VARIATION OF Cu AND Cd IN PLANKTON OF HAMILTON HARBOUR

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IN

PELAGIC PLANKTON OF HAMILTON HARBOUR

By

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Abstract

Temporal variation in the levels of Cu and Cd in zooplankton, phytoplankton and water taken from five pelagic stations in Hamilton Harbour during three separate sampling periods (June, August and October/November 1990) was examined. Significant seasonal variation occurred in the Cu and Cd levels measured in water and phytoplankton (Cu Water: $df=2$, $F=32.28$, $P \leq .0001$, Cu Phytoplankton: $df=2$, $F=48.94$, $P \leq .0001$ and Cd Water: $df=2$, $F=18.98$, $P \leq .0001$, Cd Phytoplankton: $df=2$, $F=58.81$, $P \leq .0001$). However, the Cu levels observed in zooplankton did not vary significantly with season ($df=2$, $F=1.79$, $P \leq .1919$). The maximal levels of Cd in zooplankton in November may be due to increased ingestion of material that is resuspended during turnover. Similarly, the peak levels of Cu and Cd recorded in November in phytoplankton may be due to a combination of processes: a) change in the size structure of the phytoplankton community to smaller individuals with higher metal sorption capacities and/or b) contamination of phytoplanktonic tissue samples by resuspended material.

Phytoplankton metal levels (Cu and Cd) are negatively correlated with chl _a_ concentrations ($n=45$, $r=-.8171$, $P \leq .0001$ and $n-41$, $r=-.5961$, $P \leq .0001$, respectively). These relationships are likely the result of a dilution effect.

Zooplankton Cd levels were positively correlated with water and phytoplankton Cd concentrations (n=41, r=.3211, P≤.0407 and r=.7667, P≤.0001) while Cu levels were not correlated with any of the variables tested. The difference between the correlatedness of Cd levels in zooplankton to water and phytoplankton Cd levels compared to the lack of this type of relationship with regard to Cu may be attributable to the biological function of each metal in zooplankton. Cu is required in small amounts for physiological processes and may be regulated whereas Cd has no known biological function.

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Introduction

Aquatic organisms may take up metal through food, water, or a combination of these two pathways (Huckabee and Blaylock 1972 cited in Enk and Mathis 1977; Prosi 1979; Simkiss and Taylor 1989). The principal route of uptake varies for organisms with differing lifestyles and physiological requirements. Obviously, for autotrophic organisms such as phytoplankton, the mode of transfer of metals into the cells must be through water. This transfer may occur passively, through adsorption of metals to cell walls, or actively, which involves the use of energy to actively transport metal into the cell.

For zooplankton, metal uptake from water appears to be the predominant route of transfer (Kay 1985; Prosi 1979). Although they may accumulate metals from food as well, this pathway is generally considered to be of lesser importance (Prosi 1979). Zooplankton may process copious amounts of water through filter-feeding practices or constant aeration of respiratory surfaces. This may account for the seeming lack of importance of food as a contributive factor to metal burdens in these organisms. Furthermore, there is little evidence to suggest that food chain enrichment of metals, in the classic sense (i.e. biomagnification: as with organic

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contaminants, where highest trophic levels coincide with highest concentrations of the pollutant), does occur. The behaviour of mercury is an exception to this generalization; linkages between Hg levels in organisms and their respective trophic positions in the food web have been observed.

The metal burden of biota hinges principally upon bioavailability. Physiological processes such as excretion will affect the total amount of metal found in an organism at any given time, as will inherent properties of the organism such as age, size, weight and sex (Boyden 1977; Prosi 1979; Uthe and Chou 1988). However, the potential for biological uptake is governed or limited by the availability of the metal(s) to the organism.

Factors that regulate the availability of metals are chemical, physical and/or biological in nature. Goudey (1983) studied phytoplankton in a contaminated body of water and summarized the factors which can affect the availability of metal in solution, binding to the cell surface and subsequent uptake (see Table 1). The chemical and physical factors cited by Goudey as influencing metal bioavailability may be equally applicable to heterotrophic pelagic zooplankton.

Although the aqueous medium has been stressed as the most important avenue for the passage of metals into heterotrophic pelagic organisms (see above), in dynamic

systems, such as aquatic ecosystems, the significance of transfer through food should not be discounted.

Prosi (1979) highlights an important point:

"in respect to metal enrichment of heterotrophic organisms, the proportion of metal concentrations in water to those in nutrients is of decisive importance. The breathing mechanism in all aquatic organisms permits the uptake of heavy metals from water in which the metal supply is constant for all species. For this reason, the uptake from food assumes a greater importance because its heavy metal concentrations are prone to greater variations."

Laboratory investigations which have attempted to identify the most important pathway for the uptake of metals (i.e. food or water) into aquatic organisms have typically involved only one or two types of food sources (Williams and Giesy 1978; Bertram and Hart 1979; Carney et al. 1986; Hatakeyama 1987; van Hattum et al. 1989); the variation in metal content inherent in a "natural" milieu of food sources would not be represented in such experiments. This variation may potentially be significant in systems where the food resources of heterotrophic organisms change along a temporal scale. Thus, the findings of van Hattum et al. (1989) and Williams and Giesy (1978), that water served as a more important pathway for the uptake of cadmium into isopods and mosquitofish, may not necessarily be applicable to all model or natural systems.

The uptake kinetics of metals from dietary or aqueous sources are different and it has been suggested that

ingestion of contaminated material may be important in the long term accumulation of metals. Benayoun et al. (1974) compared equilibrium Cd concentration factors based on stable element measurements with those obtained from radiotracer experiments and concluded that exchange between Cd in the water and that in euphausiid tissue (a marine zooplankter) was a relatively slow process. Similarly, Renfro et al. (1975) found that ^{65}Zn in organisms had not reached isotopic equilibrium with the isotope in the water even though net ^{65}Zn accumulation in shrimp and crabs had ceased by the end of their experiment.

Food quality and availability have been implicated as mediating factors of heavy metal toxicity. Winner et al. (1977) found that *Daphnia magna* maintained on vitamin-enriched algae were less sensitive to chronic Cu stress than those fed a commercial trout-granule diet. Field populations of the calanoid copepods, *Acartia tonsa*, showed increasing tolerance to Cu with increasing food rations up to a threshold point where the LC_{50} values then remained constant (Sosnowski et al. 1979). Similarly, Chandini (1989) showed that at high food levels (4.5 X 10^6 cells ml^{-1} *Chlorella*) the toxic effect of Cd to *Daphnia carinata* was greatly reduced.

To further confuse the issue of metal dynamics, each

of the major groupings of factors affecting metal availability in solution (Table 1) may change along a temporal scale. In temperate ecosystems, chemical, physical and biological conditions vary on daily, seasonal, and annual scales (Wetzel 1983). Moreover, it has been suggested that metals may interact antagonistically or synergistically, that is, the uptake of one metal may affect that of another.

Historically, two approaches have been employed to study temporal variation and trophic level distribution of trace elements. The common link between the two approaches is that they both have been divergent of each other. A review of the literature reveals very few studies in which the food web distribution of metals, with respect to a temporal scale, has been examined. Most investigations have been centred on either i) the temporal variation of one component of a food web with respect to metal burden, or ii) the distribution of metals across all components of a food web at one time. A summary of available literature is provided in Table 2.

Mackie et al. (1987) examined the seasonal variation of metal levels in zooplankton from a set of lakes differing in pH, Ca content, dissolved organic carbon content and distance from a known metal source. In an analysis of the variation of zooplankton metal levels within one acidified

lake they found that the accumulation of certain metals was correlated with each other and also that, for some metals, zooplankton community composition played an important role. Specifically, the percentage of the total biomass contributed by cyclopoid copepods was significantly negatively correlated to the level of Cd in the zooplankton. Furthermore, a significant but weak positive correlation was found between the levels of Cd and Zn. However, these researchers did not attempt to relate the metal content of zooplankton to that of phytoplankton.

