

## **2008 Status of the Lake Ontario Ecosystem**

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### **Introduction**

North America's Great Lakes have experienced accelerated ecological change since the 1950s. Overfishing, cultural eutrophication, and contaminant discharge in the 1950s and 1960s led to degradation of water quality, loss of habitat, and declines in native fish populations. Such stresses raised public concern about the condition of the Great Lakes and led to a new period of environmental awareness and restoration in the 1970s, 1980s, and 1990s. The historic Great Lakes Water Quality Agreement (GLWQA) between the United States and Canada in 1972, later revised in 1978 and 1987 (IJC 1988), marked a new era of ecosystem management. The agreement placed limits on phosphorus loads to each of the Great Lakes, initiating the process of oligotrophication (declining productivity). Phosphorus levels declined in Lake Ontario, but at the same time, demand for a salmonid sport fishery increased, alewife exhibited highly variable population dynamics, pelagic zooplankton production declined, pelagic fish stocks rebounded, and nonindigenous species (NIS) such as the zebra mussel, quagga mussel and the predatory cladocerans *Bythotrephes longimanus* and *Cercopagis pengoi* proliferated (Christie *et al.* 1987, EPA 1993, Mills *et al.* 2003, Mills *et al.* 2005). Clearly, Lake Ontario is an

ecosystem in a state of flux, and the public's interest in and awareness of the Lake Ontario environment continues to increase. It is incumbent upon the scientific and management communities to understand these ecosystem changes and their implications to the condition of this important freshwater resource.

The condition of Lake Ontario's lower food web determines the lake's ability to support a healthy fish population, and monitoring key indicators in the lower food web is one approach toward assessing ecosystem change. From 1995-2008, we have conducted a research program (hereafter referred to as the biomonitoring program) in Lake Ontario that measured indicators of ecosystem health as identified by the Lake Ontario Pelagic Community Health Indicator Committee (EPA 1993). The primary objective was to evaluate temporal and spatial patterns in total phosphorus (TP), soluble reactive phosphorus (SRP), chlorophyll *a* (chl *a*), Secchi depth, and crustacean zooplankton (density, biomass, and size structure). While earlier analyses were focused necessarily on intra-annual and short-term interannual patterns, we now have sufficient data to evaluate long-term trends.

Long-term data can be used to evaluate trends for statistical significance and to evaluate the efficacy

of the current monitoring program. For example power analysis is used to determine minimum sample sizes for accurate and reliable statistical results, to determine the power of a statistical test in detecting a given effect size, and to make appropriate adjustments to a research program. Here, we report the status of Lake Ontario's lower food web in 2008. In addition, we evaluate long-term trend data for statistical significance and use power analysis to answer specific questions regarding key indicators. The results of these analyses provide the framework needed for managers to incorporate the lower food web into decision-making processes related to fish stocking and harvest.

The biomonitoring program has been successfully accomplished through interagency support of the New York State Department of Environmental Conservation (NYSDEC) including regional staffs at Watertown, Cortland, and Avon, the U.S. Fish & Wildlife Service Lower Great Lakes Fishery Resources Office (USFWS), the Ontario Ministry of Natural Resources (OMNR), the U.S. Geological Survey – Great Lakes Science Center (USGS), and Cornell University. The program has also allowed us to (1) provide a benchmark for comparison to evaluate changes in the lower food web of Lake Ontario, and (2) link with closely allied programs such as recent Sea Grant projects assessing changes in Lake Ontario's mysid population. The biomonitoring variables we have chosen to examine contribute to research on Lake Ontario's embayment, nearshore, and offshore habitats which are crucial to understanding the coupling between ontogenetic patterns of habitat utilization by fish and whole-lake predator-prey interactions, and for understanding the role of inshore habitats in food web dynamics.

### **Report Objectives**

Using data from 2008, we addressed the following questions:

- (1) *Do biomonitoring variables differ between embayment, nearshore, and offshore habitats?*
- (2) *Does zooplankton community structure suggest a healthy balance between predator and prey fish?*

Using data from 1995 – 2008, we addressed the following questions:

- (1) *What trends are evident for each biomonitoring variable in each habitat? Are the trends significant? Do the trends differ among habitats?*
- (2) *What is our power to detect change in each variable given a pre-determined effect size? Should the sampling program be modified based on the results of the power analysis?*

### **Methods**

#### Sampling

We measured total phosphorus (TP), soluble reactive phosphorus (SRP), chlorophyll *a* (chl *a*), water temperature, Secchi depth, zooplankton density, zooplankton biomass, and zooplankton size at offshore, nearshore and embayment sites in Lake Ontario. Samples were collected from three embayment and five nearshore sites biweekly from May through October 2008 (12 sampling weeks). Embayments sampled were Chaumont Bay, Sandy Pond, and Sodus Bay. Nearshore sites were Galloo Island-Lake, Chaumont Bay-Lake, Sandy Pond-Lake, Sodus Bay-Lake, and Oak Orchard-Lake. Nearshore sites had depths ranging from 8.5m to 13.7m, whereas depths of offshore sites ranged from 24m to 176m. The R/V's Kaho and Seth Green collected offshore samples from April through October 2008. Offshore sampling totaled 37 samples taken from 13 sites. Figure 1 shows a map of sampling locations.

#### Water Chemistry

Water samples were collected for analysis of chl *a* and two phosphorus fractions: total phosphorus (TP) and soluble reactive phosphorus (SRP). Each sample was obtained by using an integrated water sampler (1.9cm inside diameter Nalgene tubing) lowered to a depth of 10m or bottom minus 1m where total depth was 10m or less. The tube was then closed off at the surface end and the column of water transferred to 2L Nalgene containers. From each sample a 100mL unfiltered aliquot sample was frozen for later analysis of TP (Menzel and Corwin 1965). We also filtered 1-2L of water through a Whatman 934-AH glass fiber filter that was frozen for later analysis of chl *a* using the standard acetone extraction method (Strickland and Parsons 1972).

A 100mL sample of filtered water was also frozen for later analysis of SRP (Strickland and Parsons 1972).

#### Quality Assurance - Quality Control

In 2008, Quality Assurance and Quality Control (QAQC) analyses were conducted for TP and SRP. Water samples were collected from embayment and nearshore habitats using the standard sampling methods.

In July, 10 aliquots of raw water were taken from one sample for TP analysis. An additional 10 aliquots of filtered water were taken from the same sample for SRP analysis. One subset of these samples was analyzed at CBFS; another subset was analyzed at the Upstate Freshwater Institute (UFI), an EPA certified laboratory located in Syracuse, NY, for comparison with CBFS results. In the event that significant differences ( $p < 0.05$ ) arose between CBFS and UFI laboratory results, an adjustment equation would have been generated and applied to CBFS results (TP and/or SRP) in order to account for those discrepancies. The adjustment equation is a regression equation that adjusts the CBFS data in relation to the UFI data.

In 2008, we also conducted replicate sampling for analyses of TP, SRP, and chl *a*. Triplicate samples were collected at each nearshore and embayment location twice in August. From each of the three samples, one aliquot was taken for TP, one for SRP, and one for chl *a* analysis. The samples were analyzed at CBFS to determine the amount of variability between samples.

#### Zooplankton

Zooplankton samples were collected with a standard 0.5m diameter, 153 $\mu$ m mesh nylon net. At most embayment and nearshore sites, we strained a 10m water column. At offshore sites, we sampled a 5-40m water column. Zooplankton were anesthetized using antacid tablets, then preserved in the field with 95% ethyl alcohol. Single samples were collected on a biweekly basis at embayment and nearshore sites from May to October, except for July and August when two replicate samples were collected per site on each date.

In the CBFS laboratory, each sample was strained through a 1.02mm mesh cup to separate *C. pengoi*

and other larger organisms (>1mm in length) from smaller zooplankton (<1mm). This was done because *C. pengoi* form clumps in the sample, making the usual random subsampling of 1mL samples inappropriate. For each sample that contained clumps of *C. pengoi*, two analyses were performed, one on the smaller zooplankton and one on the larger zooplankton (including *C. pengoi*) that were caught in the 1mm mesh strainer. The larger zooplankton were measured and enumerated by subsampling at least 100 organisms from a gridded, numbered petri dish in which the sample had been homogeneously separated.

The subsample was examined through a compound microscope at 10-40X magnification. Images from the sample were projected onto a digitizing tablet that was interfaced with a computer. Zooplankton were measured on the digitizing tablet and identified to species, excluding nauplii and copepodites (Pennak 1978, Balcer *et al.* 1984). To calculate the total number of large crustaceans and *C. pengoi* in the clumped part of the sample, we used a ratio of wet weights of the subsample to wet weights of the total sample. Wet weights were determined using a Sartorius balance. For the smaller sized zooplankton samples, we counted and measured at least 100 organisms from one or more 1mL random subsamples using the same microscope and digitizer technique. In earlier years of this project an electronic touch screen (1995-1997) and a 20X microprojector (1998-2000) were used for measuring the zooplankton (Hambright and Fridman 1994).

We used length:dry-weight regression equations (CBFS unpublished data) to estimate zooplankton biomass. Biomass of *C. pengoi* was estimated using two different regression equations. Nearshore samples were analyzed using the regression  $\ln W = 1.7164 + 2.3703 \ln L$ , where W represents biomass ( $\mu$ g) and L represents length (mm). Offshore samples were analyzed using the regression  $\ln W = 1.3690 + 2.7686 \ln L$ . Each regression is based on the original length:dry weight regression (determined using unpreserved specimens) established in 1998, but that have since been modified to increase the estimated weight by 42% to compensate for shrinkage in length due to preservation (Giguere *et al.* 1989).

### Data Analyses

#### *(1) Spatial Variability*

For each of the eight embayment and nearshore study parameters (TP, SRP, chl *a*, water temperature, Secchi depth, zooplankton density, size, and biomass), we compared the biweekly averages between the two habitats (nearshore  $n=5$  and embayment  $n=3$ ) using a paired t-test for means. For the comparisons between offshore, nearshore, and embayment habitats, we examined each of seven study parameters (TP, SRP, chl *a*, water temperature, zooplankton density, size, and biomass) with paired two-sample t-tests assuming unequal variances. These comparisons were made with data that were collected during the same sampling week. Zooplankton density and biomass were log transformed, and each zooplankton relationship was considered significant following the Bonferroni adjusted  $\alpha=0.0167$  ( $p<0.0167$ ). Differences in each chemical/physical relationship were considered significant at  $p<0.05$ .

#### *(2) Zooplankton Community Dynamics*

We divided zooplankton into the following seven groups: daphnids (*Daphnia mendotae*, *D. pulicaria*, *D. retrocurva*); bosminids (*Bosmina longirostris*, *Eubosmina coregoni*); calanoid copepods (*Diaptomus minutus*, *D. oregonensis*, *D. sicilis*, *D. ashlandi*, *Epischura lacustris*, *Eurytemora affinis*, *Limnocalanus macrurus*); cyclopoid copepods (*Acanthocyclops vernalis*, *Diacyclops thomasi*, *Mesocyclops edax*, *Tropocyclops prasinus*); other cladocera (*Ceriodaphnia quadrilangula*, *Chydorus sphaericus*, *Leptodora kindtii*, *Diaphanosoma sp.*, *Alona sp.*, *Holopedium gibberum*, *Polyphemus pediculus*, *Camptocercus sp.*), *Bythotrephes longimanus* (formerly *cederstroemi*); and *Cercopagis pengoi*. We compared average biweekly biomass proportions of each group between embayment and nearshore habitats using a paired t-test to determine differences in community composition between these areas. In addition, we compared the biomass of each group in July-August using t-tests assuming unequal variance on log (x+1) transformed data for embayment, nearshore, and offshore (water depth of 15m or more) areas.

