J. Great Lakes Res. 31 (Supplement 2):35–49
Internat. Assoc. Great Lakes Res., 2005

Trends in Temperature, Secchi Depth, and Dissolved Oxygen Depletion Rates in the Central Basin of Lake Erie, 1983–2002

Noel M. Burns[1,*], David C. Rockwell[2], Paul E. Bertram[2], David M. Dolan[3], and Jan J.H. Ciborowski[4]

[1]*Lakes Consulting*
42 Seabreeze Rd.
Devonport, New Zealand 1309

[2]*Great Lakes National Program Office*
United States Environmental Protection Agency
77 West Jackson Blvd.
Chicago, Illinois 60604

[3]*Natural and Applied Sciences*
University of Wisconsin – Green Bay
Green Bay, Wisconsin 54311

[4]*Department of Biological Sciences and Great Lakes Institute for Environmental Research*
University of Windsor
Windsor, Ontario N9B 3P4

ABSTRACT. *We examined temperature trends in a 20-year set of monitoring records collected at multiple deep-water stations in the central basin of Lake Erie. Data collected were statistically corrected ("deseasonalized") to remove biases resulting from irregular sampling intervals within years. Depth-integrated summer temperature has increased by an average (±SE) of 0.037 ± 0.01°C per year. An observed reduction of Secchi depth (SD) by 7 ± 3 cm/y seems to be unrelated to variation in either total phosphorus (TP) or chlorophyll* a *concentrations. Midsummer midbasin SD values varied widely between 4 and 10 m, possibly depending on whether phytoplankton were concentrated in the epilimnion (giving shallow SD), or whether phytoplankton had settled out of the epilimnion into the lower layers, giving deeper SD values. Hypolimnetic volume-corrected oxygen depletion (HVOD) rates have also been highly variable, ranging from 2.68 to 4.72 mg/L/mo. These rates are sensitive to production of oxygen in the thermocline and hypolimnion by photosynthetically active phytoplankton that have settled from the epilimnion. The HVOD rate in any year was correlated with the previous year's TP loading into Lake Erie. Since TP loading trends largely reflect the consequences of improving water treatment through the 1980s and increasing contributions from tributary run-off sediments through the 1990s, there is little direct evidence to suggest that the appearance of dreissenids has directly influenced hypolimnial oxygen depletion rates in the central basin. The observation that central-basin HVOD tracked the reductions in TP loadings through the 1980s may be the first affirmation that central basin hypolimnetic oxygen dynamics can be regulated by phosphorus inputs. This implies that TP loads must continue to be regulated if we wish to minimize oxygen depletion rates as a strategy to reduce the frequency of episodic central basin anoxia.*

INDEX WORDS: *Lake Erie, phosphorus, turbidity, chlorophyll a, oxygen deficit, Dreissenidae.*

INTRODUCTION

Lake Erie is the most rapidly changing and perhaps most intensely studied of the Great Lakes. Because almost 7 million people live on or near the shores, it has suffered the greatest amount of human-related perturbation. But it has also shown the greatest amount of change coincident with restoration efforts. Historical concerns about the environmental conditions of the lake have produced a long and detailed data set relating water quality

[*]Corresponding author. E-mail: lakescon@xtra.co.nz

conditions to human activity. Cultural eutrophication reached its greatest impacts in the early 1970s (Burns and Ross 1972), prompting binational efforts to control inputs of phosphorus. Implementation of the Great Lakes Water Quality Agreement (International Joint Commission 1978) included actions on the part of the U.S. and Canada to reduce and limit annual total phosphorus (TP) loadings to 11,000 tonnes/y or less.

In order to measure the potential effectiveness of nutrient controls on water quality, the National Water Research Institute (NWRI) of Environment Canada began annual monitoring in 1977 (Charlton *et al.* 1999). The U.S. EPA Great Lakes National Program Office (GLNPO) monitoring program for Lake Erie began in 1983 to provide data for numerical, nutrient-based eutrophication models (Chapra 1977, Lesht 1985, DiToro and Connolly 1980). The resulting time trends in water quality have been reported several times at irregular intervals (Rathke and Edwards 1985, Makarewicz and Bertram 1991, Charlton *et al.* 1999, Makarewicz *et al.* 2000). Loadings of total phosphorus to Lake Erie had reached target levels of 11,000 tonnes/y by the late 1980s and this appeared to be reflected by improving water quality conditions (Makarewicz and Bertram 1991). This transition was possibly facilitated by the establishment of dreissenid mussels whose filter feeding capabilities were credited with reducing chlorophyll *a* levels and increasing water clarity in various shallow Great Lakes locations (Nicholls and Hopkins 1993, Fahnenstiel *et al.* 1995, Howell *et al.* 1996, Vanderploeg *et al.* 2001). Annual monitoring in Lake Erie by GLNPO was suspended for several years in the mid 1990s. The discovery in the late 1990s of an increasing frequency of central basin episodic anoxia prompted a resurgence of interest in water quality, and ultimately prompted the Lake Erie Trophic Status collaborative study in 2002. However, understanding of current conditions at that time was hampered by the lack of a complete record of nutrient loadings and measures of water quality.

This study uses the GLNPO data collected during annual spring and summer cruises between 1983–2002 to assess trends in water temperature, clarity (Secchi depth), dissolved oxygen concentration, and derived estimates of hypolimnetic volume-corrected oxygen depletion in the central basin of Lake Erie. We evaluated patterns of change over the entire period of record by supplementing years with missing information with data collected by NWRI. Additional objectives were to ascertain whether trends were present during the intervals between the beginning of the period of record and the appearance of dreissenids in the central basin (1983–1989), and the time subsequent to that (1990–2002). Although there is much documentation of local effects of dreissenids in shallow waters, it is unclear whether dreissenid populations have the capacity to influence water clarity and nutrient dynamics at the scale of an entire Great Lakes basin.

METHODS AND MATERIALS

Sources of Data

The Great Lakes National Program Office (GLNPO) of the U.S. Environmental Protection Agency has collected physicochemical data from up to 11 stations in central Lake Erie (Fig. 1; nominal depth 21–24 m) several times per year from 1983 to 2002. In this paper, we report on water temperature, Secchi depth (SD), and dissolved oxygen concentration (DO) data analyzed for the detection of trends with time (Burns and Ross 2002). Concurrently-collected information on chlorophyll *a* (Chl *a*) and total phosphorus (TP) are analyzed and interpreted by Rockwell *et al.* (2005). The trends are derived from plots of all the data recorded for the epilimnion and isothermal layers for SD and temperature after deseasonalizing (see below). The computer program, LakeWatch (Lakes Consulting 2000a) was used to determine the trends. This program enables the trends to be calculated using each datum value separately, or as averages of all the values measured on a particular day, week, or month. This reduces bias introduced between periods of frequent sampling or numerous stations sampled as against periods of infrequent sampling or when only a few stations were sampled in a period. Trends have been calculated using these four procedures and compared.

Although GLNPO did not sample Lake Erie during 1994 and 1995, up to seven additional DO monitoring surveys for the years 1991–93 and 1997–2001 were conducted. The National Water Research Institute (NWRI) of Environment Canada undertook monthly DO monitoring surveys at 12 central basin stations for the years 1994-2001 (Fig. 1). Thus, the Environment Canada data span the 1994–96 gap in the GLNPO data and provide information comparable with that of GLNPO for the years 1997–2000.