I am aware of only one study which has dealt with the temporal variation of metal levels across different trophic levels. Winner et al. (1990) indirectly addressed the effects of chronic Cu stress by measuring the densities of members of a freshwater lentic community. However, Gächter and Geiger (1979) examined the temporal variation of Cu, Cd, Hg, Zn and Pb levels in water and across several trophic levels in a series of field enclosures. They found that the concentration of these metals in phytoplankton (>20 μm) and zooplankton (>300 μm) was variable with time but highest for phytoplankton, lower for zooplankton and lowest for fish fry. Furthermore, they noticed that peak concentrations in phytoplankton did not necessarily coincide with peak concentrations in zooplankton nor did the height of concentration peaks in zooplankton correspond to the

height of corresponding phytoplankton peaks. Gächter and Geiger concluded that their results confirmed observations made by Feldt and Melzer (1978) and Knauer and Martin (1972) which indicated that it is very unlikely that inorganic metals are bioaccumulated in the food chain.

It seems that, in the present literature, there is a paucity of information that deals directly with the partitioning of metals across different trophic levels and how biological uptake of metal elements may change with time in aquatic systems. Furthermore, there is a need to perform an investigation of this type in an actual field setting i.e. one that is not contained in an enclosure. Forcing or controlling processes evident in natural situations may be altered (hindered or enriched) in enclosures, for instance, consider how water circulation and sediment resuspension may be substantially different in an enclosure versus a whole lake situation. By investigating metal behaviour in a 'natural' system and removing the potential artefacts of enclosure experiments it is hoped that a greater understanding of the real situation will result.

Hence, the underlying purpose of the present investigation was to describe the temporal variation of copper and cadmium, during one ice-free season, in the water, phytoplankton and zooplankton of Hamilton Harbour and to compare the results with those of previous researchers.

Methods and Materials

Study Site Description

Hamilton Harbour is located at the western tip of Lake Ontario. The harbour has an east-west axis of 8 km and a north-south axis of 5 km. It is triangular in shape and has a surface area of 22 km². The Burlington Ship Canal (820 m x 88 m x 9.5 m) bisects the natural sandbar that separates the Harbour from Lake Ontario and permits exchange between the two bodies of water (Fig. 1). The pelagic zone (> 6 m depth) occupies 1785 ha or 83% of the harbour's total area whereas only 380 ha make up the littoral zone. With a maximum depth of 23 m in the central basin and a mean depth of 13 m the total volume of the harbour is 2.8×10^8 m³.

Two small sub-basins are located at either end of the harbour. Cootes Paradise has an open-water surface area of 250 ha and is connected to the harbour at its far west end via the Desjardin Canal. Windermere Basin, having a surface area of approximately 40 ha, is located in the southeast corner of Hamilton Harbour. Both sub-basins have a mean depth of 0.7 m.

The Harbour receives drainage from a watershed of 500 km² that is composed of industrial, residential and agricultural land (Barica et al. 1988; Harlow and Hodson

1988). The major runoff for the area occurs between February and April; the total annual runoff is approximately 1.1 to 2.1 x 10^8 m^3. From all sources, including natural runoff, sewage treatment plants and Lake Ontario, the Harbour receives a total of 2.6-3.8 x 10^6 m^3 of water per day. Natural runoff, composed of inflows from Cootes Paradise, Red Hill and Grindstone Creeks, and storm water runoff from the cities of Hamilton and Burlington, accounts for only 2-19% of the water entering the Harbour. Conversely, sewage treatment plants account for 7-16% whereas inflow from Lake Ontario accounts for 72-87%. Although the Harbour has a theoretical residence time (i.e. volume/inflow) of 500 days, exchange with Lake Ontario shortens the residence time to 73 and 107 days for winter and summer respectively.

Hamilton Harbour is one of the most polluted sites in the Great Lakes and has been designated as an area of concern by the Canada-U.S. International Joint Commission (Barica et al. 1988; Barica 1989). Point sources of metal pollutants for the Harbour include industry, wastewater treatment plants and storm sewers and tributaries. Most industries are now connected to wastewater treatment plants. However, two steel mills (STELCO and DOFASCO) presently still discharge some wastes directly into the Harbour

(Harlow and Hodson 1988). These wastes include treated process effluents plus cooling and storm water sewer water. The industries which line the southern shore of the Harbour collectively use 27 m^3 s^{-1} of water and return a similar amount of effluent (Barica 1989).

Several wastewater treatment plants, dealing with both industrial and municipal wastes, empty into Hamilton Harbour. The effluents from the Hamilton and Burlington plants enter the Harbour directly whereas the Dundas and Waterdown plants discharge indirectly into the Harbour via Cootes Paradise and Grindstone Creek respectively. Burlington and Waterdown are towns that are located on the north shore; Dundas is situated west of the Harbour.

During periods of high rainfall, the combined sewer system in Hamilton frequently overflows, releasing untreated sewage and stormwater to the Harbour. Overflows from above the escarpment in Hamilton drain into Redhill and Chedoke Creeks while those below flow directly into the Harbour via 20 major outfalls (MOE 1981; Holmes 1986). All runoff that enters the Harbour from Burlington drains through small creeks or is discharged directly through storm sewers. Cootes Paradise outflow (which includes Spencer Creek), Redhill Creek, and Grindstone Creek are the major tributaries to Hamilton Harbour. Each of these carries

stormwater inputs as well as natural runoff.

Five non-point sources of pollution to the Harbour
have been identified. These include shipping, spills,
atmospheric inputs, contaminated groundwater from landfill,
and the resuspension of sediments. Bilge water, ballast
water, accidental discharges of cargo or fuel tanks, and
wash water from the decks of bulk carriers are types of
shipping wastes that may act as a sources of metals to the
Harbour. The sediments of the Harbour are extensively
contaminated (Fig. 2) and may be subject to resuspension or
bioturbation (McIntosh et al. 1978; Reynoldson 1987; Robbins
1982; Wood 1975) processes.

The Harbour exhibits unique limnological phenomena.
It displays a high degree of physical variability which
results in oscillations, mixing, and an unstable thermal
structure. Although a thermocline does develop at depths of
7-8 m (Harris 1975), Barica (1989) has shown that exchange
of the Harbour water with that of Lake Ontario through the
Burlington Ship canal has significant effects upon the water
quality in the Harbour. Barica (1989) has calculated that
without the beneficial exchange of water between the Harbour
and Lake Ontario the concentrations of pollutants per unit
volume and the algal biomass would be about 50% higher.
Entry of the 'cleaner' Lake Ontario water effectively
dilutes the concentrations of pollutants in the Harbour and

oxygenates its hypolimnion, thereby reducing the release of sediment-bound contaminants under anoxic conditions.

The phytoplankton population of Hamilton Harbour exhibits a strong seasonal succession (Harris and Piccinin 1980). The domination by diatoms and small phytoflagellates in the spring is followed by a short period where coccoid green algae are most abundant. A mixture of diatoms and green algae predominate once the maximum summer temperature is reached; diatoms and flagellates return later, during periods of homogeneous mixing, to form the principal phytoplankton populations.

Few studies have been done on the zooplankton of the Harbour; however, it has been shown that large rotifer populations dominate the community for most of the year. Analogous to the phytoplankton, the zooplankton of the Harbour also undergo seasonal succession. The spring peak is made up almost entirely of *Keratella quadrata* whereas, during summer stratification, epilimnetic species are dominated by a cladoceran, *Bosmina longirostris* and a rotifer, *Branchionus angularis* (Harris 1976).

Experimental Design

Due to the labour intensive nature of this project, sampling of the components of the pelagic food chain of Hamilton Harbour was punctuated during three time periods. Only the ice-free seasons were sampled. Sampling took place

during the months of June, August and October/November (1990).

Five stations, approximately equi-distant from each other and located on a transect that bisected the pelagic zone of the Harbour longitudinally (Fig. 1), were characterized during each sampling effort. Although others (RAP 1989) have defined the pelagic zone as the area having a water depth >6 m, for the purposes of this project, the pelagic zone was arbitrarily defined as the area of open water bounded by the 10 m contour. This was done to reduce the risk of contaminating samples by resuspended sediments.

Three replicate samples of water, phytoplankton, and zooplankton were obtained from each station during each season.

Field Measurements, Sampling Techniques and Protocols

During each sampling effort a series of physical and chemical measurements were taken. Upon arrival at the sampling station water depth was determined using a weight attached to a rope demarcated at 1 m intervals. Light penetration was estimated using a standard black and white Secchi disk. Temperature and dissolved oxygen concentration were measured at 1 m intervals from the water surface to the benthic sediment with a YSI Model 51B-5739 thermistor-O_2 probe.