#### *(3) Temporal Variability and-Power Analysis*

For spring TP, seasonal SRP, summer chl *a*, and summer zooplankton biomass, we performed

power analyses by habitat. First, trends were plotted for each variable and tested for significant slope ( $p<0.10$ ) using linear regression. Given the amount of noise inherent in ecological data, we elected to use this less stringent acceptance criterion. We then calculated the number of years it would take to observe a significant effect for each parameter/habitat combination (TRENDS software) using an effect size of 5% (i.e. 5% per year change) and the observed variance. An effect size of 5% was chosen because it equates to an overall change of 37% after 10 years—a change large enough to be considered biologically significant. We also calculated our power to detect change using an effect size of 5%, the observed variance, and the actual number of years that monitoring has occurred. Power is the probability that a statistical test will reject the null hypothesis when it is false (Hatch 2003), and in general, a power level of 0.8 is considered adequate (Cohen 1988).

### **Results**

#### QAQC

In 2008, QAQC analyses were conducted for July TP and SRP samples. A comparison of samples analyzed at CBFS with those at Upstate Freshwater Institute (UFI) showed that TP values were not significantly different ( $p>0.05$ ). SRP samples did show a significant difference ( $p<0.001$ ). However, some SRP values were below the level of quantification for UFI. Low concentrations combined with high sample to sample variability precluded the use of UFI data for SRP adjustment. As a result, there was no adjustment of the regression equation in either set of data.

The analysis of August embayment and nearshore TP, SRP, and chl *a* triplicate samples showed that 50% of samples had a coefficient of variation (CV) greater than the predetermined acceptance level of 10% (range=0-63%). Triplicate samples that had a CV greater than 10% were accepted due to the low levels of TP, SRP, and chl *a* detected in Lake Ontario. Consequently, we pooled triplicate TP, SRP, and chl *a* samples for each August replicate sampling effort and reported an average value for each.

### Habitat Comparisons

*Secchi depth, chl a, TP, SRP, and water temperature.* Embayments were characterized by high concentrations of chl *a*, TP, and SRP and low Secchi depths compared to nearshore and offshore sites. Embayments had the lowest water clarity during all sampling weeks ( $p < 0.0001$ ), with Secchi values always less than 5m and often less than 2m, while Secchi depths greater than 10m were not uncommon in nearshore and offshore sites (Figure 2a). May through October chl *a* concentration was over 300% greater in embayments (7.2µg/l) than in the other habitats ( $p < 0.0001$ ) (Figure 3a). Nearshore and offshore habitats were similar with respect to chl *a* ( $p = 0.65$ ), with concentrations seldom exceeding 3µg/l. Seasonal TP concentration in embayments (16.2µg/l) was more than twice as high ( $p < 0.0001$ ) as in nearshore (7.1µg/l) and offshore sites (6.7µg/l), where concentrations were rarely greater than 10µg/l (Figure 4a). Seasonal SRP was nearly twice as high ( $p < 0.0001$ ) in embayments (5.0µg/l) as in nearshore (2.8µg/l) and offshore sites (2.6µg/l) (Figure 5a). Mean May through October embayment water temperature (19.3°C) was significantly higher ( $p = 0.008$ ) than nearshore temperature (17.6°C) (Figure 6). Seasonal (May – Oct) mean values for each parameter are summarized in Table 1. Long-term trends in Secchi depth, chl *a*, TP and SRP are presented in Figures 2b, 3b, 4b, and 5b.

### *Zooplankton density, biomass, and mean length.*

In 2008, May through October zooplankton density ( $p < 0.001$ ) and biomass ( $p < 0.001$ ) were significantly greater in embayments (140.5/L, and 177.8µg/L respectively) than in nearshore sites (12.0/L and 21.4µg/L respectively) (Table 2, Figure 7a & c). Average size of zooplankton was significantly smaller ( $p < 0.01$ ) in embayment sites (0.42mm) than in nearshore sites (0.47mm) (Table 2, Figure 7b), a finding consistent with previous years.

During July-August, the average embayment zooplankton density (151.7/L) and biomass (195.7µg/L) were significantly greater than the density (10.1/L) and biomass (22.3µg/L) of nearshore samples, but the embayment average size (0.42mm) was significantly less ( $p < 0.05$ ) than the average size of nearshore samples (0.52mm;

Table 3). The average size of zooplankton in offshore samples (0.60mm) was significantly greater than the average size of nearshore samples ( $p < 0.05$ ) and embayment samples ( $p < 0.001$ ). Offshore zooplankton biomass (17.6µg/L) was significantly different than biomass of nearshore and embayment samples (Table 3). Zooplankton density of offshore samples (6.9/L) was not significantly different than density of nearshore samples (10.1/L) but was significantly less than embayment samples (151.7/L).

### Zooplankton Community Dynamics

*Seasonal (May – Oct).* In 2008, three zooplankton taxa differed significantly in proportion of total biomass between embayments and nearshore sites (Table 2; Figure 8). Daphnids and cyclopoid copepods had a significantly greater proportion of biomass in embayments ( $p < 0.05$ ), while calanoid copepods had a significantly greater proportion in nearshore habitats ( $p < 0.001$ ). The proportion of *C. pengoi* biomass was on average greater in the nearshore sites (4% of total zooplankton biomass) than embayment sites (less than 1% of total zooplankton biomass), but the difference was not statistically significant (Table 2).

*Summer.* In July-August there were eleven notable differences between components of the zooplankton community in the three different habitats (Table 3). Bosminid biomass at offshore sites (1.13µg/L) and nearshore sites (2.70µg/L) were significantly lower than bosminid embayment biomass (41.2µg/L) ( $p < 0.001$ ). Other cladoceran biomass at embayment sites (53.5µg/L) was greater than offshore sites (9.65µg/L) and nearshore sites (5.64µg/L) ( $p < 0.001$ ). Daphnid biomass at embayment sites (27.1µg/L) was greater than offshore sites (0.67µg/L) and nearshore sites (2.29µg/L) ( $p < 0.01$ ). Calanoid copepod biomass at nearshore sites (8.93µg/L) was greater than offshore sites (1.58µg/L) ( $p < 0.01$ ). Cyclopoid copepod biomass at embayment sites (68.3µg/L) was greater than offshore sites (1.90µg/L) and nearshore sites (0.67µg/L) ( $p < 0.001$ ). In addition, the total community density and biomass was greater at embayment sites (151.7/L and 195.7µg/L, respectively) than offshore sites (6.91/L and 17.6µg/L, respectively) and nearshore sites (10.1/L and 22.3µg/L, respectively) (Table 3). *Cercopagis pengoi* biomass was significantly

higher at offshore sites (2.15µg/L) and nearshore sites (1.43µg/L), compared to embayment sites (0.26µg/L) ( $p<0.001$ , and  $p<0.02$ , respectively) (Table 3).

*Cercopagis pengoi*. In 2008, *C. pengoi* was present in 61 samples (50%). In offshore habitats, it was detected in 24 samples (60%), and the maximum biomass of *C. pengoi* from a single offshore sample was 7.01µg/L at Main Duck on July 14, 2008. In nearshore habitats, *C. pengoi* was detected in 25 samples (50%), and the maximum biomass of *C. pengoi* from a single nearshore sample was 7.67µg/L at Galloo Island-Lake on July 14, 2008. In embayments, *C. pengoi* was detected in 12 samples (39%), and the maximum biomass from a single embayment sample was 2.21µg/L Chaumont Bay-Bay on July 14, 2008. During its July peak biomass, *C. pengoi* mean length was 1.27mm, 1.22mm, and 1.24mm in offshore, nearshore, and embayment habitats, respectively (Table 4).

*Bythotrephes longimanus*. The May to October frequency of detection in 2008 for *B. longimanus* was 40 samples (33%). In offshore habitats, *B. longimanus* was detected in 14 samples (35%) and its maximum biomass at a single offshore site was 11.8µg/L at Smoky Point on October 14, 2008. In nearshore habitats, *B. longimanus* was detected in 17 samples (34%), and its maximum biomass at a single nearshore site was 15.2µg/L at Sodus Bay-Lake on October 10, 2008. In embayments, *B. longimanus* was detected in 9 samples (29%), and the maximum biomass of *B. longimanus* from a single embayment site was 1.15µg/L at Sodus Bay-Bay on October 10, 2008. During its peak biomass in Sept-Oct, *B. longimanus* mean length was 2.51mm, 2.50mm, and 2.24mm in offshore, nearshore and embayment habitats, respectively (Table 5).

#### Temporal Variability and Power Analysis

Power analysis was performed for spring TP (1995 – 2008), seasonal SRP (1995 – 2008), summer chl *a* (1995 – 2008), and summer zooplankton biomass (1998 – 2008). Each habitat (embayment, nearshore, and offshore) was analyzed separately. An effect size of 5% (5% per year decline) was used for each parameter, an amount that represents a 37% reduction over 10 years. In the case of summer zooplankton biomass, an effect size of 10% per year was used

as well, an amount that represents a 61% reduction over 10 years.

Spring TP (1995 – 2008) showed declining trends in all three habitats. Linear regression on  $\log_e$  transformed data showed declines of 2%, 1%, and <1% per year in embayment, nearshore, and offshore habitats, respectively, but none of the trends were significant ( $p=0.33$ ,  $p=0.51$ , and  $p=0.77$ ). Using an effect size of 5% per year, the observed variance,  $\alpha=0.05$  and  $\text{power}=0.90$ , it would take 17, 19, and 18 years to gain the ability to detect a 5% decline per year in each habitat, respectively. Mean seasonal SRP concentrations (1995 – 2008) increased in all habitats. Linear regression on  $\log_e$  transformed data showed increases of <1%, 2%, and 5% per year for embayment, nearshore, and offshore habitats, respectively. However, the trends were not significant ( $p=0.84$ ,  $p=0.64$ , and  $p=0.22$ ). Using an effect size of 5% per year, the observed variance,  $\alpha=0.05$  and  $\text{power}=0.90$ , it would take 18, 17, and 18 years to gain the ability to detect a 5% per year increase in SRP in each habitat, respectively.

For summer chl *a* (1995 – 2008), nearshore concentrations increased while embayment and offshore concentrations declined. Linear regression on  $\log_e$  transformed data showed an increase of 5% per year in nearshore habitats, and declines of 3% and <1% per year in embayment and offshore habitats, respectively. The nearshore trend was significant ( $p=0.0003$ ), but the embayment and offshore trends were not ( $p=0.36$  and  $p=0.61$ , respectively). Using an effect size of 5% per year, the observed variance,  $\alpha=0.05$  and  $\text{power}=0.90$ , it would take 23 years in the embayment habitat and 16 years in the offshore habitat to gain the ability detect a significant decline of 5% per year.

Summer zooplankton biomass (1998 – 2008) declined in each habitat (Figure 9). Linear regression on  $\log_e$  transformed data showed declines of 4%, 6%, and 15% per year in embayment, nearshore, and offshore habitats, respectively. Embayment and nearshore trends were not significant ( $p=0.37$  and  $p=0.29$ , respectively), but the offshore trend was significant ( $p=0.03$ ). Using the observed variance,  $\alpha=0.05$  and  $\text{power}=0.90$ , it would take 24 and 30 years to detect a 5% decline per

year in embayment and nearshore habitats, respectively. In order to gain the ability to detect a 10% decline, it would take 16 and 20 years of sampling in embayment and nearshore habitats, respectively.

To permit more timely management actions, it can be argued that a less stringent (e.g.,  $\alpha=0.10$ ) significance criterion be used in the power analysis so that trends can be detected “earlier”, but power estimates are relatively insensitive to changes in this parameter (Hatch 2003). Using  $\alpha=0.1$ , the time needed to gain the ability to detect change is reduced by only 1-3 years in the case of all parameters examined.

