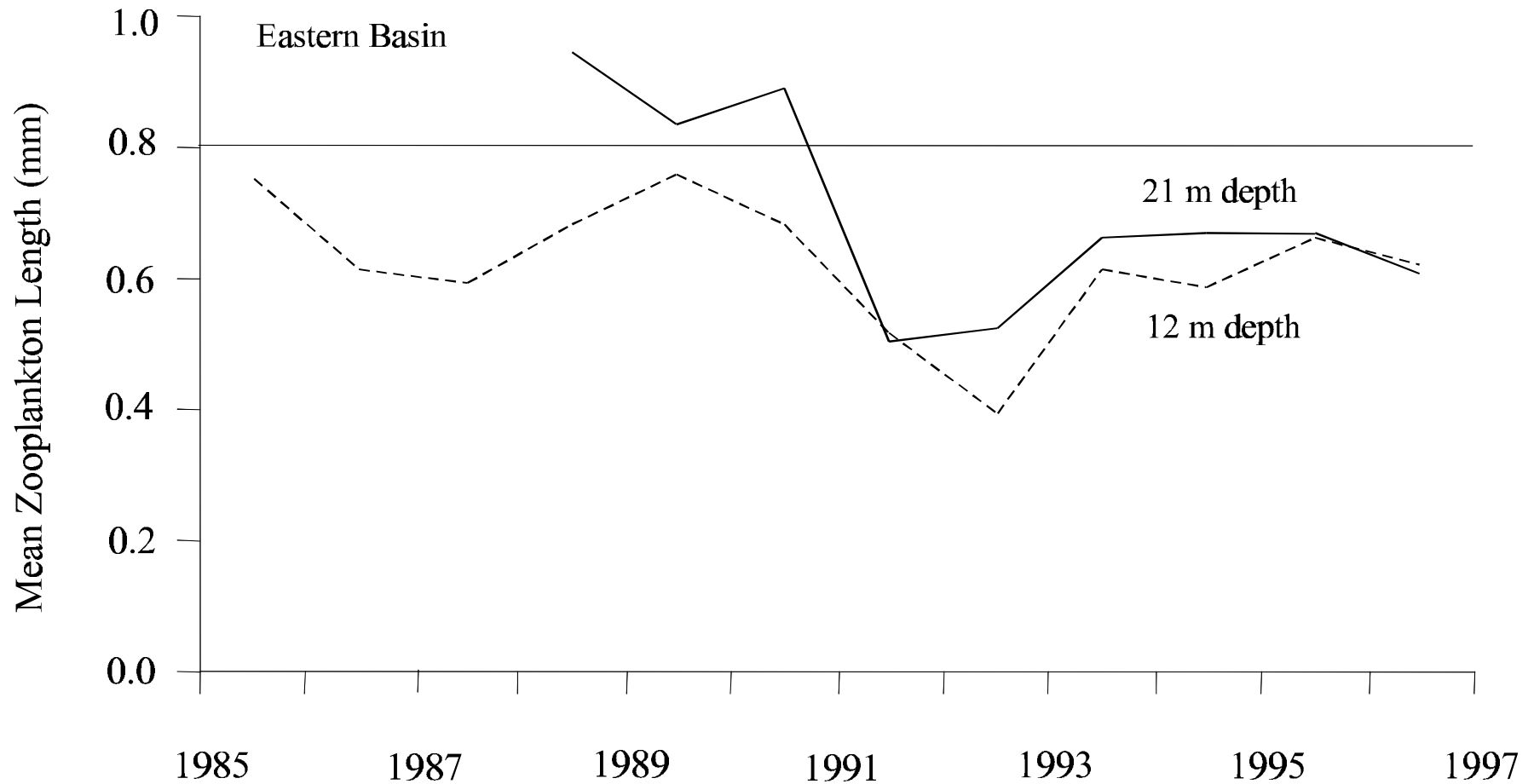
Fig. 13.2a. Historical trends in the average June 24th - September 7th mean length of zooplankton period in the eastern basin of Lake Erie. Data were collected with a 153-u mesh net at sites 21 m and 12 m deep off of Dunkirk by the New York Department of Environmental Conservation (Mr. Don Einhouse and Mr. Bill Culligan, unpublished data). 0.8 is the optimum mean length (Mills et al. 1987)