Using a Water Puppy Model 12560 self-priming bilge pump attached to a length of polypropylene/silicon tubing (5/8 inch i.d. x 7/8 inch o.d), powered by an automotive battery, water samples were obtained from the middle of the euphotic zone (1X the Secchi depth). For trace element analyses 500 ml of water were collected and immediately acidified to pH < 2 with 3 ml of 8 N nitric acid. Finally, 2 L of water were collected for chl $\underline{a}$ analyses and the determination of metal concentrations in the algal biomass.

Depending upon the thermal structure of the water column, zooplankton samples were taken from the mixed water layer or the epilimnion. Specifically, the whole water column was sampled when no thermocline was apparent but during periods of stratification only the epilimnion was characterized. To avoid possible bias through contamination of the samples from benthic sediments and/or organisms that dwell in the sediments, the water column was arbitrarily defined as extending from the surface (depth=0 m) to 2 m above the benthic substrate.

An integrated sample of zooplankton was obtained with the apparatus pictured in Fig. 3 and the following method. Water was pumped on board and subsequently filtered through two nets. The nets should have retained organisms that had dimensions > 200 μm and those whose dimensions occupied the range < 200 μm and > 63 μm. The latter

fraction will henceforth be simply referred to as the > 63 μm fraction. The tubing, demarcated at 1 m intervals, was pulled up through the water column at a constant rate (usually 1 m/min). Damage to the organisms was minimized by ensuring that the vessel which supported the nets was full of water before the actual sampling began. At the end of each sampling run, the organisms which had collected on the nets were washed into the removable sample bottles located at the apex of each net. The sample bottles were then topped up with filtered lake water and capped.

A variety of precautions were taken to reduce the risk of contaminating samples with metal and to minimize cross-contamination between samples from different stations. All sampling was done from a fibreglass boat with an outboard engine attached. As each sampling station was approached, the engine was turned off and the boat was allowed to drift away from the water which had been expelled from the engine. The anchors were then thrown overboard and sampling did not commence until the boat remained in a relatively stationary position.

All sample bottles and containers were soaked in 8 N nitric acid for 24 hrs. and rinsed three times with both distilled and distilled-deionized (Nanopure) water. All acid-washed containers were subsequently rinsed with their respective samples three times before filling. Trace metal

analytical quality concentrated nitric acid and distilled-deionized water was used to make up the soaking solution and also the 8 N solution which was used to acidify and preserve the water samples for trace metals determination. The sample containers and bottles were of polyethylene construction (Nalgene).

Prior to each sampling season, the tubing that was used in conjunction with the pump was rinsed with 8 N nitric acid and then rinsed three times with both distilled and distilled-deionized water. Before use at each station, the pump and its tubing was cleared of any air-locked water from the previous station and rinsed with a quantity of sample. Similarly, the nets were also rinsed between stations.

Although the pump had a metal housing, its impeller and inner parts were made of Neoprene-like material. The net frames were constructed of polyethylene tubing and sealed with silicone. The lead weights that were attached to both the depth gauge and the pump's tubing were covered in plastic and sealed with cloth-backed adhesive tape.

To prevent deterioration, all samples were stored (in the field) and transported to the laboratory in a darkened ice-chilled insulated container. All samples that required initial laboratory processing underwent it on the same day of their collection.

Sample Processing

Upon arrival at the laboratory all water samples for trace metal analyses were filtered through a glass fibre filter (Whatman GF/C Glass Microfibre Filter). The filtrate was retained and stored at 4°C until the appropriate analyses could be performed. To prevent cross-contamination, the filter funnel apparatus was rinsed once with 8 N nitric acid and three times with distilled and distilled-deionized water between samples.

For chl a analyses, two replicate aliquots of the 2 L water sample were deposited onto glass fibre filter papers. Similarly, for determination of the trace metal content of phytoplankton tissue, an appropriately-sized aliquot was filtered through a pre-weighed glass fibre filter. Depending upon the season and the apparent standing crop, aliquots for chl a and trace metal determination of the phytoplankton tissue ranged in size from 150-300 ml and 200-1200 ml respectively. The phytoplankton tissue and its associated filter paper was dried at 90°C for 24 hrs., placed in a desiccator for 24 hrs., and then weighed. All filter papers and their associated tissue samples for chl a and trace metal determinations were stored in the dark at -20°C until the appropriate analyses could be performed.

Preliminary analyses showed that the > 63 μm zooplankton size fraction was extensively contaminated with

phytoplanktonic cells. Thus, upon returning to the laboratory, the > 63 μm zooplankton sample was placed in a 500 ml separatory funnel and bubbled with nitrogen gas for approximately 30 min. This caused the algal cells to float to the surface while the zooplankton remained near the bottom of the funnel. The purified zooplankton sample was then removed and treated in the same manner as were the > 200 μm zooplankton samples.

Filtered lake water was added to each zooplankton sample and the volume made up to 200 ml. According to the method outlined by McCauley (1984), three subsamples of each zooplankton sample were taken. All subsample volumes for the > 63 μm size fraction were 2.5 ml whereas those of the > 200 μm size fraction ranged from 2.5 to 5 ml. A volume of concentrated sugared formalin solution equal to that of the subsample volume was added to each zooplankton subsample to give a final concentration of 4% formalin and 60 g l^{-1} sucrose. After the subsamples had been taken, the zooplankton remaining from the 200 ml of lake water were deposited onto a pre-weighed glass fibre filter and dried, weighed, and stored in the same manner as were the phytoplankton tissue samples.

All glass fibre filters that were used to hold tissue samples were first dried for 24 hrs. at 90°C, stored

in a desiccator for 24 hrs. and then promptly weighed. All
weight measurements were performed on a Sartorius-Werke GMBH
analytical balance ($\pm$ .1 mg).

<u>Laboratory Analyses</u>

<u>Chlorophyll a</u>

The SCOR/UNESCO procedure (Strickland and Parsons
1972) was used to determine chl <u>a</u>. Absorbance of the
extracts was read against a 90% acetone blank on a Zeiss
PMQ II spectrophotometer. Rectangular cuvettes with a light
path of 5 cm were used. All cuvettes were rinsed with
reagent grade acetone between samples.

<u>Zooplankton Enumeration</u>

A Sedgewick-Rafter cell counter with a capacity of
1.4 ml was used to hold samples; all zooplankton were
counted at 100X magnification under an Olympus BH-2 compound
microscope. Subsamples from the > 200 μm zooplankton size
fraction were counted in their entirety whereas those from
the > 63 μm fraction were diluted and a smaller subsample
counted. Entire subsamples from the > 63 μm size fraction
were periodically enumerated to ensure that dilution and
further subsampling did not distort the counts.

The taxonomic references used during sample counting
included those of Donner (1956), Chengalath et al. (1971),
Ruttner-Kolisko (1974), Pennak (1978), Smith and Fernando
(1978), Fitzpatrick (1983) and Balcer (1984). The

zooplankton were identified to species or genus and the values of their abundance and/or biomass were later clumped into major taxonomic groups.

To obtain an estimate of the amount of biomass contributed by the various types of zooplankton, at least ten separate subsamples were chosen randomly from each zooplankton fraction (the > 200 μm and the > 63 μm zooplankton) and the length and width of at least 50 of each type of cladoceran and 10 of each type of rotifer were measured. Paired t-tests comparing the mean widths and lengths of 13 types of organisms common to both size fractions revealed that there were no significant differences in width or length (T=1.44, P=.1747 and T=1.35, P=.2027, respectively) between the organisms retained separately on the two nets. Thus, the abundance data for the two zooplankton size fractions were pooled and the average length of each type (i.e. genus or species, not the larger major grouping) of animal was calculated (see Appendix - Table 1). These numbers were substituted into appropriate length-weight regression equations (see Table 2 in the Appendix) to yield wet weights. For animals whose occurrence was relatively rare, body lengths were estimated from literature values and from at least 10 actual measurements. Wet weight values, per individual, of each zooplankton taxon were multiplied by their respective

abundances (Tables 3 and 4 in the Appendix) to obtain a measure of the biomass contributed by each taxon at each sampling time.

Trace Metal Analyses

In order to measure Cu and Cd content, tissue samples and their associated glass fibre filter papers were placed in polypropylene conical tubes and 3 ml of 1.0 N nitric acid added. The samples were digested in an 80°C water bath for 8 hrs. and then centrifuged at 7000 rpm for 15 min. The supernatant was removed with acid-washed glass pipettes and stored in acid-washed 6 ml polyethylene scintillation vials.