We also determined what our power is to detect a 5% effect size (5% per year change) given the number of years of data currently available. For spring TP, power to detect a 5% decline over 14 years is 0.77, 0.63, and 0.68 in embayment, nearshore, and offshore habitats, respectively. For summer chl *a*, power to detect a 5% decline over 14 years is 0.80 in the offshore habitat and 0.47 in the embayment habitat. For summer zooplankton biomass, power to detect a 5% decline over 11 years is 0.28 and 0.20 in embayment and nearshore habitats, respectively.

## **Discussion**

### The Importance of Lower Food Web Monitoring

Lake managers recognize the importance of using ecological indicators to assess management actions and restoration efforts. Information on ecological indicators, such as the biennial State of the Lake Ecosystem Conference (SOLEC) reports, stress the utility of indicators in evaluating ecosystem status and highlight the need for long-term data. Collecting data from spring through fall is important because seasonal patterns inform selection of data for trend analyses (e.g. regression and power analysis), allowing us to select which parameters are the best ecological indicators (spring TP, summer chl *a*, summer zooplankton). Acquiring data from spring (pre-stratified conditions), summer (stable stratified conditions), and fall (after overturn) is desirable to get a complete picture of lower food web dynamics. Spring is the time of isothermal water temperatures and provides the initial chemical conditions for the year prior to significant uptake of nutrients by the biota. Spring conditions also

provide a picture of the amounts of nutrients available for the biological activity that will occur during the year, making spring TP a logical indicator choice. The summer stratified period characterizes the peak production period for many species; therefore, summer chl *a* and zooplankton biomass were chosen as indicators. The fall period is useful to track species such as *Bythotrephes* whose biomass peaks later in the year.

The lower food web provides the basic necessities – nutrients and food – in freshwater lakes to support and sustain higher trophic levels that include fish. Total phosphorus is related to trends in nutrient status and fertility and chl *a* provides a measure of algal food resources potentially available to zooplankton. In turn, zooplankton are critical food for zooplanktivores like the alewife (Mills et al. 1992, 1995), and alewife are the primary prey of salmonines (Brandt 1986; Lantry 2001). Therefore, zooplankton stocks, juvenile alewife abundance and growth, and growth of young salmon and trout are tightly linked (O’Gorman et al. 1987, 1997). In Lake Ontario, growth of age-1 alewives is dependent on epilimnetic zooplankton and the number of competing age-0 alewives (O’Gorman et al. 1997). Consequently, the link of the youngest cohorts of alewife to zooplankton is important, as these fish represent the bulk of alewife production in Lake Ontario and are critical to supporting a substantial piscivore sport fishery. Alewife numbers declined through the 1990s and shifted their spring depth distribution following the establishment of *Dreissena* spp. (O’Gorman et al. 2000, 2004). Both *Cercopagis* and *Bythotrephes* are predators on zooplankton and, in turn, are eaten by alewife (Mills et al. 1992; Bushnoe et al. 2003). Because these two invasive zooplankton species are large bodied crustaceans and their abundance is inversely related to alewife abundance, they can provide an excellent biological signal of alewife population status.

### Habitat Comparisons

Embayments remained the most productive habitat in 2008 with highest zooplankton density and biomass, chl *a*, TP, and SRP. Nearshore and offshore habitats were similar with the exception of zooplankton size and zooplankton community composition; zooplankton mean length was greater in the offshore in summer due to the presence of *C. pengoi*, and *B. longimanus*. One

might expect nearshore areas to have higher levels of TP, chl *a*, and zooplankton biomass. In this study, we have not observed significant differences in the aforementioned parameters when comparing sites at 10m depth with much deeper offshore waters. However, Makarewicz et al. (in review) observed significantly higher nutrients levels over depths <5m compared to the offshore. Embayment habitats continue to be nutrient rich while TP concentrations in nearshore and offshore habitats remain below the target of 10ug/L set by the Great Lakes Water Quality Agreement.

#### Zooplankton Community Dynamics

Nutrients, food quality, invasive species, and predation can all affect zooplankton abundance, biomass, and community structure. Phosphorus is a key element necessary for growth of plants and animals, and low phosphorus concentrations limit production in freshwater systems. Lake Ontario's phosphorus levels have been declining since the early 1970s (Figure 4b) and declines in zooplankton would be expected. Johannsson (2003) reported that the Lake Ontario zooplankton community responded to declines in TP with changes in both species composition and biomass. *Chydorus sphaericus* and *Ceriodaphnia lacustris*, species preferring nutrient-rich conditions, declined or disappeared in the early 1990s subsequent to declines in phosphorus. Offshore epilimnetic summer zooplankton biomass has decreased 13% per year since the early 1980s (Figure 10).

Low phosphorus concentrations and high water clarity in Lake Ontario in 2008 may impact food quality for zooplankton by altering C:N:P ratios. Because elemental phosphorus is required for rapid growth and reproduction, productivity of zooplankton species such as large daphnids declines when algal C:P content is high. Low P content in algal food has been demonstrated to reduce growth rate and productivity of daphnids and other fast-growing species (Stern and Schulz 1998, Stern and Elser 2002). Continued oligotrophication of Lake Ontario will impact the food supply to zooplankton, and the microbial food web will become a more important pathway of energy transfer to zooplankton and to fish.

Zooplankton biomass can also decline as a result of predation by planktivorous fish, mysids or

predatory zooplankton, particularly the recent invaders *Cercopagis pengoi* and *Bythotrephes longimanus*. *Bythotrephes* biomass in offshore habitats was at an all time high in 2008 (Table 5); biomass in nearshore and embayment habitats was lower than in 2007, but comparable to 2006. Pangle et al. (2007) demonstrated that *Bythotrephes* decreased the productivity of *Daphnia retrocurva* and *Bosmina* spp. in Lakes Michigan and Erie. While we do not have productivity estimates for *Daphnia*, the proportion of daphnid biomass decreased in nearshore areas in late summer and late fall—time periods when *Bythotrephes* biomass increased (Figure 8). Warner et al. (2006) observed significant declines in abundance of bosminids, *Diacyclops thomasi*, and copepod nauplii during peak *Cercopagis* abundance in nearshore Lake Ontario. This pattern was evident in 2008 as well; the proportion of nearshore bosminid biomass declined in the summer as the proportion of *Cercopagis* increased (Figure 8). Additional evidence of *Cercopagis* effects on small zooplankton include (1), Benoit et al. (2002) who found that invasion by *Cercopagis* was correlated with a decline in juvenile epilimnetic cyclopoids, and that production of juvenile copepods decreased both through direct predation and through a shift in copepod vertical distribution to colder waters and (2), Laxson et al. (2003) who found that *Cercopagis* fed on small-bodied zooplankton (*D. retrocurva* and *B. longirostris*) in laboratory experiments, and noted a decline in the abundance of *D. retrocurva*, *B. longirostris*, and *Diacyclops thomasi* between 1999 and 2001 coinciding with an increase in abundance of *Cercopagis*.

#### Temporal Variability and Power Analysis

Of the twelve parameters evaluated for long-term trends, two were significant ( $p < 0.1$ ), summer chl *a* in the nearshore (increase) habitat and summer zooplankton in the offshore habitat (decrease). Results of power analyses on the remaining eight chemical parameters suggest that an additional 3 – 9 years of data would be needed in order gain the ability to detect a 5% change with high power (0.9). For zooplankton in nearshore and embayment habitats, an additional 5 – 19 years of data would be needed to gain the ability to detect the same effect. If a less stringent significance criterion ( $\alpha = 0.1$  as opposed to  $\alpha = 0.05$ ) is used in the power analysis, the time needed to

gain the ability to detect change is reduced by 1-3 years.

Decline rates observed for epilimnetic summer zooplankton biomass (1998 – 2008) in embayment, nearshore, and offshore habitats were biologically significant (4 – 15% per year), but high interannual variability limits our power to detect statistically significant declines. If more precise annual estimates could be obtained, interannual variability should decrease and power to detect change would increase. Our power to detect a 5% change over 14 years in spring TP, seasonal SRP, and summer chl *a* is high, but the same is not true for zooplankton. Power to detect a 5% change in summer zooplankton biomass over 11 years is 0.28 for embayment habitats and 0.2 for nearshore and offshore habitats. The general goal is to achieve a minimum power level of 0.8 (Cohen 1988). Precision of annual estimates could be increased by collecting replicate tows at each site and pooling the samples in the field.

#### Balance Between Lower Food Web and Fish

Most planktivorous fish are size-selective predators, and the mean length of zooplankton is an indication of the abundance of planktivores in the system. Zooplankton mean length has been used as an index of the balance between plankton eating fish and fish predators (Mills and Schiavone 1982). In lakes where planktivory is high, smaller species like *Bosmina* dominate and zooplankton mean length is small. When healthy populations of quality-sized piscivores are present, populations of planktivores are controlled, larger-bodied zooplankton are more abundant, and mean body lengths of crustacean zooplankton are greater than 0.8-1.0 mm (Mills and Schiavone 1982). The dominance of small crustacean zooplankton, on the other hand, points to the absence of sufficient number of predators to suppress planktivore abundance.

Alewife (*Alosa pseudoharengus*) is the dominant planktivore in Lake Ontario, and intense planktivory by these fish has historically structured the zooplankton community toward small species like *Bosmina*. Zooplankton are the principal food of juvenile and adult alewife and these fish can account for more than 96% of predation on zooplankton in the offshore (Rand et al. 1995). Alewife abundance declined 42% from

the early 1980s to the early 1990s (O’Gorman et al. 2000), and changes in the zooplankton community were observed in response to this decline. Among the more significant changes during this period were a shift to larger zooplankton species and increased abundance of summer cyclopoid copepods.

In Lake Ontario, the impact of planktivory by alewife may be greater at one time of the year than another. For instance, adult alewives migrate inshore in the spring and return to the offshore through July. Consequently, one would expect that the impact on offshore zooplankton by adult alewife would be highest in epilimnetic waters sometime mid-summer through early fall. For the period mid- July through mid-August 2008, we found that offshore zooplankton mean length was 0.60mm, suggesting either a balance between piscivores and alewife or a dominance of piscivores. We lean toward the latter explanation as 2008 populations of adult and juvenile alewife apparently were not sufficiently abundant to suppress larger sized zooplankton species like *Cercopagis pengoi* and *Bythotrephes longimanus*. Finally, lakes with large populations of alewife typically have zooplankton that are very small (0.3 to 0.4 mm, CBFS unpublished data), whereas offshore crustacean zooplankton in Lake Ontario during mid summer to early fall were considerably larger, averaging 0.60mm in length. The mechanism behind larger mean size could be the result of a decrease in smaller zooplankton associated with predation by *Cercopagis* and *Bythotrephes* (Warner et al. 2006), which is an indirect effect of decreased alewife abundance.

The relationship between zooplankton average size and alewife abundance is complicated by the presence of the invasive predatory cladocerans, *Cercopagis pengoi* and *Bythotrephes longimanus*. *C. pengoi* possesses a tail spine several times its body length, preventing young-of-the-year planktivores from preying upon it until their gape is large enough to accommodate the spine (Bushnoe et al. 2003). In Lake Ontario, *Cercopagis* abundance is inversely related to bosminid density so declines in *Bosmina* could contribute to a shift in the mean size of zooplankton toward larger organisms. However, *C. pengoi* is a moderately large sized cladoceran and adult alewife are known to feed on it. *C. pengoi* was present in 50% of Lake Ontario

zooplankton samples collected in 2008, providing additional circumstantial evidence that adult alewife numbers were not sufficient to suppress its occurrence. *Bythotrephes* was detected in Lake Ontario in only one sample in 2002, none in 2003, and in only three samples in 2004. *Bythotrephes* is occurring more frequently (numbers of samples collected each year are similar) in recent years (35 samples in 2005, 47 samples in 2006, 38 samples in 2007, and 40 samples in 2008) providing yet one more line of evidence that adult alewife abundance is currently suppressed in Lake Ontario. Interestingly, the increase in *Bythotrephes* and *Cercopagis* and associated food web disruption has not led to a decrease in alewife growth rate or condition (O’Gorman et al. 2008), perhaps due to the density dependent effect of low alewife abundance.