Normalized rates of dissolved oxygen depletion were calculated for all years according to the

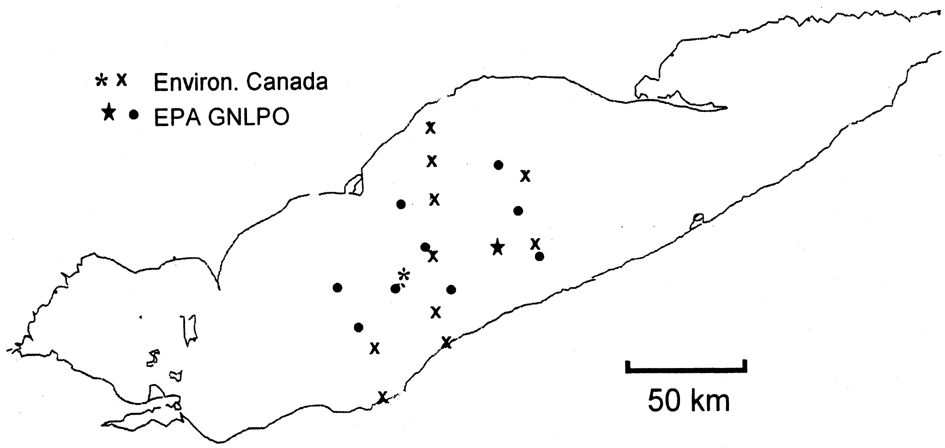

FIG. 1. Map of Lake Erie showing location of EPA GLNPO monitoring stations (filled circles and star) and Environment Canada stations (x's and asterisk). Station marked by the asterisk is Environment Canada station 984. GLNPO station 78 is indicated by a star.

method of Rosa and Burns (1987). The values reported here for 1970–1980 were taken from Burns and Ross (2002), and for 1981–1986 from Rathke and McRae (1989), as reported in Bertram (1993). Data from 1987–2003 were collected and analyzed by GLNPO by the same methodology.

Because the Burns and Ross (2002) method for calculating DO depletion rates differs slightly from that of Rosa and Burns (1987), both sets of results are presented here for comparative purposes. The Rosa and Burns (1987) method accounts for changes in hypolimnetic properties on a survey interval basis whereas the Burns and Ross (2002) method uses an annual basis.

Sampling Details

The EPA central basin stations were sampled by GLNPO 2–3 times during each open water season. Samples were collected at minimum during the "spring" (isothermal) and "summer" (stratified) seasons. Station 78 (the master station, Fig. 1) was often sampled more intensively than the other stations. The sampling regime varied according to the thermal condition of the water column and the station being sampled.

Temperature

Vertical temperature profiles were determined from temperature measurements recorded 0.5 m apart from 1 m below the surface to 1 m above the bottom with a cable-mounted temperature probe accurate to the nearest 0.1°C. Meter-recorded surface water temperatures were always within 2% of standard mercury thermometer readings.

Secchi Depth

Secchi depth was measured to the nearest 10 cm at each sampling location with an all white 20-cm diameter Secchi disk lowered from the shaded side of the boat. Secchi depth was the mean of the depths at which the disk was last seen during lowering and first seen during raising.

Dissolved Oxygen

All stations were routinely sampled 1 m below the water surface and 1 m above the bottom on all dates. During spring, the master station was also sampled at depths of 5 and 10 m below the surface, whereas the other stations were sampled midway between the surface and the bottom. During summer, the depth of the thermocline at each station was determined from the vertical temperature profiles. Four additional master station samples were collected: from mid-epilimnion depth, 1 m above the upper knee of the thermocline (lower epilimnion), at the inflection point of the thermocline, and 1 m below the lower knee of the thermocline (upper hypolimnion). The same depths were sampled at non-master stations if a thermocline was present. If no thermocline was present, 5-m and 10-m depths were sampled.

Dissolved oxygen concentration was recorded by both EPA and Environment Canada using electronic probes attached to a shipboard meter with 95% C.I. precision of 10% when DO was > 5 mg/L, and within 0.5 mg/L when DO concentration was ≤ 5 mg/L. The accuracy of D.O. readings was frequently verified by shipboard Winkler titrations of water samples collected from the measured depth (U.S. EPA 2003).

Total Phosphorus and Chlorophyll a

The measurement systems for TP and Chl *a* changed during the monitoring period, with a slightly more sensitive method being used for the TP analyses after 1992. From 1983 to 1992, TP was estimated using the sulfuric acid and persulfate digestion, with a detection limit of 2 µg/L. The molybdate-ascorbic acid method was used from 1993 onward, providing a detection limit of 1 µg/L. Chlorophyll *a* concentration was determined by acetone extraction from 1983 to 1993. Thereafter, (1996–2002) it was measured directly by fluorometry. Detection limits for both methods were < 0.1 µg/L

Statistical Analysis—Trend Detection in the Physico-chemical Variables

All data were analyzed using "LakeWatch" software (Lakes Consulting 2000a), a program designed to analyze monitoring data from lakes and reservoirs. The methodology embodied in the program and used here is described in detail in Burns *et al.* (1999) and Lakes Consulting (2000b). Some of the methodology is described below. This software was used to determine progressive among-year changes in deseasonalized TP and Chl *a* (reported by Rockwell *et al.*, 2005, temperature, and SD data.

The trend values used in this study are those derived from all the observed epilimnetic and isothermal values. As a check on the stability of estimates resulting from different forms of averaging, long-term trend lines were estimated for patterns of Chl *a*, SD, TP, and temperature determined from individual values and for values averaged over each day, week, and month. The trend values were comparable for most of the modes of calculation except for some monthly averages, which produced consistently lower values and higher standard errors than estimates calculated for shorter intervals (Table 1).

The available data were first deseasonalized before trend analysis. The daily average epilimnetic temperatures observed during the 20 y of monitoring the central basin were plotted as a function of the time of year of collection only (Fig. 2A). A polynomial curve fitted to these data represents the annualized pattern of average temperatures, from which the expected temperature for each day of the year can be calculated. The difference between the temperature observed on any specific calendar date and the annualized value for that date yields a residual value for that particular day, which we term the deseasonalised value. The observed and residual data are plotted against time (see Fig. 3), and least squares straight line plots are fitted to the data. A *t*-test evaluates the null hypothesis that the slope of this line is zero. A low p-value means that there is a low probability that the fit of the line is attributable to chance. The slopes of both the lines fitted to the data in Figure 3A are significant ($p < 0.05$). Since the units along the x-axis in Figure 3 are years, the slope of the line estimates the change per year in the dependent variable, and the p-value gives the statistical significance level of the computed change.

Three trend analyses were performed. One analysis produced single trend lines for the entire period of record. The data sets were then split into

TABLE 1. Annual rates of change (±SE) of chlorophyll* a *(Chl a), Secchi depth (SD), total phosphorus concentration (TP), and epilimnetic temperature estimated from all observed data values, data averaged on a daily basis, data averaged on a weekly basis, and data averaged on a monthly basis.

Variable	Units	Trends using each observed data value	Trends using data as basin averages for a day	Trends using data as basin averages for a week	Trends using data as basin averages for a month
Chl a	$mg/m^3/y$	–0.05 ± 0.03	–0.05 ± 0.03	–0.04 ± 0.04	0.031 ± 0.04
SD	m/y	–0.07 ± 0.03	–0.07 ± 0.03	–0.07 ± 0.03	–0.04 ± 0.03
TP	$mg/m^3/y$	–0.07 ± 0.06	–0.06 ± 0.06	–0.07 ± 0.08	–0.11 ± 0.11
Temperature	°C	0.03 ± 0.01	0.03 ± 0.02	0.04 ± 0.02	0.03 ± 0.03