After appropriate dilution with distilled-deionized water, Cu and Cd concentrations were determined with a Varian GTA-95 Series graphite furnace atomic absorption spectrophotometer. Nitrogen gas was used in conjunction with the graphite tube atomizer.

Water samples were atomized directly - they were not diluted prior to the spectrophotometric analyses. The detection limits for Cu and Cd were 1 and .2 ug l^{-1}, respectively.

Statistical Analyses

All statistical analyses were performed on an IBM-compatible personal computing system. Bar graphs and line

plots were constructed using the computer program, Sigma-Plot Version 3.10 (Jandel Scientific, 1987). QUATTRO PRO Version 1.0 (Borland International Inc., 1989) was used to store data and perform simple mathematical calculations. SAS (Release 6.03, SAS Institute, 1987) was used to perform analysis of variance on the data. The repeated measures option was used with the general linear model ANOVA to determine main effects. It was assumed that the three samples taken from each station during each season were replicates of each other. The HELMERT option (SAS Institute Inc. 1985, pg.482), used in conjunction with the repeated measures option, identified temporal trends in the data.

Although the zooplankton tissues from each size fraction were analyzed separately for both Cu and Cd, zooplankton metal levels are expressed as means of the two fractions. This was done because, as stated earlier, the zooplankton did not separate into the two fractions clearly on the basis of their size.

Results

Although all stations were not necessarily sampled on the same day, in order to facilitate the graphical depiction of temporal trends, the first, second and third replicate samples from each station during each sampling season will be referred to as A, B or C, respectively. For example, August (B) refers to the second replicate sample taken in the month of August for the station(s) that is(are) the current object(s) of discussion. Furthermore, the October and November sampling dates have been clumped together under the single heading of November. A complete listing of specific sampling dates for each station may be found in the Appendix (Table 5).

Chl a and Secchi depth

Stations 1-3 exhibit a similar pattern of seasonal variation in chl $\underline{a}$ levels; June values increase to a maximum in August and plummet to minimum levels in November (Fig. 4.A). While the pattern of variation of chl $\underline{a}$ at Station 4 is unique, that of Station 5 resembles the pattern exhibited by Stations 1-3 from August (A) to the end of the sample collection period (November (C)). However, the apparent differences between the stations are not significant (df=4, F=.23, P$\leq$.9178) whereas season is a significant main effect

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(df=2, F=38.03, P$\leq$.0001). The HELMERT option in the ANOVA identified the November chl a values as being significantly lower than those of August (df=1, F=83.29, P$\leq$.0001).

The magnitude of the changes in chl a is greatest at Station 5; extremes attained by chl a at Station 5 are paralleled by concomitant peaks (in the opposite direction) in the Secchi depth (Fig. 4.B). Although Stations 1-4 show a fairly similar pattern of Secchi depth response to variations in chl a (i.e. Secchi depth generally decreases as chl a levels increase), contrary to Stations 1-3 and also 5, the magnitude of the variation in chl a concentrations is not reflected by proportional changes in the Secchi depth at Station 4. From the chl a pattern of variation at Station 4, one would expect to observe greater changes in the Secchi depth than those that were actually measured. Nonetheless, there are no significant differences between the secchi depths measured at the various stations (df=4, F=.13, P$\leq$.9664).

Again, season figured as a significant factor affecting the Secchi depth values (df=2, F=6.65, P$\leq$.0061) with June values significantly lower than the mean values of August and November (df=1, F=26.56, P$\leq$.0004).

Temperature and Oxygen

The temperature profiles of all stations at the time of first sampling depict an approximately linear decrease in

temperature with depth (Fig. 5.A-E). Except for Station 5
which had attained its maximum thermal stratification at the
first August sampling (Fig. 5.E), the degree of
stratification at all stations increased from June (B) to a
maximum at August (B) (Fig. 5.A-D). In contrast to Stations
4 and 5 (Fig. 5.D-E), which showed no stratification at the
final August sampling, Stations 1-3 (Fig. 5.A-C) retained a
high degree of thermal stratification until this time. The
temperature profiles of all stations during November were
virtually isothermal.

The unique oxygen profiles of Station 4 and 5 on
June (A) (Fig. 6.D-E) roughly parallel their respective
temperature profiles (Fig. 5.D-E). However, all June oxygen
profiles from Stations 1-3 (Fig. 6.A-C) and the June (C)
profiles of Stations 4 and 5 follow a negative heterograde
curve, with metalimnetic oxygen minima generally occurring
at 11-13 m depths. Several of the heterograde curves show
curious inversions. Those which are most pronounced occur
at Station 1 (Fig. 6.A) on June (B), Station 2 (Fig. 6.B) on
June (A) and (C), and Station 3 (Fig. 6.C) on June (A); more
subtle inversions are apparent at Stations 1 and 3 on
June (C).

During August, dissolved oxygen levels in the water
column at all stations approximate a clinograde curve, with
nearly anoxic, minimum levels occurring at about 12 m. In

contrast, the November profiles show that the entire water column at all stations is well oxygenated and almost at saturation levels.

Zooplankton

For all major types of zooplankton that were sampled, sampling station never figured as a significant factor affecting the biomass values (df=4, Cladocerans: F=1.80, P≤.2052; Cyclopoids: F=1.59, P≤.2521; Rotifers: F=1.23, P≤.3574; Harpacticoids: F=.63, P≤.6492).

The cladocera, composed of *Bosmina* and *Daphnia sp.*, formed a dominant component of the zooplankton community at all stations at all times (Fig. 7.A-E). The cladocera show strong seasonal differences in their standing crop levels (df=2, F=143.36, P≤.0001); the biomass values recorded in November are 10 times lower than the peak levels measured in June. June biomass levels are significantly higher than the mean of both August and November (df=1, F=159.28, P≤.0001) while those of August are higher than those of November (df=1, F=32.16, P≤.0002).

Consisting exclusively of *Cyclops sp.* and their associated nauplii, the cyclopoid group exhibits peak maximum standing crop levels in August; June values are lower than the mean of August and November (df=1, F=27.16, P≤.0004). Seasonal variation is significant at P≤.0006 (df=2, F=10.96).

The rotifers, like the cladocerans, form a dominant part of the zooplankton community in June, but additionally also in November, with minimum levels occurring in August. June biomass levels are significantly higher than the mean August and November values (df=1, F=68.74, P$\leq$.0001); August values are less than those of November (df=1, F=18.43, P$\leq$.0016). Again, seasonal variation is significant (df=2, F=68.25, P$\leq$.0001).

The harpacticoids, including only *Leptodora kindti*, contributed to the zooplankton biomass only in June, with an obviously significant seasonal effect (df=2, F=8.22, P$\leq$.0071).

Metals

Copper

For all fractions tested, sampling station was not a significant main effect in relation to copper concentrations (df=4, Water: F=.5, P$\leq$.7376; Phytoplankton: F=.57, P$\leq$.6882; Zooplankton: F=3.36, P$\leq$.0545).

Water and phytoplankton Cu concentrations vary significantly with season (df=2, F=32.28, P$\leq$.0001; and df=2, F=48.94, P$\leq$.0001, respectively). Cu concentrations in water are higher in June (Fig. 8.A) than the mean values of August and November (df=1, F=108.33, P$\leq$.0001) whereas phytoplankton Cu concentrations are higher in November (Fig. 8.B) than in August (df=1, F=77.63, P$\leq$.0001) and June.

Contrary to the behaviour of Cu in water and phytoplankton, zooplankton Cu levels do not vary significantly with season (df=2, F=1.79, P≤.1919). Although sampling station is not a significant main effect, visual inspection shows that the levels of Cu in the zooplankton from Station 1 (Fig. 8.C) follow a different pattern of temporal variation than those of Stations 1-4. An analysis of variance with Station 1 data removed also did not yield a significant seasonal effect (df=2, F=.69, P≤.5157).

Phytoplankton Cu concentrations were negatively correlated with both water Cu concentrations and chl $\underline{a}$ concentrations (n=45, r=-.3563, P≤.0163 and r=-.8171, P≤.0001, respectively). Zooplankton Cu concentrations were not significantly correlated with any of the variables analyzed (Appendix - Table 6).