#### Significant Findings

The biomonitoring dataset is becoming sufficiently long-term that some trends are now emerging. For example, zooplankton biomass in the offshore epilimnion of Lake Ontario has been significantly declining (linear regression,  $p=0.03$ ) since the late-1990s at a rate of 15% per year (80% reduction, 1998 - 2008). Long-term data (1981 - 2008) show a similar pattern. For example, offshore summer epilimnetic zooplankton density declined 13% per year during that time (Figure 10) resulting in a 99% reduction over 28 years; zooplankton densities in 2008 are 1% of the levels found three decades ago. Dwindling zooplankton populations mean fewer food resources for alewife, and food-deprived alewife can lead to fewer of these fish to sustain a significant salmonine population. The fall biomass of large-bodied *Bythotrephes longimanus* in field samples was the highest on record since the Lake Ontario biomonitoring program began. This combined with the high frequency of occurrence (33%) of *Bythotrephes* provides yet one more signal that alewife populations are depressed. Increasing trends in both SRP (although not significant,  $p=0.05$ ) and chl *a* ( $p=0.0003$ ) in the nearshore suggest possible linkages to zooplankton and *Dreissena*. Increases in SRP have been noted in other freshwater lakes (Oneida Lake, for example) and may be associated with a complex of interactions tied to sediment-dreissenid mussel interactions or to the release of iron-bound phosphorus in the guts of *Dreissena* (Hecky 2004). At this point, we do not

know what is causing the 5% per year increase in nearshore chl *a* but two factors are likely involved including increased availability of SRP and reduced grazing by a declining zooplankton population.

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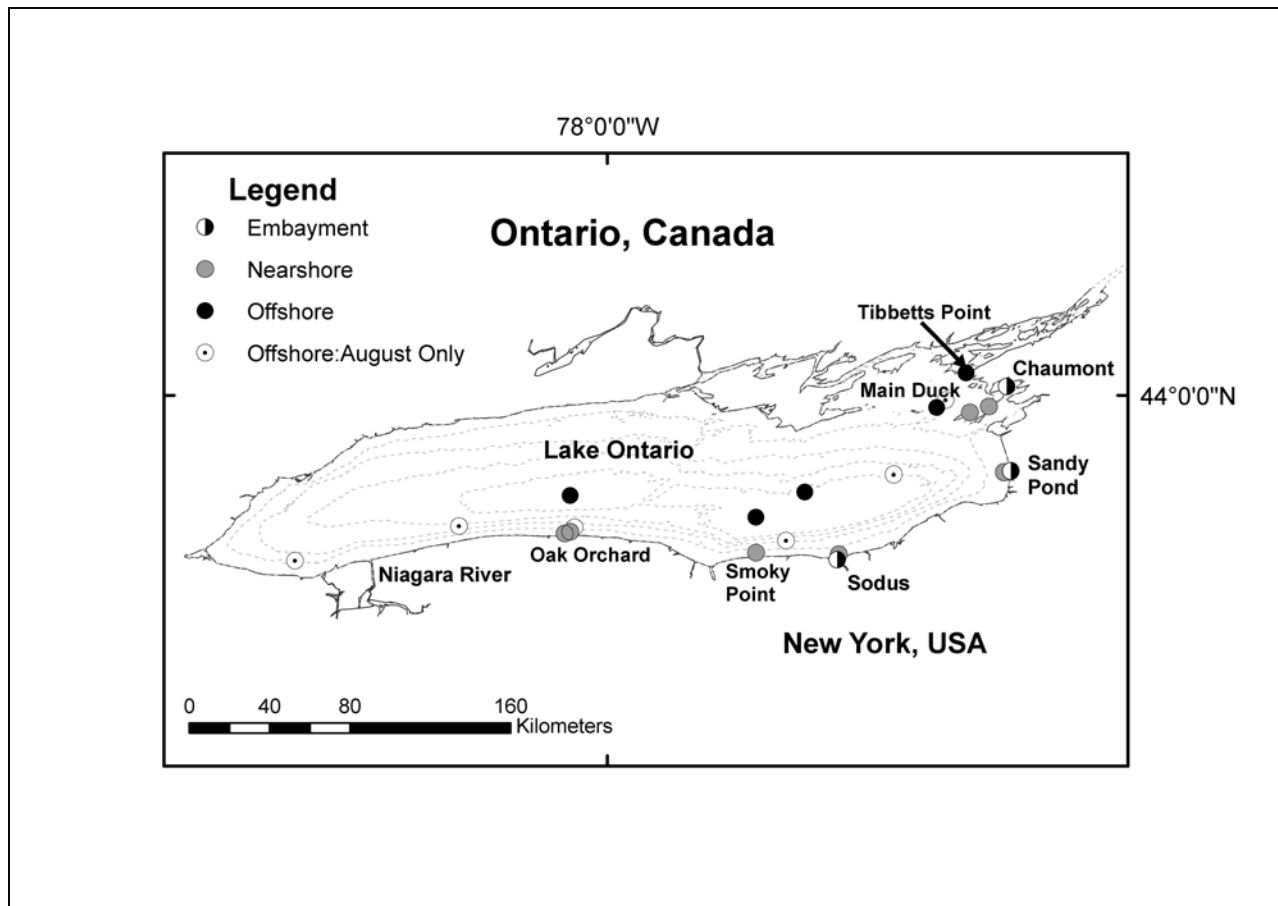


Figure 1. Map of biomonitoring program sites, 2008.

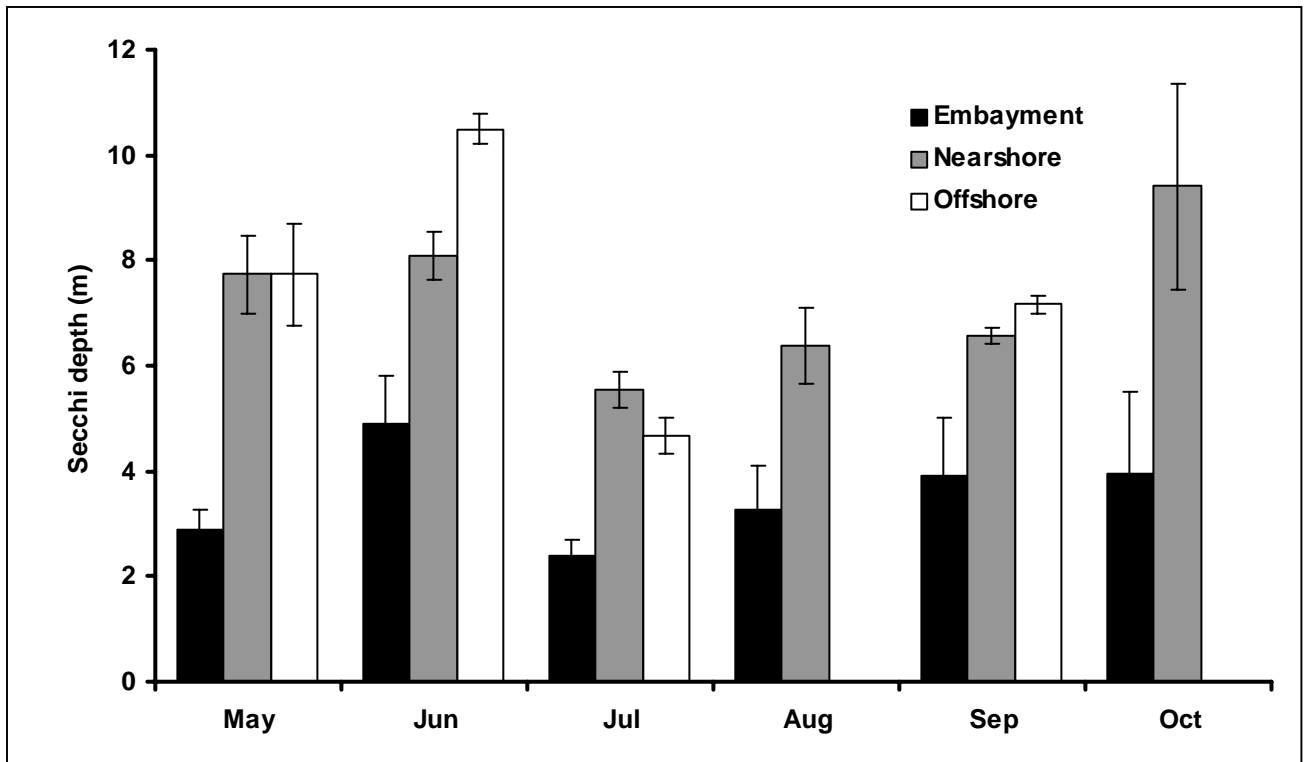


Figure 2a. Mean monthly Secchi depth (meters) for embayment, nearshore, and offshore habitats in Lake Ontario, May - October, 2008.

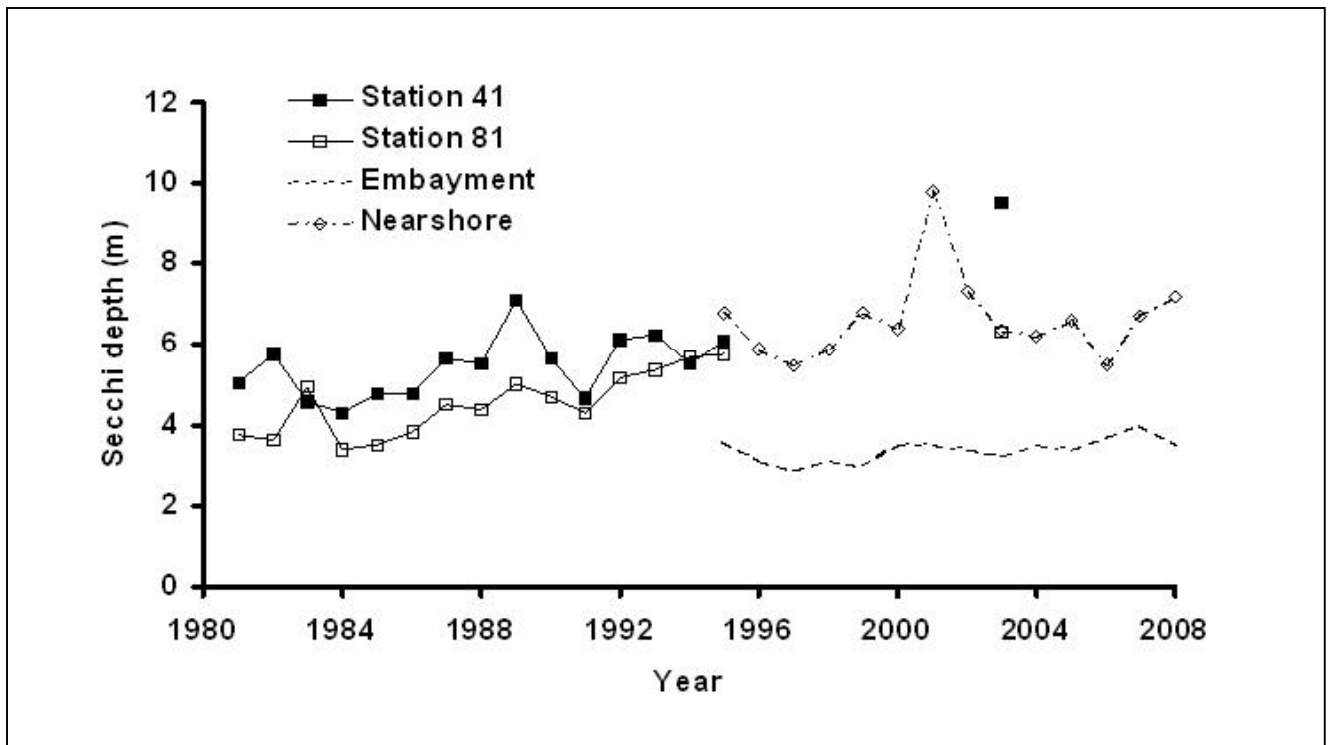


Figure 2b. Long-term mean seasonal (May - October) Secchi depth (meters) in Lake Ontario, 1981 - 2008. Data from 1981 - 1995 are from the Department of Fisheries and Oceans Canada's Bioindex Program. Data from 1995 - 2008 are from the NYSDEC Biomonitoring Program.