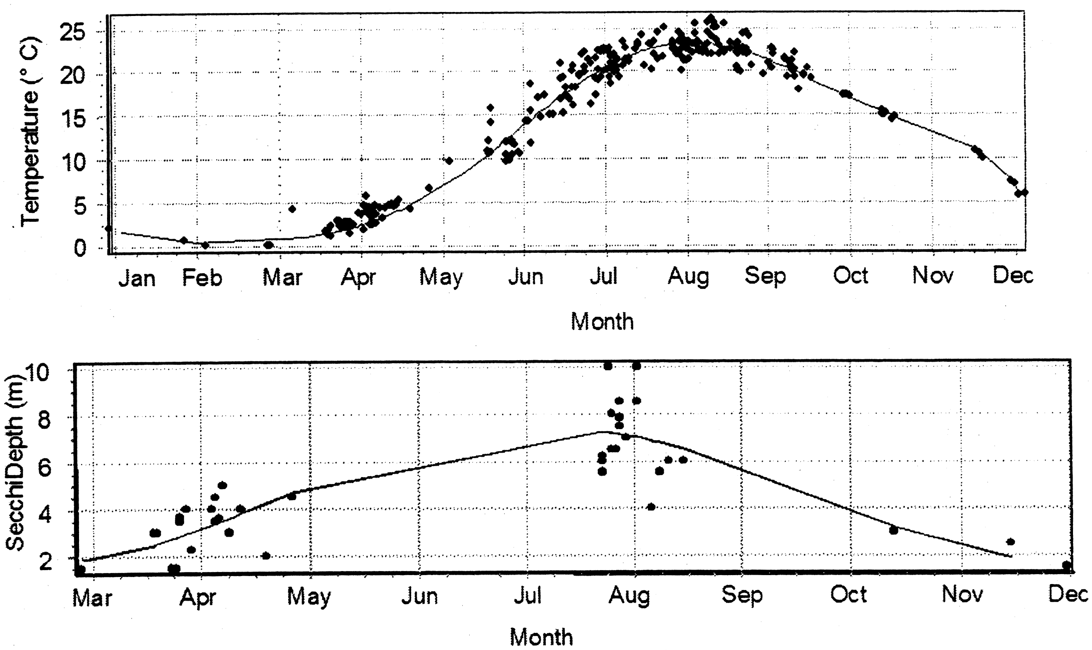

FIG. 2. Seasonal variation in epilimnetic/isothermal temperature (A), and Secchi depth (B) in the central basin of Lake Erie, 1983–2002. Temperature points represent measurements at all EPA GLNPO stations sampled. Secchi depth points are for EPA Station 78 only. Curves are fitted through data by polynomial regression.

1980–1989 and 1990–2002 periods, broadly corresponding to times prior to and following the widespread establishment of dreissenid mussels in the central basin of Lake Erie. Dolan and McGunagle (2005) observed a statistically significant breakpoint in time trends of loadings of total phosphorus to Lake Erie in 1990.

Hypolimnetic Volume-corrected Oxygen Depletion Rate (HVOD) Analysis

The DO data were used to determine the hypolimnetic volume-corrected oxygen depletion rate (HVOD) for each year in the manner described by Rosa and Burns (1987), except that all corrections were applied on an annual rather than on a sampling-interval basis. Each vertical DO/temperature profile was first examined to see whether it showed evidence of concentrations below 2.0 mg L^{-1}, hypolimnetic oxygen production, or massive reoxygenation attributable to vertical mixing. If the profile showed no evidence of any of these three perturbing effects, the depth separating the epilimnion from the thermocline was taken as the lower bound of the epilimnion for a sampling date. The depth separating the thermocline from the hypolimnion was used to demarcate the upper bound of the hypolimnion for a station on that date, and the data from that time point were included in the HVOD analysis for the year of its occurrence.

The LakeWatch software was used to plot all the hypolimnion DO and temperature data for each year to estimate the DO depletion rate (mg/m^3/d) and temperature increase rate (°C/d) on a daily basis by fitting regression lines to the data. A temperature correction factor was next applied to standardize the DO depletion rate to 10°C, (R_t) on the expectation that the rate would double for every 10°C rise in temperature (Rosa and Burns 1987). Then the value for the mean epilimnetic and hypolimnetic DO concentrations and temperatures for the stratified stations for each year were determined. The vertical mixing reoxygenation value was determined using eqn. 4 from Rosa and Burns (1987) namely;

$$\Delta O = (Oe - O_h) / (Te - Th) \times \Delta T \qquad (1)$$

where

ΔO = increase in DO concentrations if DO was conserved

ΔT = increase rate in hypolimnetic temperature

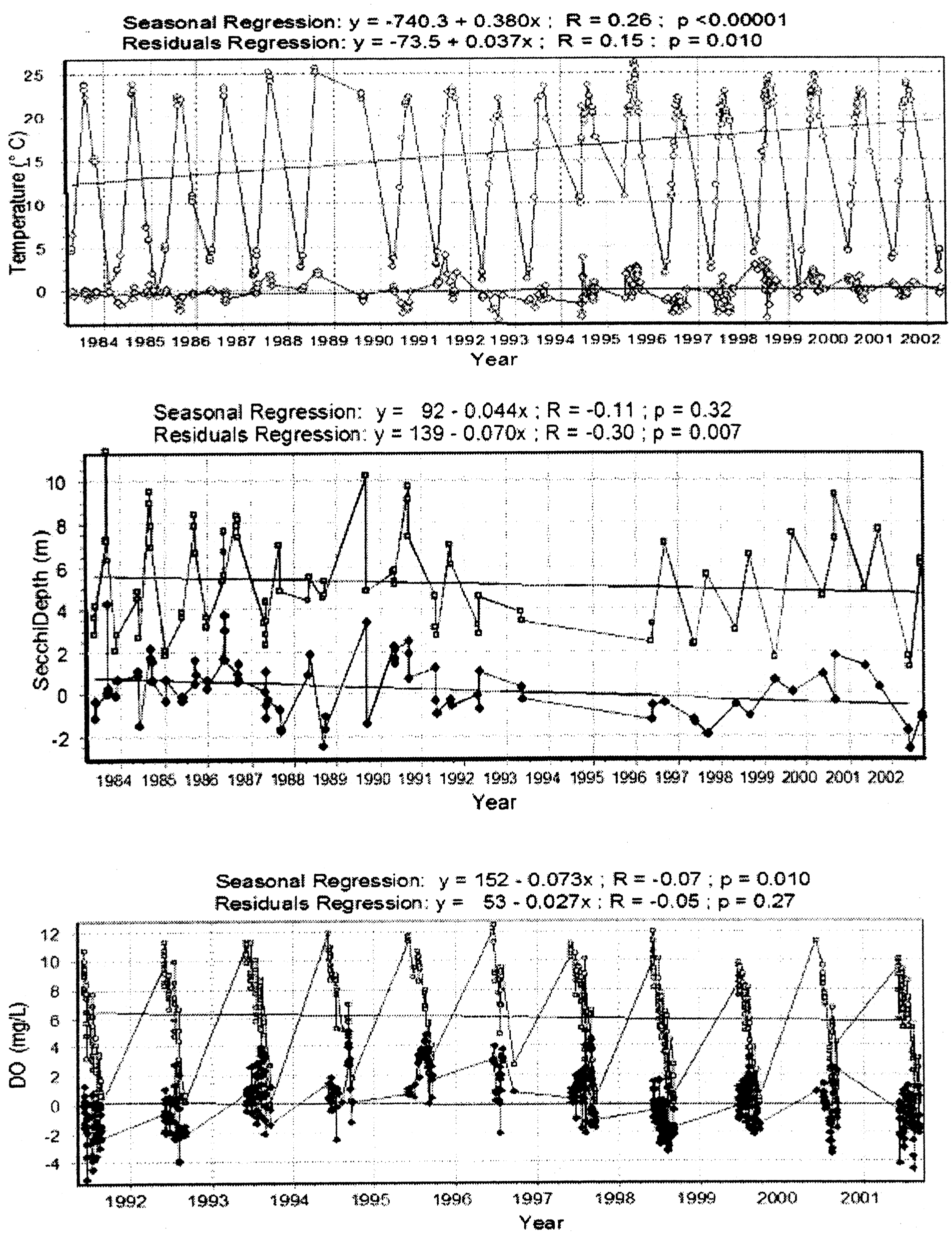

FIG. 3. Line plots of time trends of physicochemical parameters in the central basin of Lake Erie. The upper curve in each panel represents raw data. The lower curve represents residual values (observed value minus value predicted for the observation date by regression equations estimated as in Fig. 2). Straight lines drawn through the upper and lower curve in each panel represent linear regression estimates of the raw values and residual values vs. time, respectively.* A. *temperature (1984–2002);* B. *Secchi depth (1984–2002);* C. *hypolimnetic dissolved oxygen concentration (1991–2002).

O_e = average epilimnion DO concentration for stratified period
O_h = average hypolimnion DO concentration for stratified period
T_e = average epilimnion temperature for stratified period
T_h = average hypolimnion temperature for stratified period.

The vertical mixing reoxygenation value for each year was added to the temperature-corrected HVOD, R_t to obtain the rate corrected for both temperature and vertical mixing, R_{tv}.