Linear regression revealed that water Cu levels explain less of the variation in phytoplankton Cu concentrations than do chl $\underline{a}$ levels (R=.1270 versus R=.6677). The resulting equations are:

$$\log (\text{Phyto. Cu}) = -13.424 - .340 \log (\text{Water Cu})$$

F=6.254   P≤.0163

$$\log (\text{Phyto. Cu}) = -5.241 - .887 \log (\text{chl } \underline{a})$$

F=86.411   P≤.0001

Where Phyto. Cu is measured in M kg^{-1},
Water Cu " " M l^{-1},
and chl $\underline{a}$ " " ug l^{-1}.

Cadmium

Again, for all fractions tested, sampling station was not a significant main effect in relation to Cd concentrations (df=4, Water: F=.77, P$\leq$.5682; Phytoplankton: F=.99, P$\leq$.4551; Zooplankton: F=.30, P$\leq$.8734).

Water and zooplankton Cd concentrations vary significantly with time of sampling (df=2, F=18.98, P$\leq$.0001 and F=21.14, P$\leq$.0001, respectively). Water and zooplankton concentrations increase to maximum levels in November (Fig. 9.A and C); levels of Cd in water and zooplankton are lower in June than mean levels of August and November (df=1, F=20.51, P$\leq$.0011 and F=7.97, P$\leq$.0181, respectively) and August values are significantly lower than those of November (df=1, Water: F=9.00, P$\leq$.0133 and Zooplankton: F=34.41, P$\leq$.0002).

Fig. 9.B seems to indicate that phytoplankton Cd concentrations follow the same temporal pattern as do water and zooplankton Cd concentrations, however, the analysis of variance did not reveal season as a significant main effect (df=2, F=1.93, P$\leq$.1845). Although station was not identified as a significant main effect, the phytoplankton

Cd levels on Jun (A) and Nov (A) from Stations 2 and 3 seem
anomalously high (Fig. 9.B). When these data are excluded
from the ANOVA, sampling season surfaces as a significant
factor affecting the phytoplankton Cd concentrations (df=2,
F=58.81, P$\leq$.0001).

A correlation analysis that also excluded the
anomalous values (n=41) revealed that zooplankton Cd levels
are positively correlated with water Cd concentrations
(r=.3211, P$\leq$.0407), phytoplankton Cd concentrations
(r=.7667, P$\leq$.0001), and cyclopoid biomass (r=.5478, P$\leq$.0002)
and negatively correlated with cladoceran biomass (r=-.6873,
P$\leq$.0001). Phytoplankton Cd levels, similar to the behaviour
of copper, are negatively correlated with chl a levels (r=-
.5961, P$\leq$.0001) (Appendix - Table 7).

Linear regressions showed that variation in
phytoplankton Cd levels explain more variation in
zooplankton Cd levels than do any of the other factors
tested:

log (Zoop. Cd) = -5.767 + .480 log (Phyto. Cd)

R=.5879   F=55.646   P$\leq$.0001

log (Zoop. Cd) = .707 -3.14 log (Cladoceran biomass)

R=.4724   F=34.924   P$\leq$.0001

log (Zoop. Cd) = -13.706 + .784 log (Cyclopoid biomass)

R=.3000   F=16.717   P$\leq$.0002

$$\log (\text{Zoop. Cd}) = 17.222 + 1.461 \log (\text{Water Cd})$$

$$R=.1031 \qquad F=4.483 \qquad P\leq.0407$$

Where Zoop. Cd is measured in M kg^{-1},
Phyto. Cd " ",
Water Cd " " M l^{-1},
and Zooplankton biomass " " mg m^{-3} wet weight.

Cd levels in phytoplankton may be predicted with the following equation:

$$\log (\text{Phyto. Cd}) = -5.231 - 2.809 \log (\text{chl } \underline{a})$$

$$R=.3553 \qquad F=21.491 \qquad P=\leq.0001$$

Where Phyto. Cd is measured in M kg^{-1},
and chl $\underline{a}$ " " ug l^{-1}.

Discussion

As stated in the Methods section (Study Site Description), Hamilton Harbour exhibits some unique limnological phenomena. The inflow of water from Lake Ontario through the Burlington Ship Canal and also from Cootes Paradise may account for some of the apparent variation in measured parameters between stations (Figs. 4,5 and 6). Recall that Station 5, especially, and also Station 4 have patterns of variation in chl $\underline{a}$ and Secchi depth (Fig. 4.A and B) that are slightly, but not significantly, different from those of Stations 1-3. Furthermore, while Stations 1-3 retained a high degree of thermal stratification until the final August sampling date, the stratification that had been apparent at Stations 4 and 5 had already disappeared by this time. Inflows from Cootes Paradise and storm sewers have been identified as some of the major nutrient loaders to the Harbour and their points of entry are located in the vicinity of Stations 4 and 5 (Fig. 1), hence, they may possibly account for the deviations in chl $\underline{a}$ and Secchi depth variation. Nonetheless, Stations 1-3 may also be subject to nutrient loading from one of the steel processing mills, as identified by Klapwijk and Snodgrass (1985). However,

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Stations 4 and 5 are also located in areas where the water depths are much lower (17 and 13 m maximum water depths, respectively) than those at Stations 1-3 (having maximum water depths from 21 to 23 m). The potential for the whole water column to mix becomes greater as water depth decreases. Hence, the shallower water depths at Stations 4 and 5 relative to those at Stations 1-3 may be a factor contributing to the earlier breakdown of thermal stratification at Stations 4 and 5.

Stations 1-3 seem to be affected by similar forces, that is, all exhibit inversions in their oxygen profiles; Station 1 on June (B) and (C), Station 2 on June (A) and (C) and Station 3 on June (A) and (C) (Fig. 6.A,B and C, respectively). It seems likely that these inversions were caused by inflows of water from Lake Ontario through the Burlington Ship Canal. Barica (1989) and Barica et al. (1988), through measurements of ammonia distribution in the Harbour and from Lake Ontario, claim that the major zone of exchange of water between Lake Ontario and Hamilton Harbour occurs in the eastern third of the Harbour. This area corresponds closely to the location of Stations 1 and 2 and borders on the region that was sampled as Station 3.

Hamilton Harbour is thought to be a detritus-based system (RAP 1989). This distinction means that energy flows from primary to secondary producers primarily via a detrital

food chain. Sprules (1980) claims that, in this type of a system, secondary production is predominantly by small species or 'microfiltrators' such as *Bosmina*, *Ceriodaphnia*, *Chydorus*, small rotifers such as *Keratella* and *Filinia*, copepod nauplii and certain protozoans. He explains that in such lakes, which are typically very productive, most phytoplankton are about 10 μm or larger and are consequently unavailable to microfiltrators which are restricted to particles from one to several micrometers.

Although Sprules (1980) groups *Bosmina* with the 'microfiltrators' and suggests that they feed on particles that have dimensions from one to several micrometers, Müller (1985) determined that the size spectrum that can be filtered efficiently is 1-20 μm for adult *Bosmina coregoni*, 0.6-14 μm for their juveniles and 0.4-14 μm for *Bosmina longirostris*. Müller also suggested that *B. coregoni* could be expected to show a slight preference for small algae while bacteria might be the preferred food of *B. longirostris*. *B. coregoni* and *B. longirostris* were the two species of bosmonids that were found in Hamilton Harbour.

In contrast to the microfiltrators, *Daphnia*, one of the types of cladocerans found in Hamilton Harbour, are considered to be 'macrofiltrators' and specialize on particles that are approximately 10-20 μm (Sprules 1980). Some daphnid species may feed on a larger size range of

particles (Brooks and Dodson 1965; Gliwicz 1965; Hillbricht-Ilkowska 1977; Porter 1977); i.e. the largest particle diameter found in the intestinum of *Daphnia cucullata* by Gliwicz (1980) was 26 μm.

Table 3 contains a list of phytoplankton taxa that were recorded over the course of an ice-free season in Hamilton Harbour; where possible, dimensions of these taxa have been provided. The constituents of the phytoplankton population change little from year to year (Harris et al. 1979; Harris and Piccinin 1980); it is not unreasonable to expect that the phytoplankton population of 1990 may be fairly similar to that of 1975. It is apparent that the majority of taxa listed in Table 3 exceed the dimensions deemed suitable for microfiltrators; hence, the zooplankton in Hamilton Harbour are probably restricted to detrital feeding. The daphnids and the bosmonids (the cladocerans) may be able to feed on some of the phytoplankton genera listed, based on their dimensions and the respective feeding limitations of the zooplankton.