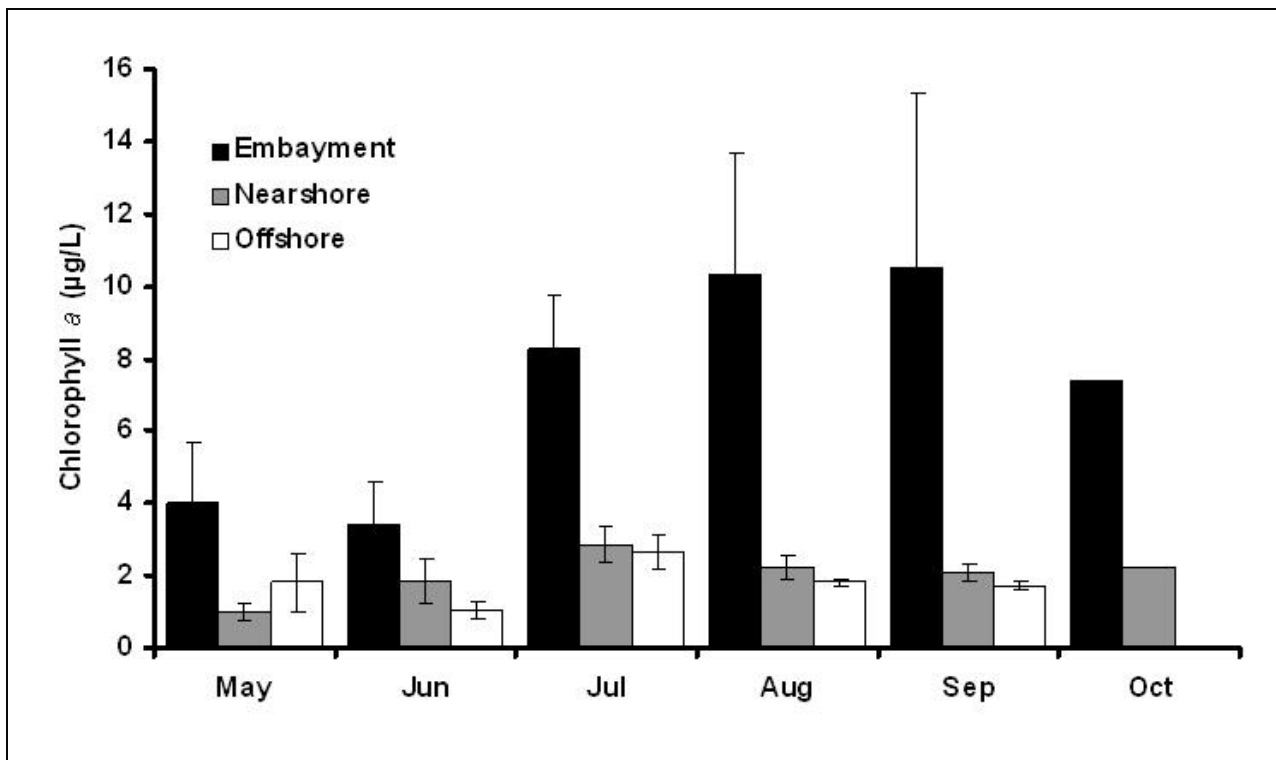


Figure 3a. Mean monthly chlorophyll *a* concentrations for embayment, nearshore, and offshore habitats in Lake Ontario, May - October, 2008.

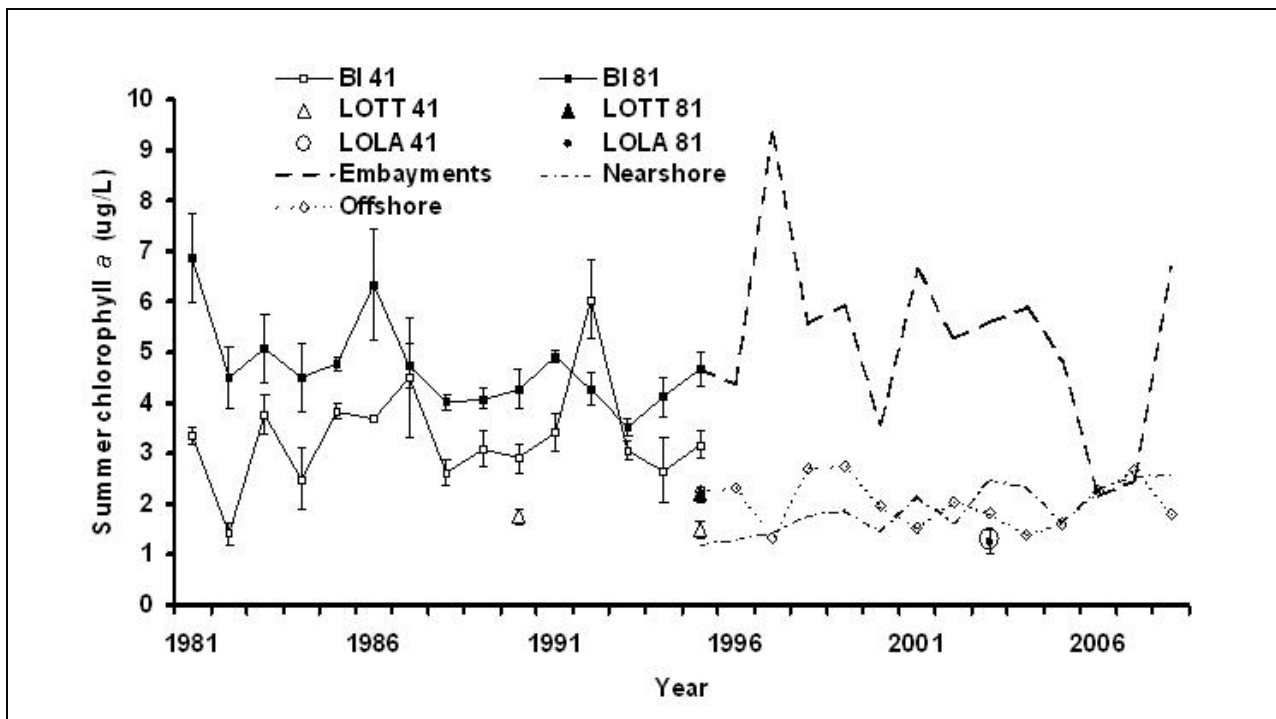


Figure 3b. Long-term summer chlorophyll *a* concentrations in Lake Ontario, 1981 - 2008. BI 41 and BI 81 are from the Department of Fisheries and Oceans Canada's Bioindex Program. LOTT 41 and LOTT 81 are from the Lake Ontario Trophic Transfer Project. LOLA 41 and LOLA 81 are from the Lake Ontario Lower Food Web Assessment. Data from 1995 - 2008 are from the NYSDEC Biomonitoring Program.

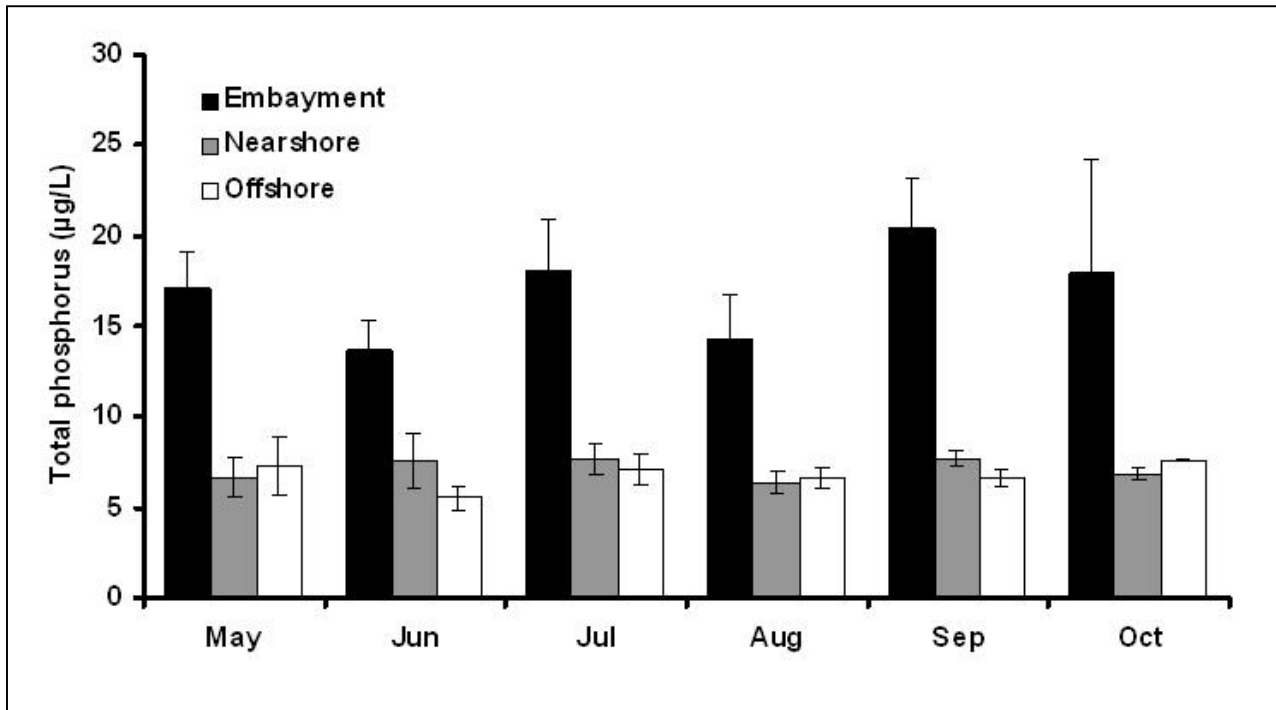


Figure 4a. Mean monthly total phosphorus concentrations for embayment, nearshore, and offshore habitats in Lake Ontario, May - October, 2008.