Next, F_y the factor for the observational year was calculated from Rosa and Burns (1987) eqn. (11) namely;

$$F_y = R_{tv} (K_s/d_s + K_w) \quad (2)$$
$$= R_{tv}/3.15$$

where
K_s = 5.89 g m^{-2} mo^{-1}, the standard sediment DO uptake rate (Rosa and Burns 1987)
K_w = 1.9 mg L^{-1} mo^{-1}, the standard water DO uptake rate
d_s = 4.7 m , the standard hypolimnion thickness.

The water oxygen demand $(WOD)_{yv}$ component of the annual R_{tv} , corrected for temperature and vertical mixing was calculated from

$$WOD_{yv} = F_y \cdot K_w$$

If SOD_{yv} is the component of the HVOD rate caused by decomposition at the sediment/water interface then,

$$R_{yv} = SOD_{yv} + WOD_{yv}$$

$$SOD_{yv} = R_{yv} - WOD_{yv}$$

The SOD_{yv} value was corrected for hypolimnion thickness to the standard hypolimnion thickness of 4.7 m (Rosa and Burns 1987) by;

$$SOD_{yvs} = SOD_{yv} \times d_y/4.7$$

Where
d_y = average annual thickness for the year, y

And the annual HVOD rate corrected for temperature, vertical mixing and thickness, R_{yvs} is given by;

$$R_{yvs} = SOD_{yvs} + WOD_{yv}$$

Correction of HVOD Rates

Examination of the profiled data often showed that the values collected in early June could not be used in the HVOD rate calculation because final stratification had not yet become established. 15 June appeared to be the date by which final stratification had always occurred. Similarly, care was taken in the selection of data from the September profiles for use in the HVOD determination. Frequently, these profiles exhibited relatively high hypolimnion temperatures, indicating that large-scale downward mixing had occurred. Since the correction for downward mixing cannot duplicate the full complexity of this natural process, there is always a measure of error in the correction. This error would become unacceptably large when a large correction is made. Thus, many September profiles were excluded from the DO depletion rate calculations because of evidence of large-scale downward mixing of thermocline water. The best interval during which to conduct DO depletion rate surveys appears to be between 15 June and the end of August of any year.

GLNPO (Bertram 1993) followed the HVOD determination method described by Rosa and Burns (1987) and carried out the same corrections as Burns and Ross (2002), described above. However, the GLNPO HVOD analysis method applies the corrections for hypolimnion temperature, thickness, and downward mixing of thermocline water on a survey interval basis rather than on an annual basis as applied by Burns and Ross (2002). To facilitate comparisons between methods, and among years with differing durations of stratification, HVOD was standardized to a monthly depletion rate (mg O_2 /L/mo) by multiplying the daily depletion rate by 30.

RESULTS AND DISCUSSION

Temperature

The results of the daily (annualized) and among-year temperature analyses are shown in Figures 2A and 3A, respectively. The winter minimum value was close to 0°C, observed in mid February (Fig. 2A). Water temperature began to rise in early-mid April. Mean summer average maximum estimated over the period of record was 23.2°C (Fig. 2A), occurring in early August. The trend line through the complete set of observed data (seasonal regression, Fig. 3A) implies an annual increase of 0.38°C/y. However, this result is biased by a greater frequency of summer sampling occurring in the latter years, especially when the NWRI data are added to the GLNPO data. When the residuals are evaluated, the effect of the extra summer sampling is removed, and a trend of mean (± SE) rate of temperature increase of 0.037 ± 0.01°C/y was obtained. This estimate is plausible in light of other reports of global (0.3°C for 1980–2000; Lowe *et al.* 2004) and regional (Wuebbles and Hayhoe 2004) warming phenomena. The increase in atmospheric temperature for the Great Lakes region between 1975 and 2000 is approximately 0.5°C (estimated from Kling *et al.* 2003). Ocean surface temperatures are changing at approximately one-half of the change in atmospheric temperature (Lowe *et al.* 2004).

McCormick and Fahnenstiel (1999) reported long-term (1901–1993) increases in annual mean water temperature of near shore, surface-water intakes at five of seven Great Lakes locations examined. However, Erie, Pennsylvania, which draws water from the central basin, was one of the two locations at which the long-term trend was not statistically significant. Trends of increasing temperature at Erie and at Put In Bay, Ohio between 1980 and 1992 were weakened by recent year-to-year extremes. Their data over the period 1980–1992 for Put In Bay and Sandusky Bay, Ohio as well as for Erie contain some of the highest as well as some of the lowest values for the entire data record at each site. Such increased variability is consistent with other regional reports of greater frequency of meteorological extremes in the U.S. Midwest (Kunkel *et al.* 1998). More marked was an increase in the estimated maximum number of consecutive days per year during which water temperatures exceeded 4°C (McCormick and Fahnenstiel 1999), which is a measure of the theoretical maximum duration of summer stratification. More of this variation was associated with the timing of spring warming than with the timing cooling in the fall. Large lakes such as Lake Erie can possibly be effective monitors of regional warming.

Secchi Depth

Secchi depth (SD) showed the trends of a non-significant decrease from 1983 to 1989 (regression coefficient of –0.16 m/y, $p = 0.16$) and a significant decrease (0.12 m/y, $p < 0.05$) from 1989 to 2002. There was a significant decrease estimated over the entire 1983-2002 period of 0.065 m/y ($p < 0.01$; Fig. 3B). The observed SD data in Figure 3B shows considerable interannual variation. Figure 2B shows the annualized data from Station 73 in the center of the central basin. The annualized July/August SD values varied substantially from 4 to 10 m. Since the stratification is strong at this time of year, resuspended benthic sediment would be restricted to the hypolimnion and cannot contribute to surface-water turbidity, so SD variability must relate to the presence or absence of phytoplankton in the epilimnion waters or to sources of sediment external to the central basin.

The TP concentration decreased significantly during the 1983–89 period and increased over the 1989–2002 interval (Rockwell *et al.* 2005). Consequently, little net change in TP concentration was evident when it was estimated over the whole 1983–2002 period. Concentrations of Chl *a* declined during both the 1983–1989 and 1990–2002 periods (Rockwell *et al.* 2005). The minor net change in TP and monotonic decline in Chl *a* concentrations over the period of record suggest that the long-term reduction in central basin SD is not directly related to primary production processes.

A reduction in transparency of open waters of the central and western basins since 1990 was similarly documented by Barbiero and Tuchman (2004). They also reported an approximate doubling of turbidity in spring post-1990 vs. the period 1982–1990. Consequently, they concluded that TP and Chl *a* trends were likely unrelated to this reduction in transparency. Instead they attributed the trends to loadings from the Maumee River, the dominant contributor of suspended sediment to Lake Erie during floods. They observed a slight reduction in summer SD in the central basin between 1992 and 2004 relative to pre-dreissenid years (1983–1986) but also a reduction in turbidity. Barbiero and Tuchman (2004) described both an increase in turbidity and an increase in SD in the eastern basin between 1990 and 2003. They suggested that calcium concentrations had decreased since the 1990s (possibly due to uptake by expanding dreissenid populations) and that this might lessen the frequency and intensity of open water calcite crystal formation during summer days, resulting in increased water clarity.

Hypolimnetic Dissolved Oxygen

Central basin hypolimnetic DO concentrations tended to increase from 1991 to 1996 and decrease thereafter (Fig. 3C). However, there was no significant linear trend observable over the 11-y period 1991–2001 ($p = 0.29$; Fig. 3C).

TABLE 2. Hypolimnetic volume-corrected oxygen depletion (HVOD) rates (mg/L/mo) estimated by methods of Rosa and Burns (1987) (R&B) and Burns and Ross (2002) (B&R). Superscripts for each year indicate data sources—1: Rosa and Burns (1987). 2: Rathke and McRae (1989). 3: Great Lakes National Program Office of the U.S. EPA (unpubl.). 4: National Water Research Institute of Environment Canada (unpubl.).