The increase in Cd concentrations in the zooplankton in November (Fig. 8.C and 9.C) may be attributable to the water circulation processes occurring at that time. Figures 5 and 6 clearly indicate that the water column is well mixed during the November sampling season; the isothermal conditions indicate that turnover is occurring. Turnover

events are known to resuspend sediments (Wetzel 1983).
Thus, it is plausible that zooplankton that feed primarily
on detritus may ingest more particles of benthic origin
during turnover events than when the water column is
stratified. The resuspension of sediment particles, which
would have higher concentrations of metals than the
overlying water, and subsequent ingestion by filter-feeding
zooplankton, may account for the higher levels of Cd seen in
the zooplankton. Cu levels in the zooplankton were not
significantly different between the three sampling seasons
(df=2, f=.69, $P \leq .5157$).

The maximum levels of Cu and Cd that are found in
the phytoplankton in November may also be partially due to
resuspension processes. Total dissolved Cd levels reach a
maximum in November. Phytoplankton uptake of metal is
dependent upon water concentrations (Table 1). Increased
concentrations of Cd in the water in November coincide with
increased levels of Cd in phytoplankton; resuspension of
benthic sediments may increase dissolved concentrations,
which would in turn potentially increase concentrations in
phytoplankton. However, this mechanism would not explain
the high concentrations of Cu observed in the phytoplankton
in November when the total dissolved Cu concentrations are
at their minimum. In fact, the water Cu concentrations are
negatively correlated with phytoplankton Cu concentrations

(n=45, r=-.3563, P$\leq$.0163) whereas water Cd levels show no significant correlation to phytoplanktonic Cd levels.

Alternatively, it is more probable that the high levels of Cu and Cd in phytoplankton result from contamination of the tissue samples by resuspended sediments. Due to the nature of the method used, it was impossible to obtain phytoplankton tissue samples that consisted entirely of only phytoplankton. It is inevitable that the filter (2 μm pore size) would have retained detrital or neustonic particles along with phytoplanktonic cells which would effectively contaminate the tissue sample. As mentioned earlier, during turnover (in November) the concentration of resuspended particles in the water column is expected to increase.

Another possible contributing factor to the increased levels of phytoplankton Cu and Cd in November involves the taxonomic composition of the community. During periods of homogeneous mixing diatoms and small phytoflagellates form the principal phytoplankton populations (Harris and Piccinin 1980). During the summer, after the water column has stratified (corresponding to the August sampling), a mixture of diatoms and larger green algae dominate the community. In late spring-early summer, following spring turnover, coccoid green algae are the predominant form of phytoplankton (corresponding to the June

sampling period). It is thought that as particle size decreases and subsequently, surface to volume ratios increase, the metal sorption capacity of particles increases (Boothe and Knauer 1972; Flegal and Martin 1977; Ramamoorthy and Rust 1978; Rendell et al. 1980; Reid and McDuffie 1981; Ajmal et al. 1982; Ward 1983; Wiley and Nelson 1984; Comans 1987). The smaller phytoplankters present during turnover, in conjunction with resuspend sediments contaminating the tissue samples, may explain the high levels of metals apparent in the phytoplankton in the late fall sampling season. However, as the phytoplanktonic samples were not evaluated taxonomically and the percentage of the "phytoplankton" samples composed of actual living phytoplankton is not known, it is impossible to attach significance to these speculations.

Mayer and Manning (1990) evaluated the metal content of suspended solids and water in Hamilton Harbo^] during April and September of 1986. Their stations HH-1, HH-2, HH-3, and HH-5 correspond to the area of the harbour sampled in the present study. I calculated mean values for their stations. It should be noted that Mayer and Manning sampled in September, while the water column was still stratified. Nonetheless, the results of the present study seem reasonable in comparison to those of Mayer and Manning in 1986. The peak values observed in the phytoplankton in

November occupy similar ranges to those for suspended solids in September 1986: Copper 7-24 X 10^{-4} M kg^{-1} in phytoplankton versus a mean 13 X 10^{-4} M kg^{-1} in suspended solids and Cadmium 30-160 x 10^{-6} M kg^{-1} (excluding anomalous values of Station 2 and 3) in phytoplankton versus a mean 43 X 10^{-6} M kg^{-1} in suspended solids. Water Cd concentrations are not available from the Mayer and Manning but a mean Cu level in water of 14 X 10^{-8} M l^{-1} calculated from their data is higher than the range of values seen in the present study, 0-3.3 Cu M l^{-1} (November). The discrepancy between the levels of Cu in water may perhaps be due to recent efforts to reduce the loading of metals, from industrial and municipal processes, to the harbour.

In retrospect, this type of a study could potentially yield more information about metal variation in zooplankton if their food sources were identified through feeding studies or gut analyses. Furthermore, if phytoplankton could be truly isolated from other confounding materials a greater understanding of metal transfer may result. At the present time no method exists that will separate detrital material (of recent biological origin or from resuspension events), bacteria, and phytoplankton, that will also simultaneously avoid the methodological problems caused by high pressure filtration or high speed

centrifugation (i.e. destruction of intact cells).

The correlation analyses showed that both Cu and Cd levels in phytoplankton may be explained by chl $\underline{a}$ concentrations. Because chl $\underline{a}$ is an estimator of phytoplanktonic standing crop levels (Wetzel 1983) and because the relationship between metal levels and chl $\underline{a}$ is negative, that is, metal levels in the phytoplankton increase as chl $\underline{a}$ levels decrease, it appears as if a dilution effect is occurring. Thus, when chl $\underline{a}$ levels are high, the levels of Cd and Cu will be low in the phytoplankton because the load of metal is 'spread out' over a larger biomass.

The behaviour of Cu and Cd differ in that Cd levels in zooplankton were positively correlated with both Cd levels in water and in phytoplankton whereas Cu levels in zooplankton showed no significant correlations to any of the variables tested. Cadmium has no known biological function while copper may be required in small amounts for various physiological processes (Simkiss and Taylor 1989). The positive correlation between Cd levels in zooplankton and water and phytoplankton and the conspicuous lack thereof with regard to Cu suggests that Cu may be regulated biologically. Regulation of metal levels through either excretion or selective or active uptake would obscure correlatedness between levels in zooplankton and those in

water or phytoplankton. Also, recall that Cu levels in zooplankton did not vary significantly between sampling seasons.

The regression and correlation analyses also revealed that as the percentage of the zooplankton community contributed by cyclopoids increases, the Cd content of zooplankton simultaneously increases ($n=41$, $r=.5478$, $P \leq .0002$). Conversely, as cladoceran biomass contribution to the total zooplankton biomass decreases, the level of Cd in zooplankton decreases ($r=-.5961$, $P \leq .0001$). These results contrast those of Mackie et al. (1987) who performed Spearman Rank correlation analyses on the concentrations of Zn and Cd in zooplankton > 250 μm from a series of lakes and found a significant negative correlation between cyclopoid biomass and Cd concentrations in zooplankton ($P < .01$ and $r=-0.716$) but no significant correlation between Cd levels in zooplankton and percentage contribution to total biomass by cladocerans.

Conclusions

Significant seasonal variation occurred in the Cu and Cd levels measured in water and phytoplankton. However, the Cu levels observed in zooplankton did not vary significantly with season. The maximal levels of Cd in zooplankton in November may be due to increased ingestion of material that is resuspended during turnover. Similarly, the peak levels of Cu and Cd recorded in November in phytoplankton may be due to a combination of processes: a) change in the size structure of the phytoplankton community to smaller individuals with higher metal sorption capacities and/or b) contamination of phytoplanktonic tissue samples by resuspended material.

Phytoplankton metal levels (Cu and Cd) are negatively correlated with chl a concentrations. These relationships are likely the result of a dilution effect.

Zooplankton Cd levels were positively correlated with water and phytoplankton Cd concentrations while Cu levels were not correlated with any of the variables tested. The difference between the correlatedness of Cd levels in zooplankton to water and phytoplankton Cd levels compared to the lack of this type of relationship with regard to Cu may be attributable to the biological function of each metal in

zooplankton. Cu is required in small amounts for physiological processes and may be regulated whereas Cd has no known biological function.