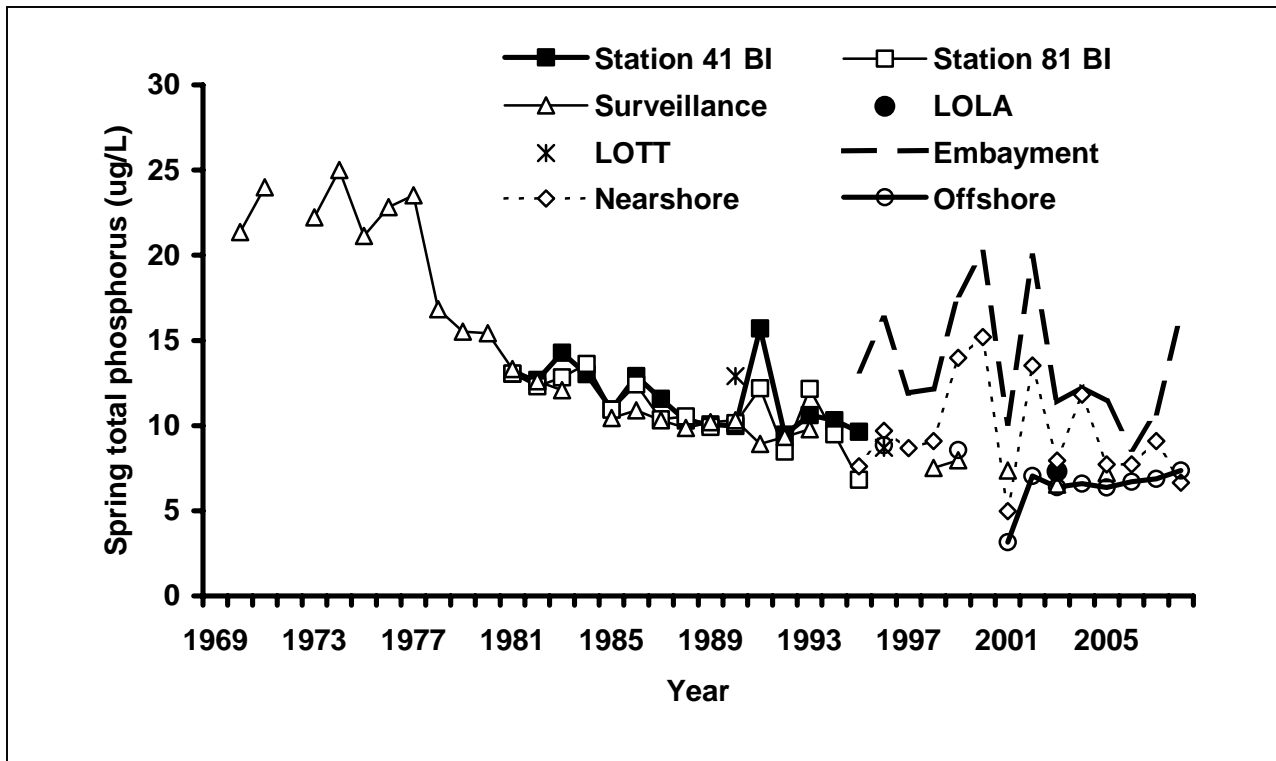


Figure 4b. Long-term spring total phosphorus concentrations in Lake Ontario, 1970 - 2008. Data from 1970 – 2001 are from Environment Canada’s Surveillance Program. Station 41 BI and 81 BI are from the Department of Fisheries and Oceans Canada’s Bioindex Program. LOTT data is from the Lake Ontario Trophic Transfer Project. LOLA data is from the Lake Ontario Lower Food Web Assessment. Data from 1995 – 2008 are from the NYSDEC Biomonitoring Program.

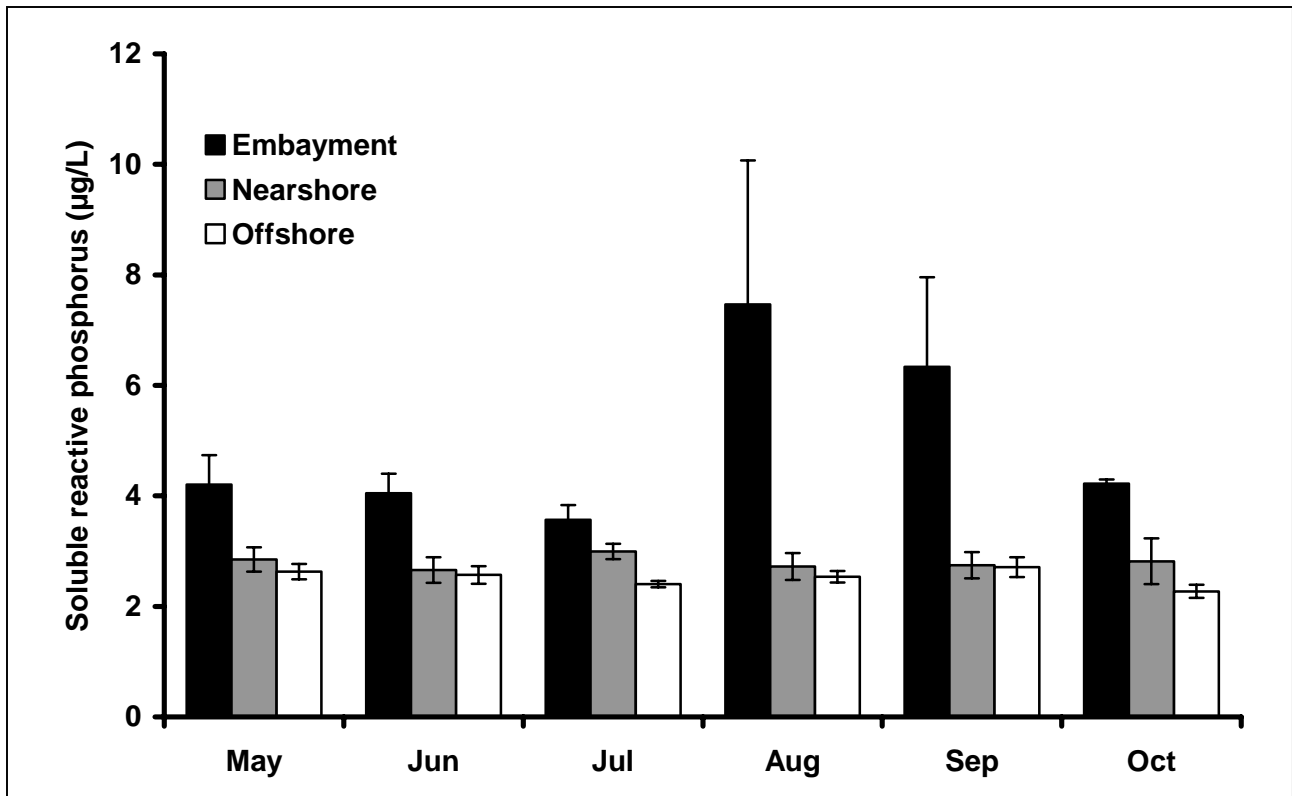


Figure 5a. Mean monthly soluble reactive phosphorus concentrations for embayment, nearshore, and offshore habitats in Lake Ontario, May - October, 2008.

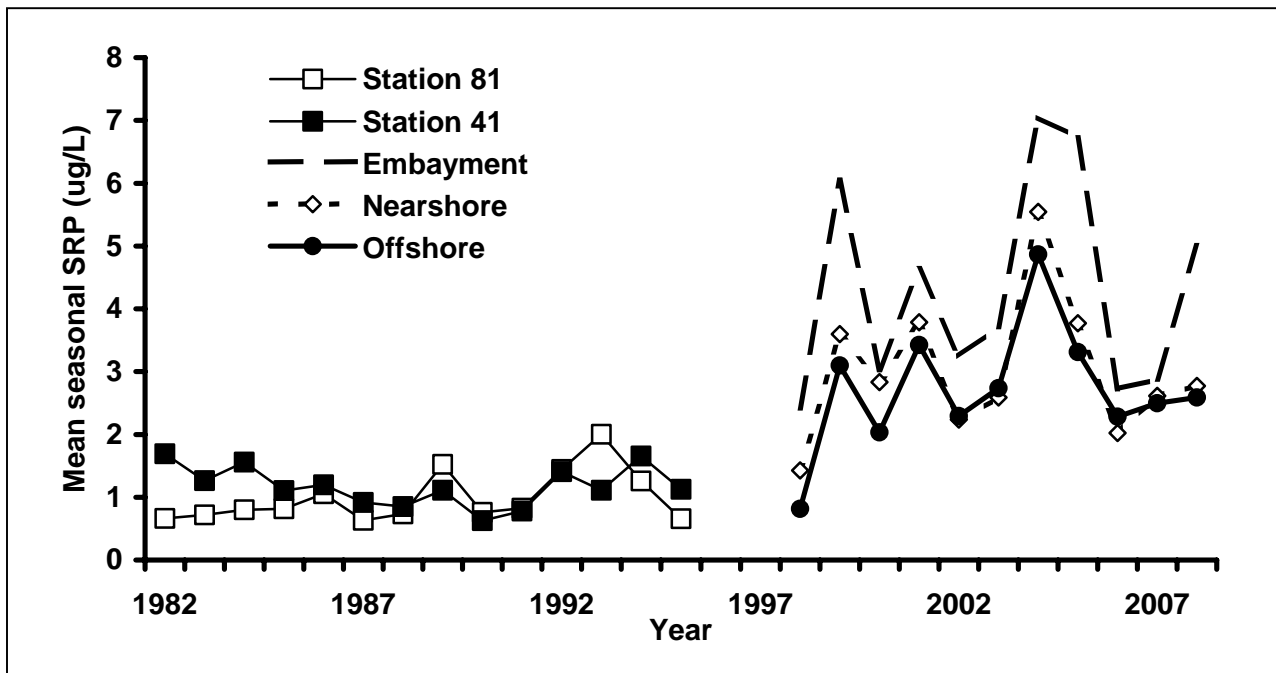
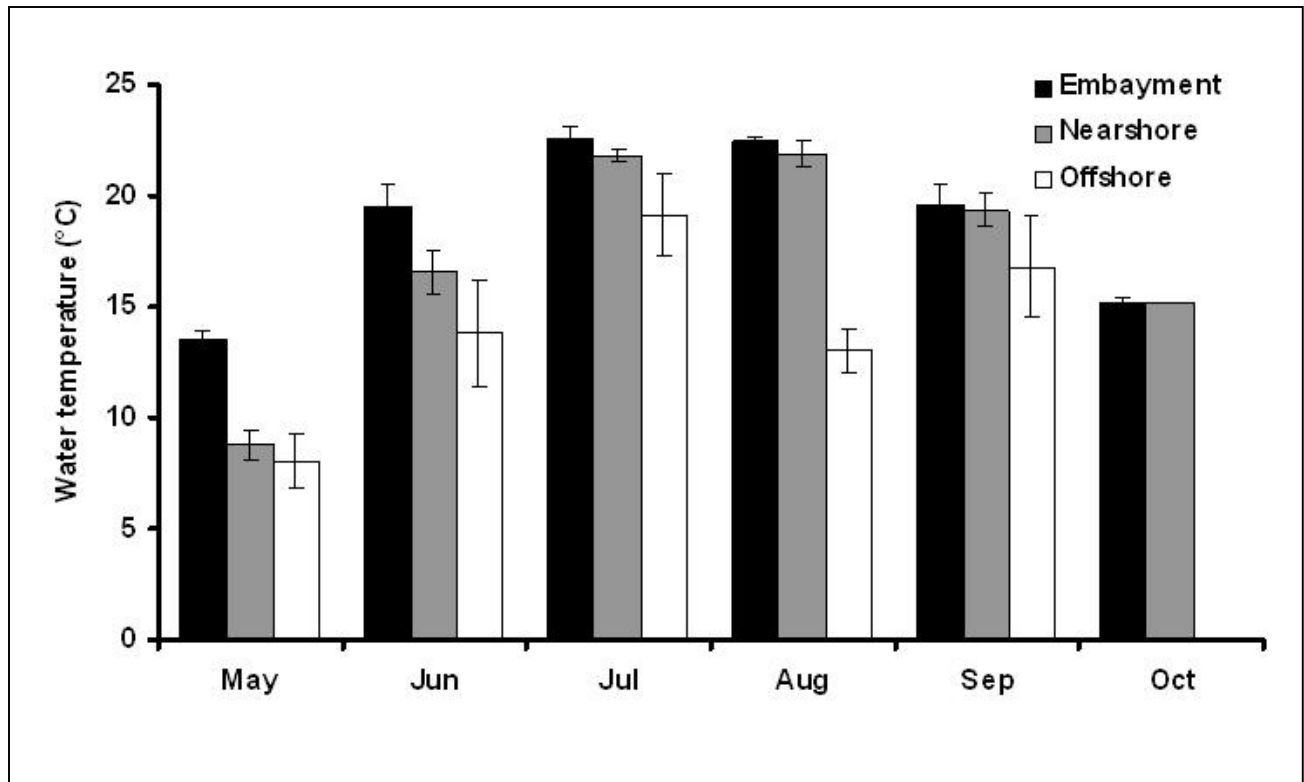


Figure 5b. Seasonal mean soluble reactive phosphorus concentrations in Lake Ontario, 1982 - 2008. Data from 1982 - 1995 are from the Department of Fisheries and Oceans Canada's Bioindex Program. Data from 1998 - 2008 are from the NYSDEC Biomonitoring Program.