Year	R&B HVOD Rate	Year	R&B HVOD Rate	B&R HVOD Rate
1970[1]	3.80	1988[3]	2.76	
1971[1]	3.76	1989[3]	2.67	
1973[1]	3.6	1990[3]	3.18	
1974[1]	4.53	1991[3,4]	3.39	3.32
1975[1]	3.45	1992[3,4]	3.24	3.33
1977[1]	3.59	1993[3,4]	3.14	3.22
1978[1]	3.32	1994[4]		2.32
1979[1]	3.72	1995[4]		3.13
1980[1]	3.43	1996[4]		2.68
1981[2]	3.05	1997[3,4]	3.20	3.40
1982[2]	3.52	1998[3,4]	3.79	4.06
1983[2]	3.35	1999[3,4]	3.19	3.58
1984[2]	3.18	2000[3]		3.02
1985[2]	3.75	2001[3,4]	2.99	3.21
1986[2]	4.72	2002[3]	3.61	
1987[3]	3.48	2003[3]	3.45	

HVOD Results

The corrected HVOD rates calculated by the method of Rosa and Burns (1987) and by Burns and Ross (2002) are shown in Table 2. Sufficient data were available to permit a comparison of estimates for 7 y during the period of record. The 1991–93 data available were from GLNPO alone, and the rates calculated from these years by the Rosa and Burns (1987) and Burns and Ross (2002) methods are similar (Table 2). However, when Burns and Ross combined the data from GLNPO and NWRI for the years 1997–99 and 2001, systematic differences were evident. The Burns and Ross method gave estimates greater by a factor of about 5% than was calculated by GLNPO. This difference was statistically significant (mean ± SE difference of 0.17 ± 0.056 mg/L/mo; paired-comparison *t*-test, n = 7, $p < 0.05$). Nevertheless, the NWRI data permitted HVOD rates to be calculated for 4 years when there were no GLNPO data. The NWRI pattern of sampling stations differs from that of GLNPO (Fig. 1), and this may account for some differences in the computed HVOD rates.

Hypolimnetic volume-corrected oxygen depletion rates showed a nonsignificant downward trend from 1970 to 1989 (p = 0.12) and a weaker nonsignificant (p = 0.35) upward trend from 1990 to 2003 (Fig. 4A). The trends were weakened by the occurrence of very high and low rates in years that are close together such as 1986 and 1988 (depletion rates of 4.72 and 2.76 mg/L/mo, respectively), and 1996 and 1998 (depletion rates of 2.68 and 4.06 mg/L/mo). This variability was unexpected because the rates have been corrected for varying temperature, hypolimnion thickness, and downward mixing conditions. The 1970–1989 trend line would be statistically significant if one ignored the 1986 datum, yielding an annual decline of 0.048 mg/L/mo (p = 0.004; $R^2 = 0.44$).

One probable source of the observed variability in DO depletion rates is hypolimnetic production of oxygen by phytoplankton growth. Additionally, the downward trend from 1970 to 1989 followed by the gradual rise following 1990 could be related to the changes in phosphorus loading over those years (Dolan and McGunagle 2005, Table 2, and below), which is reflected in changing open water TP concentrations over these years (Rockwell *et al.* 2005). The other factor often invoked to explain recent trends of increasing HVOD is the advent of *Dreissena* mussels into the central basin in 1990 and their possible effect on the amount of organic matter and nutrients in the waters of the basin.

To assess the potential relationships between TP loadings and HVOD, we regressed estimated HVOD ((calculated according to the Rosa and Burns (1987) method where available and the Burns and Ross (2002) method otherwise (Table 2) against total annual loading of TP to Lake Erie (Dolan and McGunagle 2005)). This relationship was marginally statistically significant (HVOD = $2.78 + 5.10 \times 10^{-5} \pm 2.0 \times 10^{-5}$ (TP load); $F_{[1,28]} = 6.45$, $p < 0.02$, $R^2 = 0.19$). However, regression of HVOD against the previous year's annual loading estimate gave a markedly stronger relationship ($R^2 = 0.26$, $p < 0.005$; Fig. 5). There was no indication that the relationship between HVOD and TP loading of either the current or previous year changed after the establishment of dreissenids (Fig. 5). Thus, the discrepant temporal trends in HVOD between pre-and post-dreissenid periods evident in Figure 4A can largely be accounted for by changes in the factors that influence external TP loadings (formerly municipal releases; more recently, tributary discharge patterns (Dolan and McGunagle 2005) and the environmental conditions that determine hypolimnial

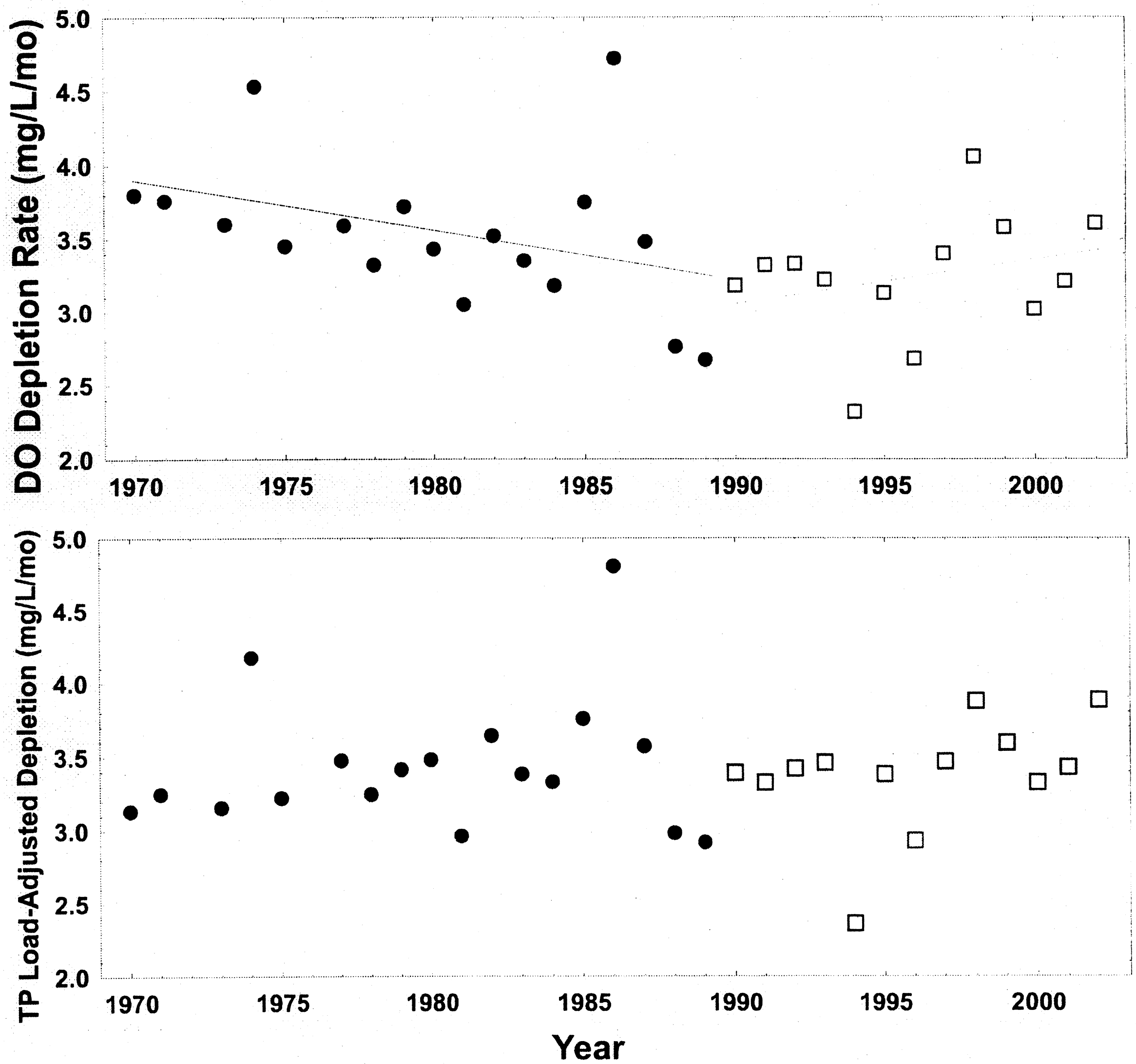

FIG. 4. **A.** ***Hypolimnetic Volume-corrected Oxygen Depletion (HVOD) trend lines, 1970–1989 (filled circles and solid line) and 1990–2003 (open squares and dashed line). Rates for 1970–1989, 1990, 2002, and 2003 were determined by GLNPO; 1991–2001 rates were determined by Burns and Ross; 1970–1989 trend line: HVOD = 68.3 – 0.033 × Year (*****p*** ***= 0.12; R^2 = 0.14). 1990–2003 trend line: HVOD = –58.3 + 0.031 × Year (*****p*** ***= 0.35; R^2 = 0.08). Neither slope is significantly different from zero. If 1986 data point is omitted, HVOD = 98.86 – 0.048 × Year (*****p*** ***= 0.004; R^2 = 0.44; SE of slope = 0.014).*** **B.** ***HVOD estimates statistically adjusted to constant 29-y mean annual loading of TP according to regression equation shown in Figure 5.***

thickness. Statistically controlling HVOD for changes in the previous year's TP loadings removes virtually all temporal trends (Fig. 4B).