TABLES

Table 1: Factors regulating the availability of heavy metals in solution. Reproduced from Goudey (1983).

Biological, chemical, and physical factors that can directly, or indirectly, regulate the availability of metals in solution, binding to the cell surface, and uptake are presented.

CHEMICAL: Speciation- free ion   inorganic   particles (clay) colloids
hydrated ion - organic -flocculates
complexed ion ligands precipitates humic/fulvic acids

Properties- complexation, exchange, and redox reactions
- solubility

Other- solution composition/concentration
- presence of binding agents
- DO, pH, and pE

PHYSICAL: - temperature, light, ionic strength, turbulence

BIOLOGICAL: - stage in life cycle
- physiological condition
- size (surface to volume ratio)
- motility/buoyancy regulation (behaviourial response-avoidance)
- adaptation or tolerance; modify surface environment by reducing the number of binding sites or excretion of organic material which can bind metals and/or sequester metals in cell
- presence of other cells may result in a localized reduction of the available metal concentration

Table 2: Summary of literature dealing with distribution of metals in food webs and temporal variation.

Author(s)	Area of study
Amiard-Triquet et al. (1980)	transfer of Cd, Pb, Cu, and Zn in neritic trophic chains
Anderson (1977)	[Cd, Cu, Pb, and Zn] in 35 genera of freshwater macroinvertebrates
Baptist and Lewis (1969)	distribution of Zn and Cr in an estuarine food chain
Borgmann et al. (1989)	effects of Cd on a lab ecosystem containing *Daphnia* and phytoplankton
Bouquegneau and Martoja (1987)	seasonal variation of [Cd] in *Murex trunculus*
Bouquegneau et al. (1979)	heavy metals in experimental aquatic food chains
Cossa et al. (1980)	seasonal variation metal content of *Mytilus edulis*
Duddridge and Wainwright (1980)	heavy metal accumulation by aquatic fungi and transfer to *Gammarus pulex*
Enk and Mathis (1977)	distribution of Cd and Pb in a stream ecosystem
Fennikoh et al. (1978)	Cd toxicity in planktonic organisms of a freshwater food web
Ferard et al. (1983)	accumulation of Cd in a fresh water food chain experimental model
Fowler and Benayoun (1974)	Cd flux through marine biota

Table 2: (cont'd)

Author(s)	Area of study
Fuller and Averett (1975)	distribution of Cu in an aquatic food chain
Funk et al. (1975)	distribution of heavy metals in an aquatic ecosystem
Guthrie et al. (1979)	biomagnification of heavy metals in a marine microcosm
Hutchinson et al. (1974)	fate of Ni and Cu in an aquatic ecosystem
Kayser (1982)	Cd effects in food chains containing marine algae and filter feeders
Kinkade and Erdman (1975)	[Cd] in a simulated freshwater ecosystem
Mackie et al. (1987)	temporal variation in [metal] of zooplankton in Central Ontario lakes
Mathis and Kevern (1975)	distribution of Hg, Cd, Pb, and Tl in a eutrophic lake
Saward et al. (1975)	effects of Cu on a marine food chain
Schafer et al. (1982)	contaminants in ocean food webs
Selby et al. (1985)	effects of Cd exposure on a hardwater mountain stream microcosm
Thomann et al. (1974)	food chain model of Cd in Lake Erie
Winner et al. (1990)	seasonal variability in sensitivity of freshwater lentic communities to Cu

Table 3: Limnetic phytoplankton taxa found in Hamilton Harbour during the ice-free season of 1975 (reproduced from Harris and Piccinin 1980). Measurements (μm) are either mean values or ranges estimated from Prescott (1962) or Findlay and Kling (1979).

Species Name	Cell Length	Width	Diam.	Colony Diam.
Chlamydomonas sphagnicola (FRITSCH & TAKEDA)			22-28	
Pandorina morum (MUELL.) BORY			4-9	500-850
Eudorina elegans EHRENBERG			16-24	60-200
Golenkinia paucispina WEST & WEST			24	
Pediastrum duplex MEYEN			15-16	105
Echinosphaerella limnetica G.M. SMITH			13	
Treubaria setigerum (ARCHER) G.M. SMITH	27-43	8-12		
Oocystis Borgei SNOW	12-23	10-17		30-60
Coelastrum microporum NAGELI			8-20	
Lagerheimia longiseta (LEMM.) PRINTZ	12	6.5		
Franceia Droescheri (LEMM.) G.M. SMITH	11-13	7-8.5	7-18	
Ankistrodesmus falcatus (CORDA.) RALFS	20-80		1.5-3.5	
Closteriopsis longissima LEMMERMAN	225-530	4-7.5		
Selenastrum minutum (NAEG.) COLLINS	7-10	3-4		
Kirshneriella sp.	2-25	2-8		100-250
Quadrigula lacustris (CHOD.) G.M. SMITH	41	6-7		43
Tetrahedron sp.				
Scenedesmus quadricauda (TURP.) DE BREBISSON	6-25	2.5-10		
Scenedesmus bijuga (TURP.) LAGERHEIM	13	5.5		
Scenedesmus acuminatus (LAG.) CHODAT	10-32	2.5-7		
Scenedesmus denticulatus LAGERHEIM	4-18	4-13		
Actinastrum Hantzschii LAGERHEIM	10-25	1-4		20-50
Tetradesmus wisconsinense G.M. SMITH	12-14.5	4-6		

Table 3: (cont'd)

Species Name	Cell Length	Cell Width	Cell Diam.	Colony Diam.
Dictosphaerium pulchellum WOOD			5-9	20-80
Tetrastrum staurogeniaforme (SCHROEDER)			3-6	
Crucigenia tetrapedia (KIRCH.) WEST & WEST			4.5-9	
Micractinium pusillum FRESENIUS	20-35		3-7	
Mougeotia sp.				
Trachelomonas sp.	22-42	12-27		
Euglena sp.	39-135	12-30		
Phacus sp.	21-66	10-34		
Peridinium sp.	59-72	43-54	54-62	
Chroococcus minutus (KUTZING) NAGELI			4-10	
Merismopedia punctata MEYEN			1.5-3.5	
Oscillatoria sp.				
Anabaena flos-aque (LYNGB.) DE BREBISSON	6-8	4-8		
Closterium acerosum (SHRANK) EHRENBERG				
Cosmarium circulare REINSCH				
Staurastrum paradoxum MEYEN	55-70	50-76		
Asterionella formosa HASSALL	40-130	1-2		
Synedra acus KUTZ	100-300	3-6		
Stephanodiscus hantzschii (GRUN)			8-20	
Stephanodiscus astraea (EHR.) GRUN			30-70	
Cyclotella meneghiniana (KUTZ)			10-30	
Fragilaria crotonensis KITTEN	40-150	2-3		
Melosira granulata (EHR.) RALFS	5-18		5-21	
Cryptomonas rostratiformis SKUJA	48-60	16-26	14-19	
Cryptomonas ovata EHRENBERG	14-68	8-26	7-20	
Rhodomonas minuta SKUJA	9-14	4-7		
Diatoma elongatum AGARDH	40-120	2-4		

FIGURES

Fig. 1: Location of sampling stations and morphometry of Hamilton Harbour. Arrows indicate the major nutrient loads as identified by Klapwijk and Snodgrass (1985). After RAP (1989).