**Figure 6. Mean monthly water temperatures in embayment, nearshore, and offshore habitats in Lake Ontario, May - October, 2008.**

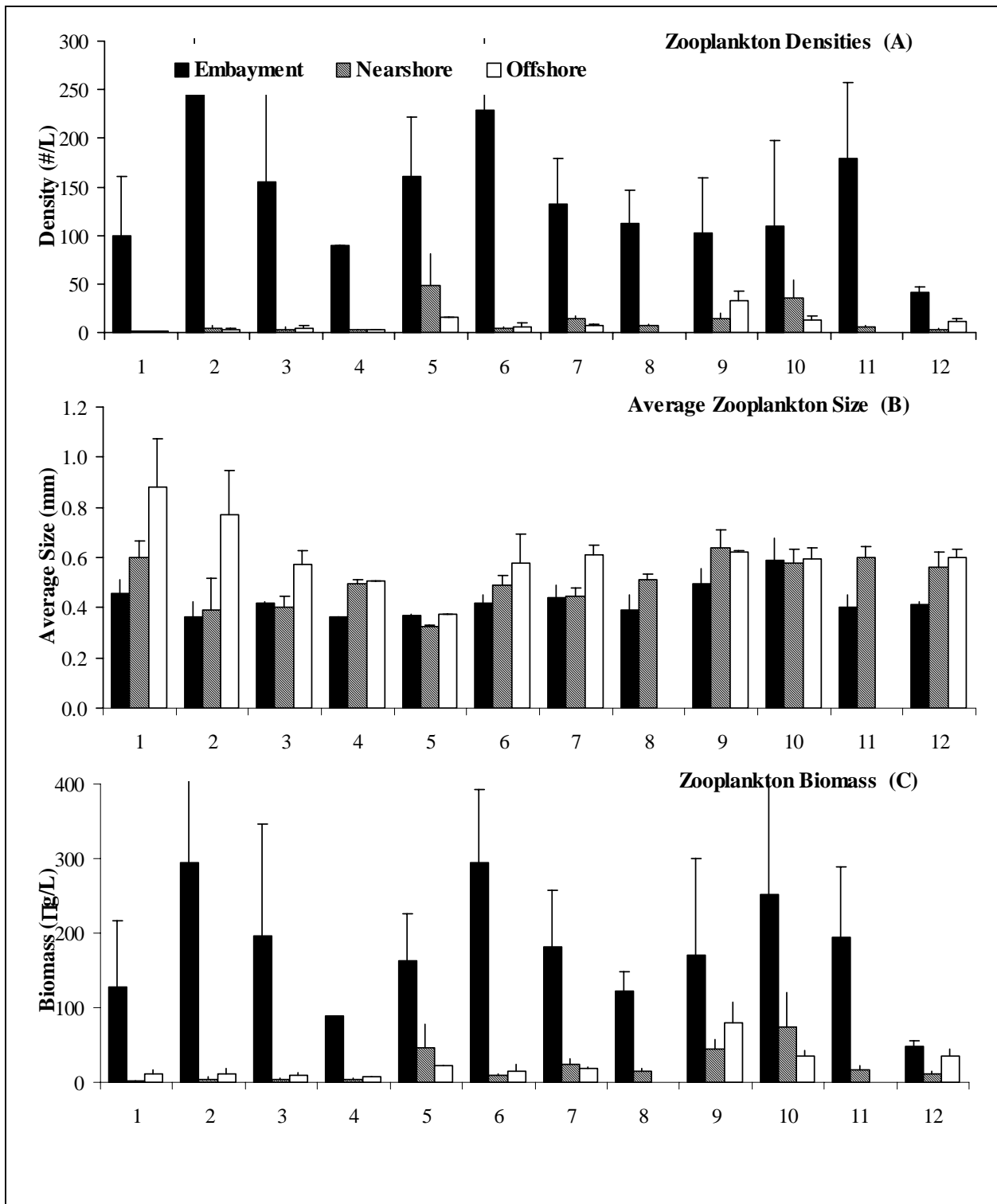


Figure 7. Biweekly means (+ 1SE) of zooplankton densities, sizes, and biomass for May through October 2008 embayment, nearshore, and offshore sites on Lake Ontario. On the x-axis, weeks sampled are designated by numbers 1-12.

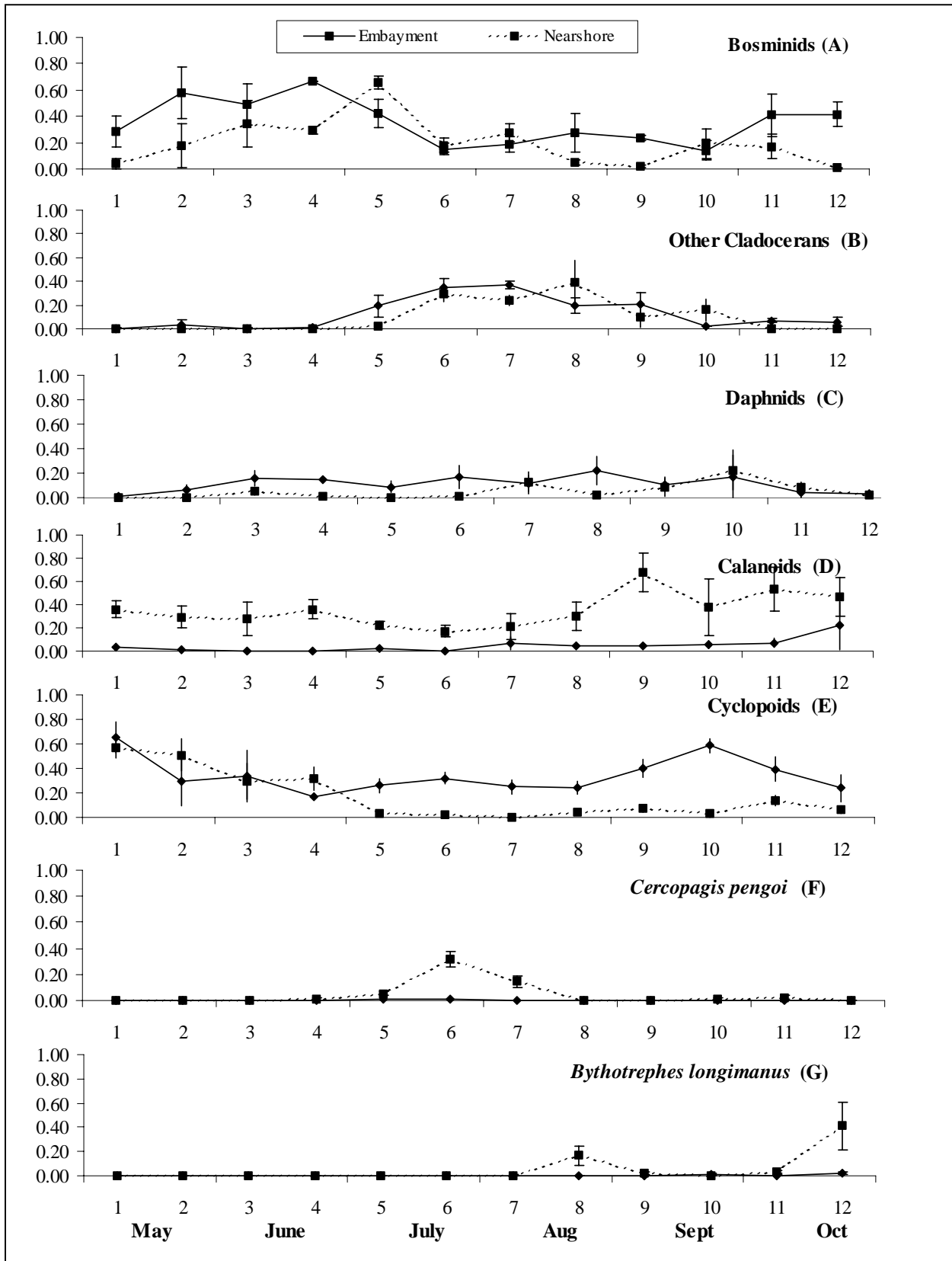


Figure 8. Proportion of total biomass of zooplankton community for embayments and nearshore areas of Lake Ontario, May - October 2008 (weeks 1-12). Zooplankton are grouped as Bosminids (A), Other Cladocerans (B), Daphnids (C), Calanoids (D), Cyclopoids (E), *Cercopagis pengoi* (F) and *Bythotrephes longimanus* (G). Bars represent +1SE.

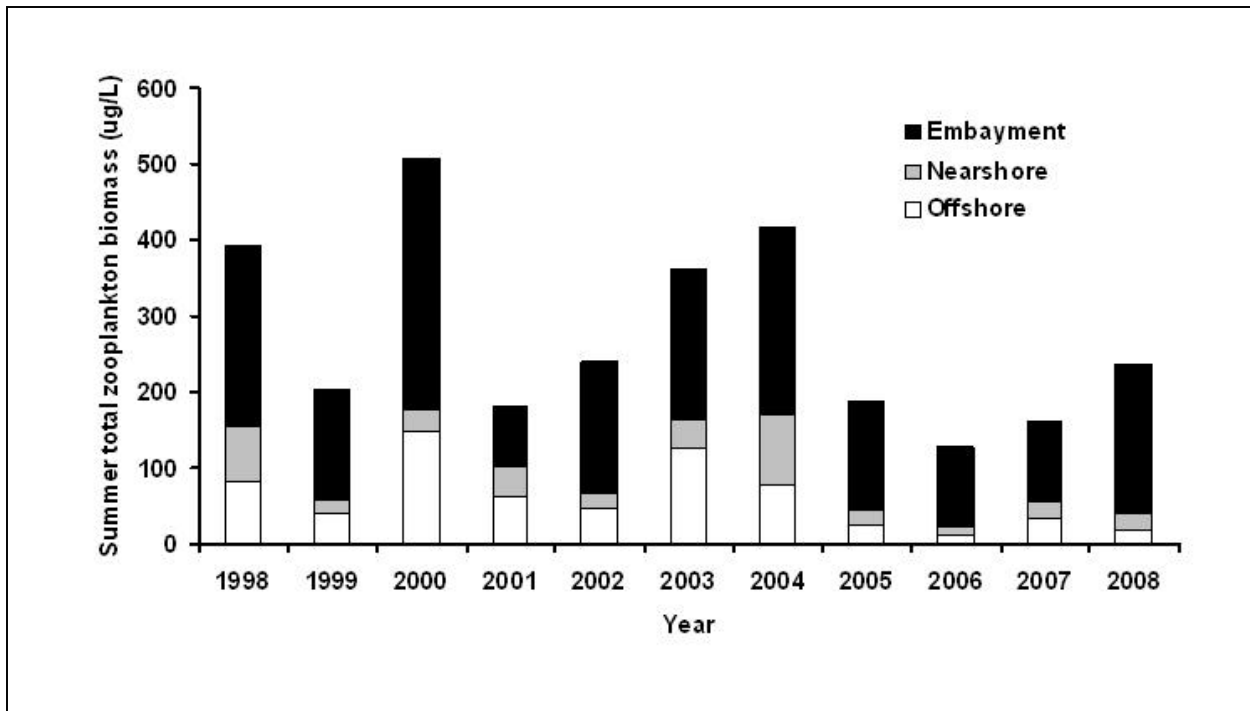


Figure 9. Mean summer total zooplankton biomass in embayment, nearshore, and offshore habitats in Lake Ontario, May - October, 2008.