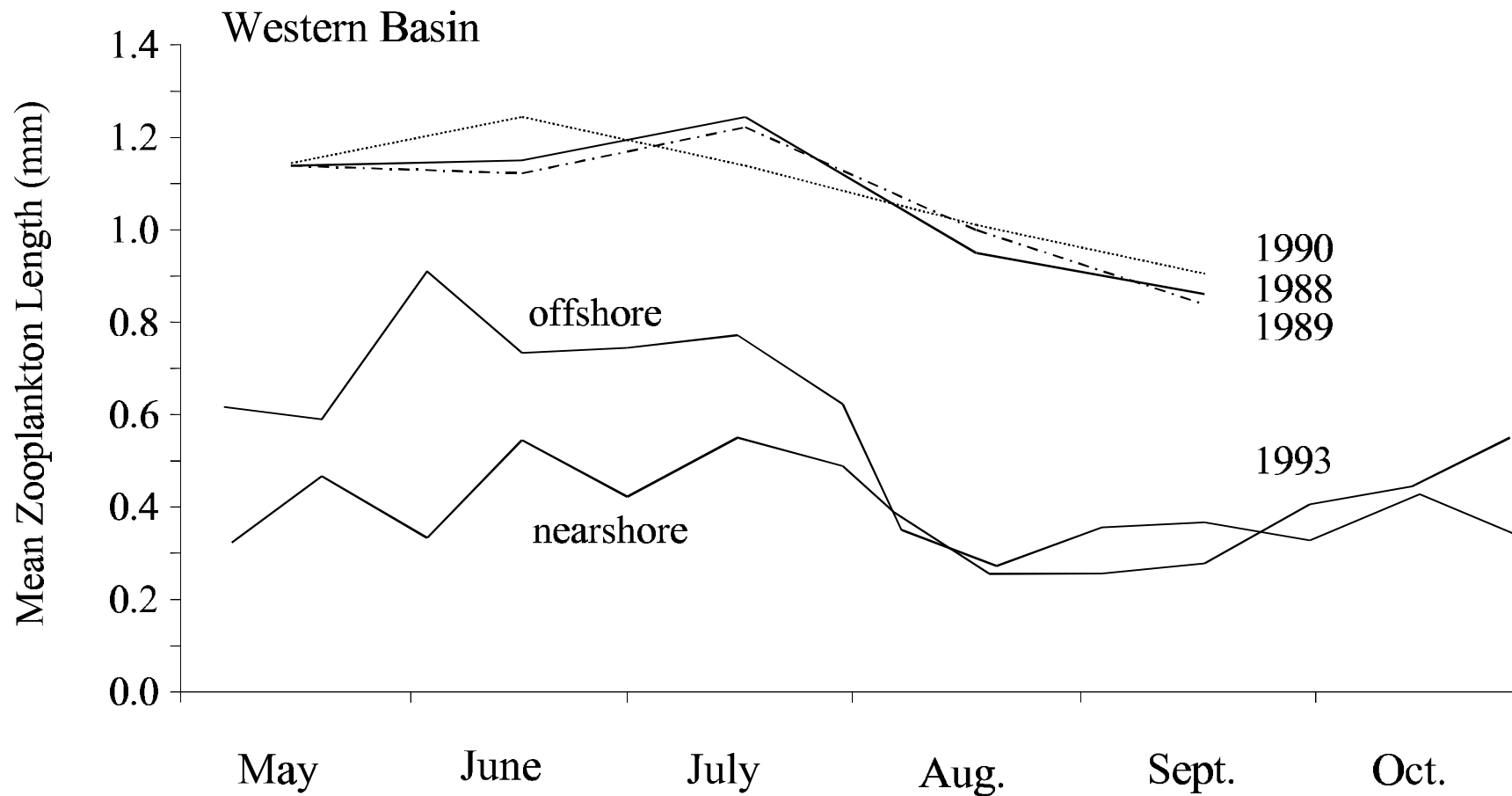
Fig. 13.2b.. Mean length of zooplankton in the western basin of Lake Erie. 1988-1990 data were collected at sites W1 and W2 with a 64-u mesh net by the Ontario Ministry of Natural Resources (Dr. J. Leach, unpublished data); 1993 data were collected at sites W1 and W3 with a 110-u mesh net (Dahl et al. 1995).