Hypolimnetic Production

Vertical profiles of DO concentration taken at in the middle of the central basin provide evidence that metalimnetic and epibenthic primary production also may also influence hypolimnetic depletion rates (Fig. 6). Primary production in the thermocline is indicated by DO supersaturation of up to 134%. Additionally, a thin layer (about 2 m thick) on the bottom of the lake where the temperature is a little lower than the overlying hypolimnion water shows evidence of primary production at the bottom of the

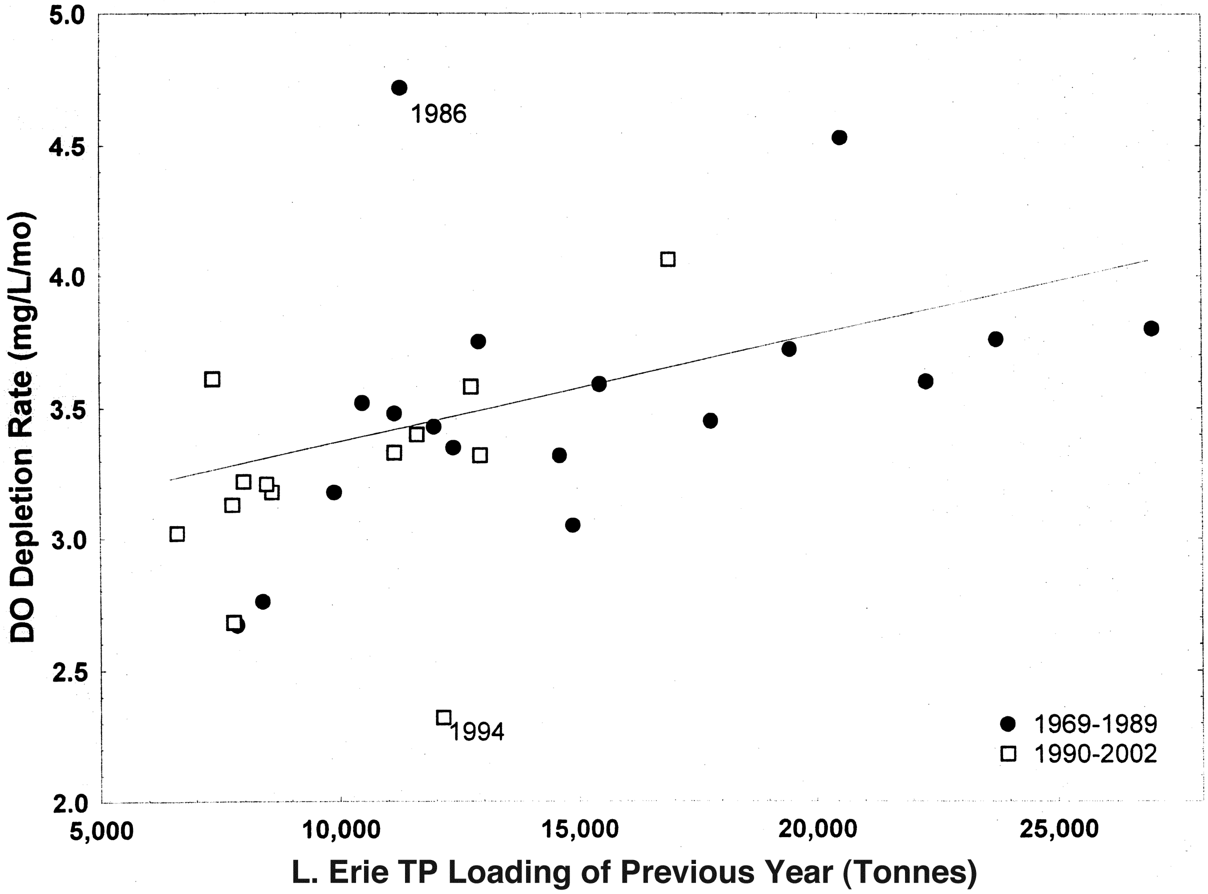

FIG. 5. Relationship between hypolimnetic volume-corrected dissolved oxygen depletion rate (HVOD; mg/L/mo) and the previous year's load of total phosphorus (tonnes) as estimated by Dolan and McGunagle (2005). Regression line takes the form HVOD = 2.70 + 5.6 × 10^{-5} tonnes ($F_{[1,28]}$ = 9.75, p < 0.005, R^2 = 0.26; SE of regression coefficient = 1.8 × 10^{-5}). Filled circles represent HVOD data for 1970–1989; open squares represent years 1990–2002.

lake because of the increased DO concentrations in this layer. Other DO profiles taken in 1996 indicated that hypolimnetic primary production continued for about a month, to the last week of July.

Evidence of hypolimnetic oxygen production is normally difficult to observe because hypolimnetic oxygen uptake counteracts the concentration increases caused by the production, but such production can be substantial nevertheless. Burns *et al.* (1996) found that midsummer hypolimnetic oxygen production was equivalent to 59% of the production occurring the 10-m thick epilimnion of a mesotrophic lake. This could account for the high oxygen concentrations observed in 1996 and 1997. Evidence of thermocline oxygen production was also found in 1994, and this would account for the anomalously low HVOD rates observed in 1994 and 1996 (Figs. 4A, 5). Carrick *et al.* (2005) and Ostrom *et al.* (2005) also found independent evidence of late spring hypolimnetic subsidies of dissolved oxygen through primary production in Lake Erie's central basin in 2002. However, in both cases, the production:respiration quotient had shifted to net heterotrophy by late summer. Evidence of live algae being deposited basin-wide on the bottom of the central basin was documented in 1970 in Project Hypo (Braidech *et al.* 1972), and these algae significantly diminished the observed sediment oxygen demand rate (Lucas and Thomas 1972). Water clarity in 1970 was high, and significant hypolimnetic oxygen production was recorded (Lucas and Thomas 1972), indicating that at least occasional years with hypolimnetic primary production occurred before the dreissenid invasion.

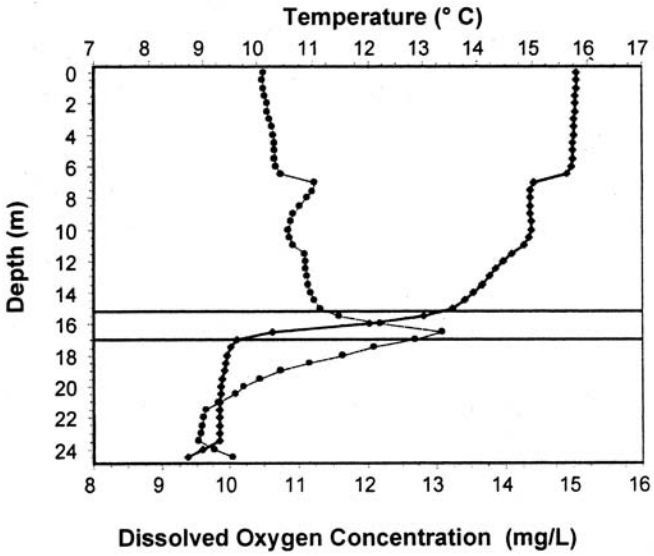

FIG. 6. DO concentration (10.3 mg/L at surface), and temperature profiles (15.8°C at surface) at Environment Canada Station 984 on 19 June 1996.