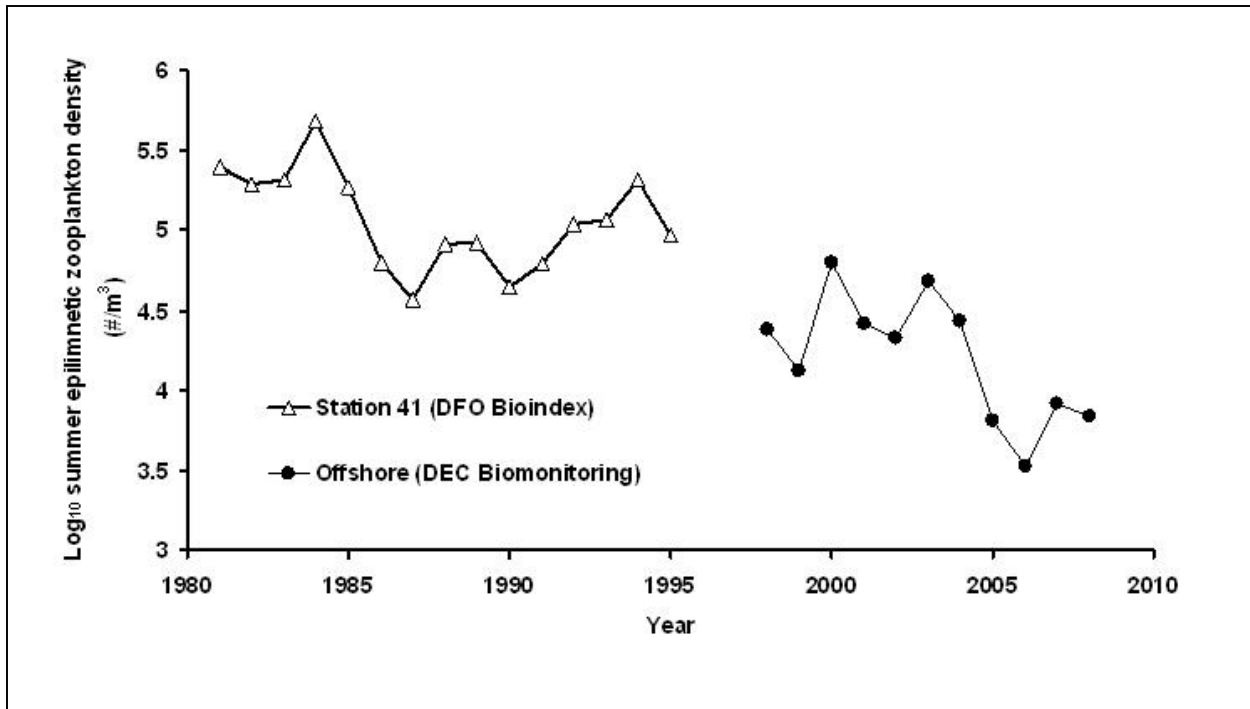


Figure 10. Summer epilimnetic zooplankton density at station 41 (Department of Fisheries and Oceans Canada's Bioindex Program), 1981 - 1995 and at offshore biomonitoring sites, 1998 - 2008.

**Table 1. Mean chl *a*, TP, SRP and water temperature ( $\pm$ SE) for embayment, nearshore, and offshore sites May - October 2008.**

Parameter	Mean $\pm$ SE		
	Embayment	Nearshore	Offshore
Chlorophyll <i>a</i>	7.2 $\pm$ 1.1	2.0 $\pm$ 0.27	1.6 $\pm$ 0.2
Total phosphorus	16.2 $\pm$ 0.92	7.1 $\pm$ 0.33	6.7 $\pm$ 0.27
Soluble reactive phosphorus	5.0 $\pm$ 0.56	2.8 $\pm$ 0.10	2.6 $\pm$ 0.09
Secchi depth	3.5 $\pm$ 0.2	7.1 $\pm$ 0.35	8.7 $\pm$ 0.86
Water temperature	19.3 $\pm$ 1.0	17.6 $\pm$ 1.4	12.7 $\pm$ 1.9

**Table 2. Comparisons of embayments and nearshore sites May-October, 2008 using paired t-tests on biweekly, log (x+1) transformed average values of zooplankton density and biomass, untransformed average size data, and the proportions of total biomass of zooplankton taxa (i.e. Bosminids, "Other" Cladocera, Daphnids, Calanoid copepods, Cyclopoid copepods, *Cercopagis pengoi*, and *Bythotrephes longimanus*). Italics indicate  $p < 0.05$ , bold indicates  $p < 0.001$ . Mean values based on all Lake Ontario sites and dates.**

Parameter	Mean		p-value
	Embayment	Nearshore	
Total zooplankton:			
Density (#/L)	140.46	12.00	<b>1.6E-06</b>
Average Size (mm)	0.42	0.47	<i>0.007</i>
Biomass ( $\mu$ g/L)	177.75	21.43	<b>7.1E-06</b>
Proportion of total biomass:			
Bosminids	0.36	0.24	0.117
Other Cladocera	0.12	0.12	0.977
Daphnids	0.13	0.07	<i>0.018</i>
Calanoid copepods	0.04	0.31	<b>4.8E-04</b>
Cyclopoid copepods	0.34	0.15	<i>0.013</i>
<i>Cercopagis pengoi</i>	0.001	0.043	0.176
<i>Bythotrephes longimanus</i>	0.002	0.056	0.223

**Table 3. Comparisons of offshore (OS), nearshore (NS), and embayment (EM) habitats during July-August 2008 using t-tests assuming unequal variance on biweekly, log (x+1) transformed mean values of zooplankton density and biomass, and untransformed average size data. Mean values for each habitat as well as p-values from the pairwise comparisons are stated for total zooplankton density, average size, biomass, *Cercopagis* biomass, and *Bythotrephes* biomass. Italicized values are significant at Bonferroni adjusted  $\alpha = 0.0167$  ( $p = 0.05$ ), bold values indicate  $p < 0.001$ .**

Parameter	Mean			Pairwise comparisons		
	OS	NS	EM	OS-NS	OS-EM	NS-EM
Total zooplankton:						
Density (#/L)	6.91	10.08	151.69	0.246	<b>3.1E-10</b>	<b>3.6E-10</b>
Average Size (mm)	0.60	0.52	0.42	<i>0.026</i>	<b>5.1E-04</b>	<i>0.020</i>
Biomass ( $\mu\text{g/L}$ )	17.56	22.31	195.68	<b>1.9E-12</b>	<b>4.6E-10</b>	<b>3.1E-07</b>
Zooplankton biomass ( $\mu\text{g/L}$ ):						
Bosminids	1.13	2.70	41.24	0.138	<b>2.2E-07</b>	<b>7.8E-07</b>
Other Cladocera	9.65	5.64	53.54	0.104	<b>9.6E-04</b>	<b>1.6E-06</b>
Daphnids	0.67	2.29	27.06	0.754	<i>0.001</i>	<i>0.002</i>
Calanoid copepods	1.58	8.93	3.43	<i>0.006</i>	0.234	0.169
Cyclopoid copepods	1.90	0.67	68.26	0.128	<b>6.8E-08</b>	<b>2.3E-08</b>
<i>Cercopagis pengoi</i>	2.15	1.43	0.26	0.143	<b>8.5E-04</b>	<i>0.018</i>
<i>Bythotrephes longimanus</i>	0.42	0.50	0.13	0.791	0.342	0.119

**Table 4. *Cercopagis pengoi* density and average size, and biomass in offshore (OS), nearshore (NS), and embayment (EM) habitats during late August 1998, August 1999, July 2000, August 2001, August 2002, July 2003, late July - early August 2004, July 2005, July 2006, July 2007, and mid July - early August 2008.**

Parameter	Year	Mean		
		OS	NS	EM
Density (#/L)	1998	0.471	0.396	0.055
	1999	0.440	0.419	0.120
	2000	0.157	0.371	0.235
	2001	0.210	0.688	0.079
	2002	0.358	0.305	0.000
	2003	0.052	0.467	0.439
	2004	0.435	0.507	0.101
	2005	0.261	0.197	0.181
	2006	1.029	0.638	0.059
	2007	0.739	0.336	0.076
	2008	0.261	0.335	0.057
Average Size (mm)	1998	1.266	1.116	1.057
	1999	1.304	1.154	1.191
	2000	1.149	1.021	1.135
	2001	1.286	1.162	1.388
	2002	1.074	1.040	-
	2003	1.060	0.970	1.076
	2004	1.267	1.124	1.140
	2005	1.295	1.262	1.068
	2006	1.095	1.168	1.097
	2007	1.289	1.098	1.136
	2008	1.265	1.221	1.240
Biomass (µg/L)	1998	3.970	2.721	0.340
	1999	5.277	3.654	1.318
	2000	0.992	2.384	1.872
	2001	1.826	6.119	1.154
	2002	1.748	2.032	0.000
	2003	0.367	2.696	3.141
	2004	3.801	3.972	0.770
	2005	2.448	2.059	1.288
	2006	6.000	5.543	0.450
	2007	6.818	2.531	0.618
	2008	2.155	3.161	0.559

**Table 5. *Bythotrephes longimanus* density, average size, and biomass in offshore (OS), nearshore (NS), and embayment (EM) habitats for September-October 1998-2008<sup>a</sup>.**

<i>Bythotrephes longimanus</i>		Mean		
Parameter	Year	OS	NS	EM
Density (#/L)	1998	no data	0.001	0.001
	1999	trace*	0.025	trace*
	2000	no data	0.011	0.004
	2001	0.000	0.009	0.000
	2002	0.000	0.000	0.000
	2003	0.000	0.000	0.000
	2004	0.005	0.003	0.000
	2005	0.023	0.009	0.006
	2006	0.017	0.011	0.004
	2007	0.013	0.020	0.006
	2008	0.030	0.013	0.003
Average Size (mm)	1998	no data	2.063	2.144
	1999	1.601	1.114	1.164
	2000	no data	1.204	1.080
	2001	-	1.034	-
	2002	-	-	-
	2003	-	-	-
	2004	2.745	2.470	-
	2005	2.232	2.272	2.276
	2006	2.559	2.603	2.119
	2007	2.585	2.610	2.513
	2008	2.508	2.497	2.244
Biomass (µg/L)	1998	no data	0.108	0.131
	1999	0.002	0.534	0.007
	2000	no data	0.256	0.067
	2001	0.000	0.107	0.000
	2002	0.000	0.000	0.000
	2003	0.000	0.000	0.000
	2004	0.567	0.359	0.000
	2005	2.665	1.076	0.704
	2006	2.816	1.930	0.410
	2007	2.223	3.485	0.926
	2008	4.810	2.019	0.342

trace\* indicates  $0 < \text{value} < 0.001$

<sup>a</sup>reports published prior to 2007 show seasonal mean values