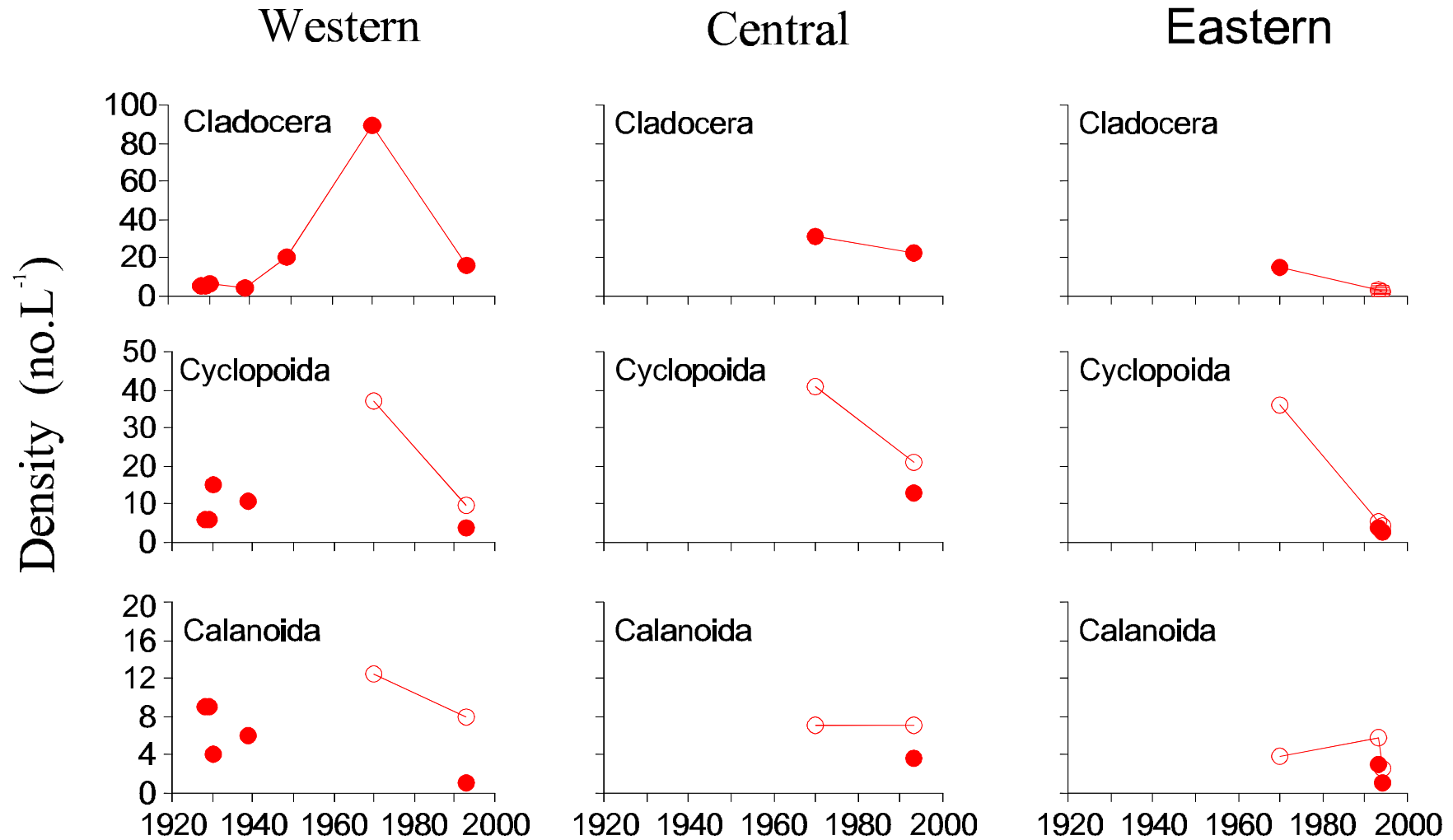
Fig. 13.3 Changes in the abundance of macrozooplankton groups in the three basins of Lake Erie. Abundance was averaged over the May - October season. 1928-30 data from, Wright (1955), 1938-39 data from Chandler (1940), 1949 data from Bradshaw (1964), 1970 data from Watson (1976), 1983-87 data from Makarewicz (1993b) 1993-94 data from O.. Johannsson, D. Graham and J. Dahl (Fisheries and Oceans Canada, unpublished data).