Hypolimnetic oxygen production can be sudden and transient. For example, on 11 June 1996, the DO profile was normal with the hypolimnion DO concentration of 10.0 mg/L slightly below that of the epilimnion. One week later on 18 June, however, the hypolimnion DO concentration had substantially increased to 11.8 mg/L, and a peak of 14.5 mg/L was observed in the thermocline. A pattern of sudden, large-scale settling out of algae from the surface waters to the hypolimnion of the central basin was observed in 1970 (Braidech *et al.* 1972). A similar event could account for the large variation in hypolimnetic concentrations observed in June 1996. The resultant high or low concentrations of epilimnetic phytoplankton could explain the large interannual variations in SD observed at Station 78.

The loss of nutrients from the epilimnion need not be restricted to settlement of phytoplankton. Charlton *et al.* (1999) postulated that late summer settling by dreissenid veligers could carry significant quantities of TP and other nutrients out of the euphotic zone.

Areal Extent of Zone of Oxygen Depletion

An approximation of the maximum areal extent of the zone of oxygen depletion was determined for each year for which data were available (Fig. 7). We deemed a station to have become anoxic if its epibenthic dissolved oxygen concentration declined

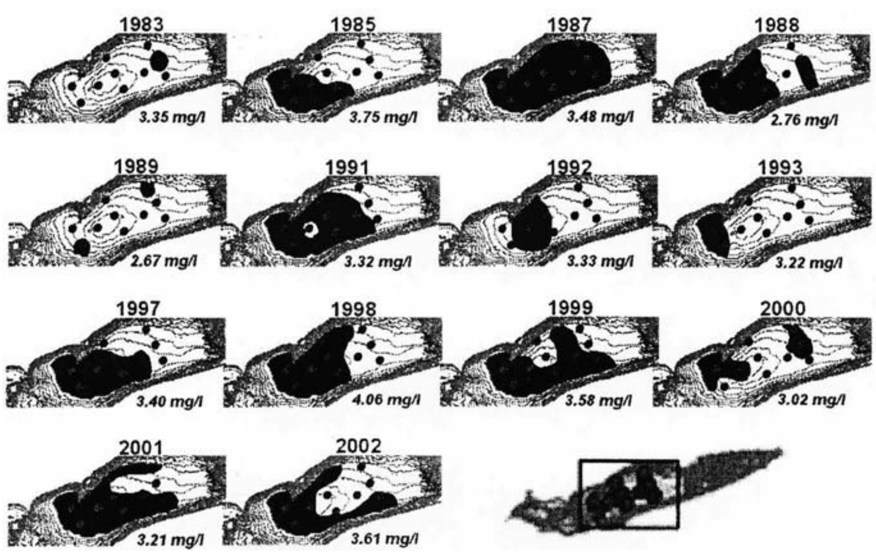

FIG. 7. Depth contour maps (1.53-m intervals; deepest contour line = 23 m) of central basin of Lake Erie showing maximum extent of central basin anoxia (< 1.0 mg/L) observed in Lake Erie from 1883–2002. The box in the bottom right-hand image shows the location detailed in the other panels. The numeral in the bottom right-hand corner of each image indicates hypolimnetic volume-corrected oxygen depletion rate (mg/L/mo) determined over the period of summer stratification of that year.

to 1 mg/L or less at any time during a sampling year. The boundaries of the anoxic zone were then estimated by drawing a 1 mg/L isopleth around the stations at which anoxia had been recorded. The 1 mg/L transition point between anoxic and non-anoxic stations was estimated using linear interpolation.

We determined whether there was any correlation between the maximum areal extent of depletion (ranked by eye) and the annual estimate of dissolved oxygen depletion calculated by the Rosa and Burns (1987) method using Spearman's rank correlation coefficient. There was no significant correlation between the two variables over the 14 y for which data were available (Fig. 7; $r_s = 0.26$, $p > 0.05$).

The effect of oxygen depletion rates on the areal extent of the dead zone depends on meteorological conditions. The thickness of the hypolimnion at set up of thermal stratification, the time that set up occurs, and the temperature of the hypolimnion all influence the amount of oxygen trapped in the hypolimnion (Rosa and Burns 1987). The extent of the anoxic zone is also affected by the duration of stratification in September. Strong winds at the beginning of the month will eliminate stratification, as occurred in 2002 (D. Rockwell, US EPA, personal communication), but a calm month will enable the stratification to endure and anoxia to occur as a result. The settling out of phytoplankton and their subsequent production of DO may have an indeterminate effect on the extent of anoxia because the oxygen produced by the algae is offset when they die and decompose in September, accelerating the rate of oxygen uptake at that time (Bums and Ross 1972, Carrick *et al.* 2005, Edwards *et al.* 2005, Ostrom *et al.* 2005).

CONCLUSIONS

Examination of the central basin temperature trends (Fig. 3) shows the importance of deseasonalizing data before trending them because this removes the biases associated with effects of irregular sampling. The temperature trend of a 0.037 ± 0.01°C increase per year of the central basin is within the global warming range of 0.02–0.04°C/y.

The SD has been decreasing at a rate of 7 ± 3 cm/y over the period of record and varies widely (4–10 m) in midsummer. This decrease does not seem to be related to changes in either TP or Chl *a* concentrations in the epilimnetic waters of the central basin, but may reflect increasingly frequent flood-related high sediment loads from tributaries. The large interannual variation in SD may partially reflect whether phytoplankton remain concentrated in the epilimnion (giving shallow SD), or have settled into the lower layers, producing deep Secchi depth values.

HVOD rates were also found to be highly variable. These rates can be strongly affected by the production of oxygen in the thermocline and hypolimnion by phytoplankton that have settled into these layers and remained photosynthetically active. The HVOD rates calculated by GLNPO show a weak tendency to decrease from 1970 to 1989 and a slight trend to increase with time from 1990 to 2003. However, these patterns seem to be related to the previous year's TP loading to the basin during the previous year. Since TP loading trends largely reflect the consequences of improving water treatment through the 1980s and increasing contributions from tributary run-off through the 1990s, there is little direct evidence to suggest that the appearance of dressenids has directly influenced hypolimnetic oxygen depletion rates in the central basin. The observation that central-basin HVOD tracked the reductions in TP loadings through the 1980s may be the first direct affirmation that central basin oxygen dynamics can be regulated by phosphorus inputs, albeit more weakly than originally hoped. This implies that TP loads must continue to be regulated if we wish to minimize oxygen depletion rates in an effort to reduce frequency of episodic central basin anoxia.

ACKNOWLEDGMENTS

We thank Murray Charlton, Environment Canada for making available the NWRI data to fill gaps in the GLNPO data record. Murray Charlton and Lynda D. Corkum provided valuable review comments and background information. This research was funded by grants from the Great Lakes National Program Office, US EPA to N. Burns and D. Dolan. This document has not been subjected to the US Environmental Protection Agency's required peer and policy review and therefore does not necessarily reflect the views of the agency and no official endorsement should be inferred.

REFERENCES

Barbiero, R.P., and Tuchman., M.L. 2004. Long-term dreissenid impacts on water clarity in Lake Erie. *J. Great Lakes Res.* 30:557–565.

Bertram, P. 1993. Total phosphorus and dissolved oxygen trends in the central basin of Lake Erie, 1970-1991. *J. Great Lakes Res.* 19:244–236.

Braidech, T., Gehring, P., and Kleveno, C. 1972. Biological studies related to oxygen depletion and nutrient regeneration processes in the Lake Erie Central Basin. In *Project HYPO—an intensive study of the Lake Erie central basin hypolimnion and related surface water phenomena,* N.M. Burns and C. Ross, eds. CCIW paper No. 6, U.S. EPA Report No. TS-04-71-208-24.

Burns, N.M., and Ross, C. 1972. *Project HYPO—an intensive study of the Lake Erie central basin hypolimnion and related surface water phenomena.* CCIW paper No. 6, U.S. EPA Report No. TS-04-71-208-24.