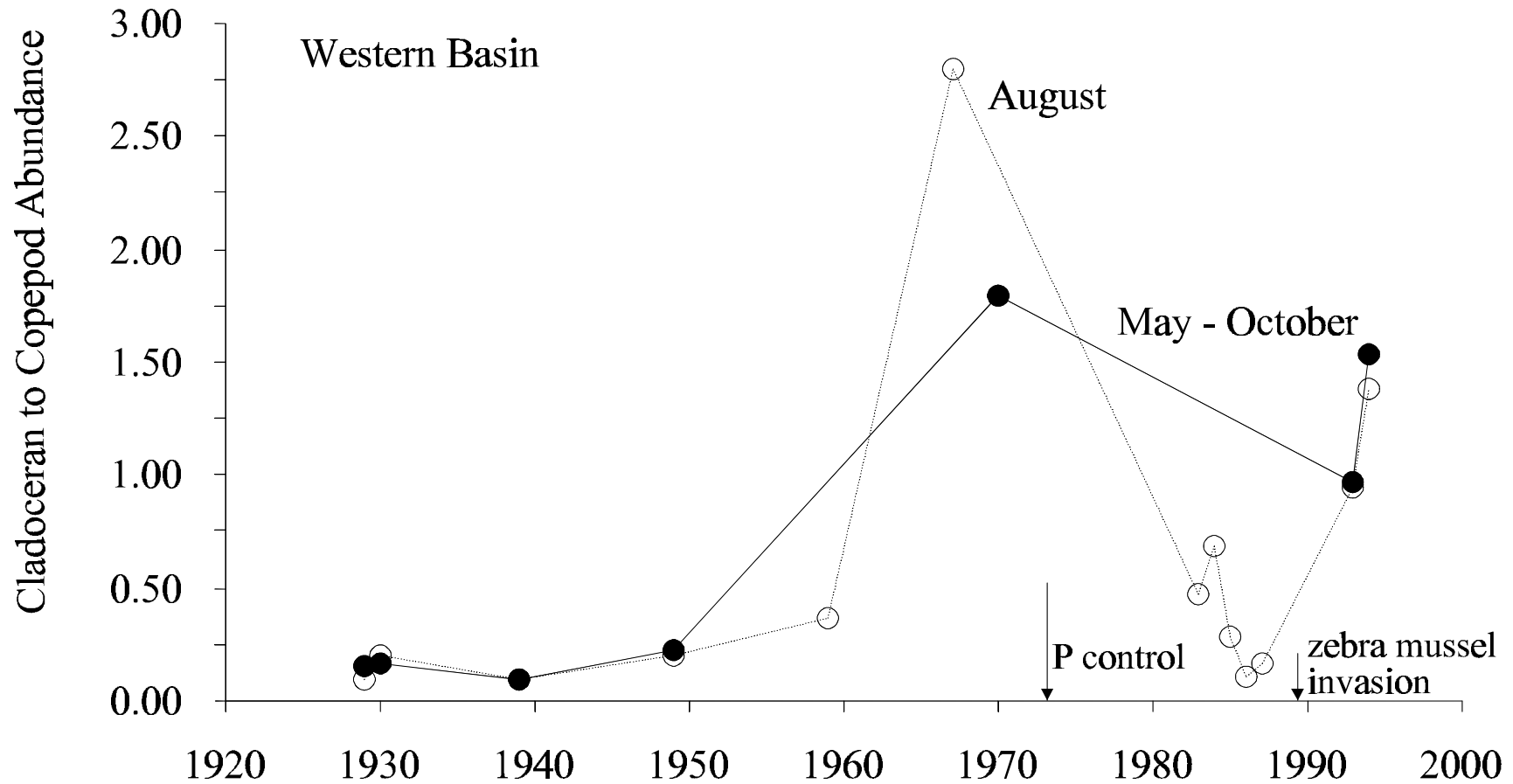
Fig. 13.4. Historical changes in the ratio of cladoceran to copepod abundances in the western basin of Lake Erie. May to October averaged data, and July 27 to August 25 averaged data. 1928-30 data from, Wright (1955), 1938-39 data from Chandler (1940), 1949 data from Bradshaw (1964), 1970 data from Watson (1976), 1983-87 data from Makarewicz (1993b), 1988-94 data, 1993-94 data from O. Johannsson, D. Graham and J. Dahl (Fisheries and Oceans Canada, unpublished data).