_____, and Ross, C. 2002. *Assessment of the US EPA Great Lakes Program Office Lake Erie central basin monitoring data, 1983 to 2001.* Seveno Ltd., Auckland, New Zealand.

_____, Gibbs, M.M., and Hickman, M.L. 1996. Measurement of oxygen production and demand in lake waters. *NZ J. Mar. Freshwat. Res.* 30:127–133.

_____, Rutherford, J.C., and Clayton, J.S. 1999. A monitoring and classification system for New Zealand lakes and reservoirs. *Lake Reservoir Manage.* 15:255–271.

Carrick, H.J., Moon, J.B., and Gaylord, B.F. 2005. Phytoplankton dynamics and hypoxia in Lake Erie: a hypothesis concerning benthic-pelagic coupling in the central basin. *J. Great Lakes Res.* 31 (Suppl. 2): 111–124.

Chapra, S.C. 1977. Total phosphorus model for the Great Lakes. J. Environ. Eng. Div., *Am. Soc. Civil Engineers* 103(EE2):147–161.

Charlton, M.N., Le Sage, R.E., and Milne, J.E. 1999. Lake Erie in transition: the 1990's. In *State of Lake Erie: Past, Present and Future.* M. Munawar, T. Edsall, and I.F. Munawar, eds., pp. 98–123, Leiden, The Netherlands: Backhuys Publishers.

DiToro, D.M., and Connolly, J.P. 1980. *Mathematical models of water quality in large lakes: Part 2—Lake Erie.* Report EPA-600/3-80-065, U. S. Environmental Protection Agency, Duluth, MN.

Dolan, D.M., and McGunagle, K.P. 2005. Lake Erie total phosphorus loading analysis and update: 1996-2002. *J. Great Lakes Res.* 31 (Suppl. 2):11–22.

Edwards, W.J., Conroy, J.D., and Culver, D.A. 2005. Hypolimnetic oxygen depletion dynamics in the central basin of Lake Erie. *J. Great Lakes Res.* 31 (Suppl. 2):262–271.

Fahnenstiel, G.L., Bridgeman, T.B., Gregory, A.L., McCormick, M.J., and Nalepa, T.F. 1995. Phytoplankton productivity in Saginaw Bay, Lake Huron: effects of zebra mussel (*Dreissena polymorpha*) colonization. *J. Great Lakes Res.* 21:465–475.

Howell, E.T., Marvin, C.H., Bilyea, R.W., Kauss, P.B., and Somers, K. 1996. Changes in environmental conditions during *Dreissena* colonizations of a monitoring station in eastern Lake Erie. *J. Great Lakes Res.* 22:744–756.

International Joint Commission. 1978. *Fifth year review of Canada-United States Great Lakes Water Quality Agreement.* Report of Task Group III. A technical group to review phosphorus loadings. 83 p.

Kling, G.W., Hayhoe, K., Johnson, L.B., Magnuson, J.J. Polasky, S., Robinson, S.K., Shuter, B.J., Wander, M.M., Wuebbles, D.J., Zak, D.R., Lindroth, R.L., Moser, L.C., and Wilson., M.J. 2003. *Confronting climate change in the Great Lakes region: impacts on our communities and ecosystems.* Union of Concerned Scientists, Cambridge, Massachusetts, and Ecological Society of America, Washington, D.C

Kunkel, K.E., Andsager, K, Conner, G. Decker, W.L., Hillaker Jr., H.J., Naber Knox, P., Nurnberger, F.V., Rogers, J.C., Scheeringa, K., Wendland, W.M., Zandlo, J., and Angel, J.R. 1998. An expanded digital daily database for climatic resources applications in the midwestern United States. *Bull. Am. Meteorol. Soc* .79: 1357–1366.

Lakes Consulting. 2000a. *LakeWatch—a program for lake and reservoir monitoring data assessment.* Lakes Consulting, 42 Seabreeze Rd., Devonport, New Zealand 1309. www.lakewatch.net [November 2005]

_____. 2000b. *Manual on monitoring lakes and reservoirs.* Seveno Ltd., P O Box 90-058, Auckland, New Zealand. www.lakewatch.net. [November 2005].

Lesht, B.M., and Rockwell, D.C. 1985. *The state of the middle Great Lakes: results of the 1983 Water Quality Survey of Lakes Erie, Huron and Michigan.* Argonne National Laboratory, Argonne, IL 60439, U.S.A. ANL/ER-85-2.

Lowe, J., Smith, F., Jenkins, G., and Pope, V. (eds). 2004. *Uncertainty, risk and dangerous climate change.* Recent research on climate change science from the Hadley Centre December 2004. Met Office, Hadley Centre, Exeter, United Kingdom. URL: http://www.meto.gov.uk/research/hadleycentre/pubs/brochures/B2004/global.pdf [November 2005]

Lucas, A.M., and Thomas, N.A. 1972. Sediment oxygen demand in Lake Erie's central basin, 1970. In *Project HYPO—an intensive study of the Lake Erie Central Basin hypolimnion and related surface water phenomenon,* N.M. Burns and C. Ross, eds., pp. 45-50. CCIW paper No. 6, United States EPA Report No.TS-04-71-208-24.

Makarewicz, J.C., and Bertram, P.E. 1991. Evidence for the restoration of the Lake Erie ecosystem. *BioScience* 41:216–223

Makarewicz, J.C., Bertram, P., and Lewis, T.W. 2000. Chemistry of the offshore surface waters of Lake Erie: pre- and post-*Dreissena* introduction (1983–1993). *J. Great Lakes Res.* 26:82–93.

McCormick, M.J., and Fahnenstiel, G.L. 1999. Recent climatic trends in nearshore water temperatures in the St. Lawrence Great Lakes. *Limnol. Oceanogr.* 44:530–540.

Nicholls, K.H., and Hopkins, G.J. 1993. Recent charges in Lake Erie (north shore) phytoplankton: cumulative impacts of phosphorus loading reductions and the zebra mussel introduction. *J. Great Lakes Res.* 19:637–647.

Ostrom, N.E., Russ, M.E., Field, A., Piwinski, L., Twiss, M.R., and Carrick, H.J. 2005. Ratios of community respiration to photosynthesis and rates of primary production in Lake Erie via oxygen isotope techniques. *J. Great Lakes Res.* 31 (Suppl. 2):138–153.

Rathke, D.E., and Edwards, C.J. 1985. *A review of trends in Lake Erie water quality with emphasis on the 1978-1979 Intensive Survey.* Report to the Surveillance Work Group, International Joint Commission, Windsor, ON.

———, and McRae, G. (eds). 1989. *1987 Report on Great Lakes Water Quality. Appendix B. Volume 1.* Great Lakes Water Quality Board Report to the International Joint Commission. Windsor, ON.

Rockwell, D.C., Warren, G.J., Bertram, P.E., Salisbury, D.K., and Burns, N.M. 2005. The U.S. EPA Lake Erie indicators monitoring program 1983–2002: trends in phosphorus, silica, and chlorophyll *a* in the central basin. *J. Great Lakes Res.* 31 (Suppl. 2):23–34.

Rosa, F., and Burns, N.M. 1987. Lake Erie central basin oxygen depletion changes from 1929–1980. *J. Great Lakes Res.* 13:684–696.

USEPA. 2003. *Sampling and analytical procedures for GLNPO's open lake water quality survey of the Great Lakes.* U.S. Environmental Protection Agency, Great Lakes National Program Office, Chicago, IL. EPA 905-R-03-002

Vanderploeg, H.A., Liebig, J.R., Carmichael, W.W., Agy, M.A., Johengen, T.H., Fahnenstiel, G.L., and Nalepa, T.F. 2001. Zebra mussel (*Dreissena polymorpha*) selective filtration promoted toxic *Microcystis* blooms in Saginaw Bay (Lake Huron) and Lake Erie. *Can. J. Fish. Aquat. Sci.* 58:1208–1221.

Wuebbles, D.J., and Hayhoe, K. 2004. Climate change in the midwest: informing regional policy decisions. *Mitig. Adapt. Strat. Glob. Change* 9:335–363.

Submitted: 2 July 2004
Accepted: 19 December 2005
Editorial handling: Jan J.H. Ciborowski