SCALE

0   1000   2000 m

−10− DEPTH CONTOUR m

■ SAMPLING SITES

STP   Sewage Treatment Plant

CANADA

HAMILTON HARBOUR

USA

BURLINGTON

Burlington STP

LAKE ONTARIO

10

Cootes Paradise

10

storm sewers

industry

10

STELCO

DOFASCO

HAMILTON

Hamilton STP

N
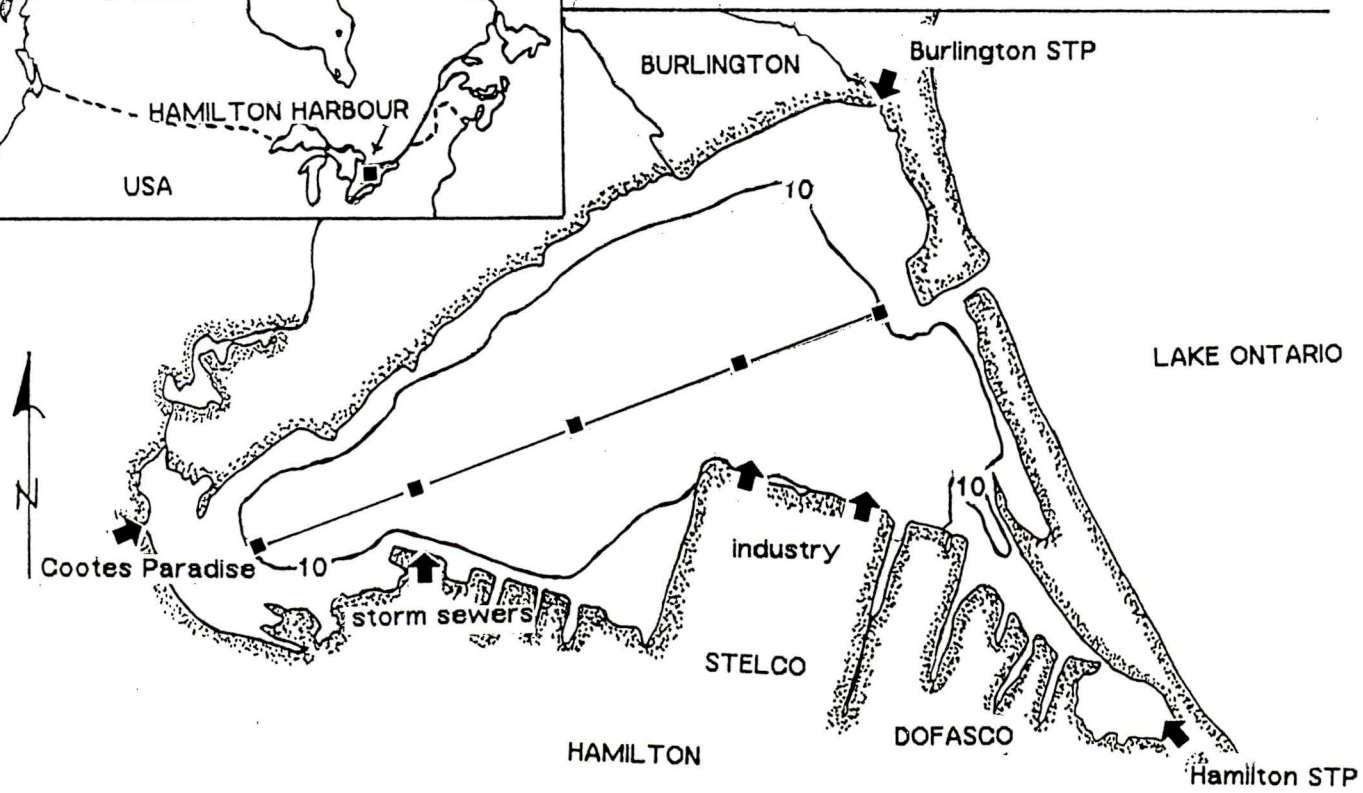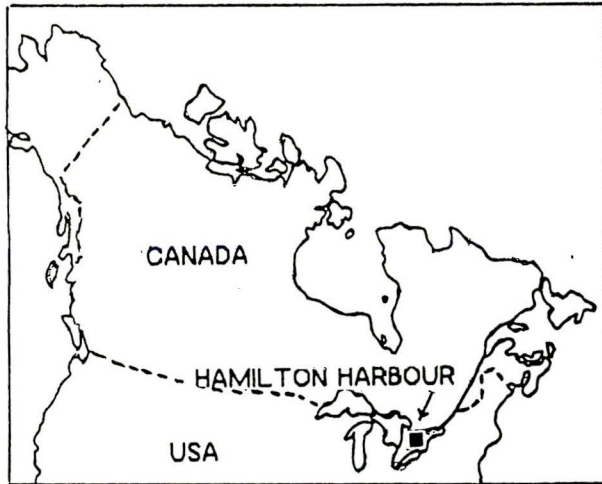
Fig. 2: Zinc concentrations (mg g^{-1}) in Hamilton Harbour surficial sediments. The guideline for open water disposal of dredged material is .1 mg g^{-1}.
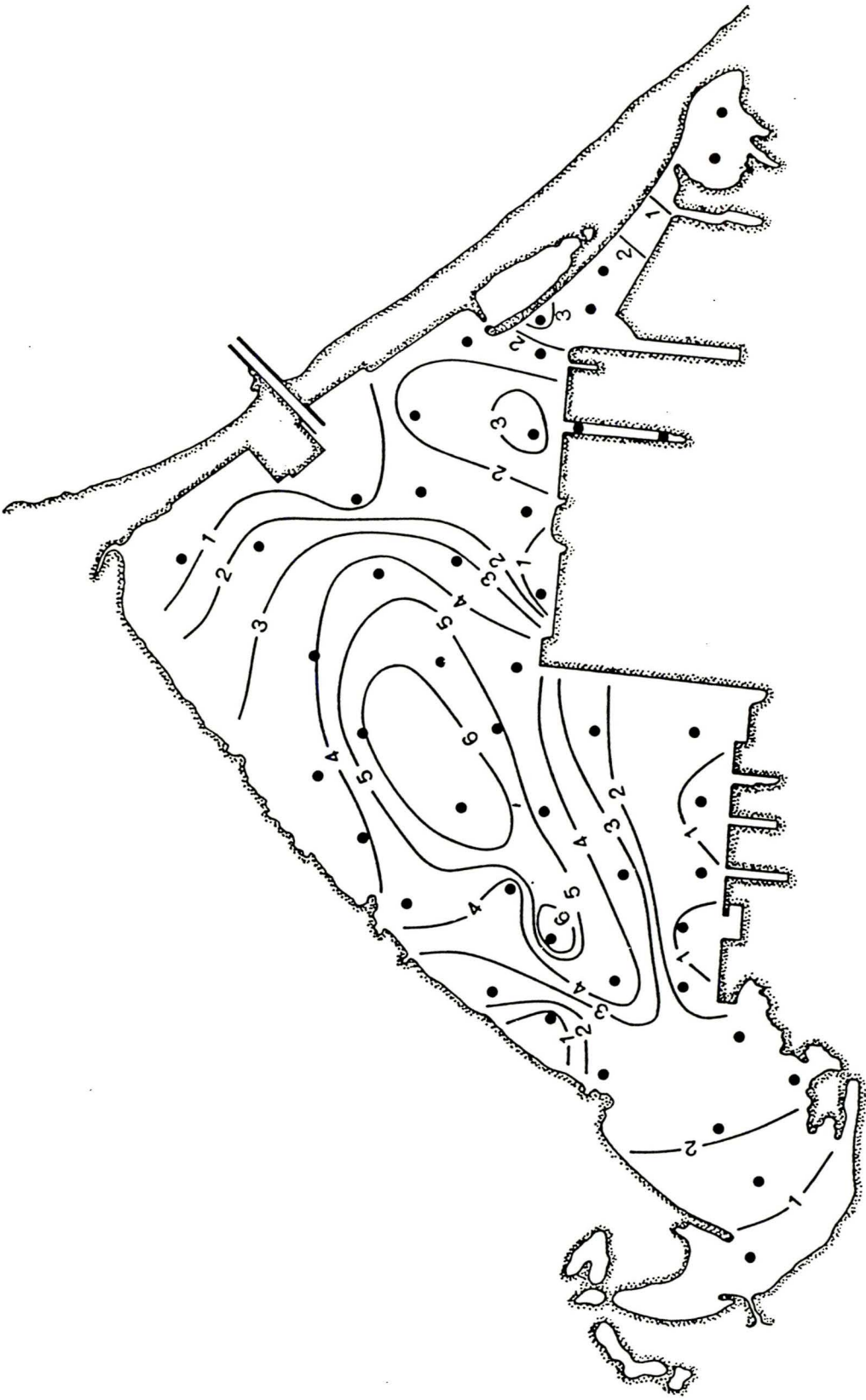
Fig. 3: The apparatus used to sample zooplankton.
Arrows indicate the direction of water flow.
A hose connected to the outflow pipe directed
spent water away from the area of sampling
(not shown).

> 200 μm

> 63 μm

Pump
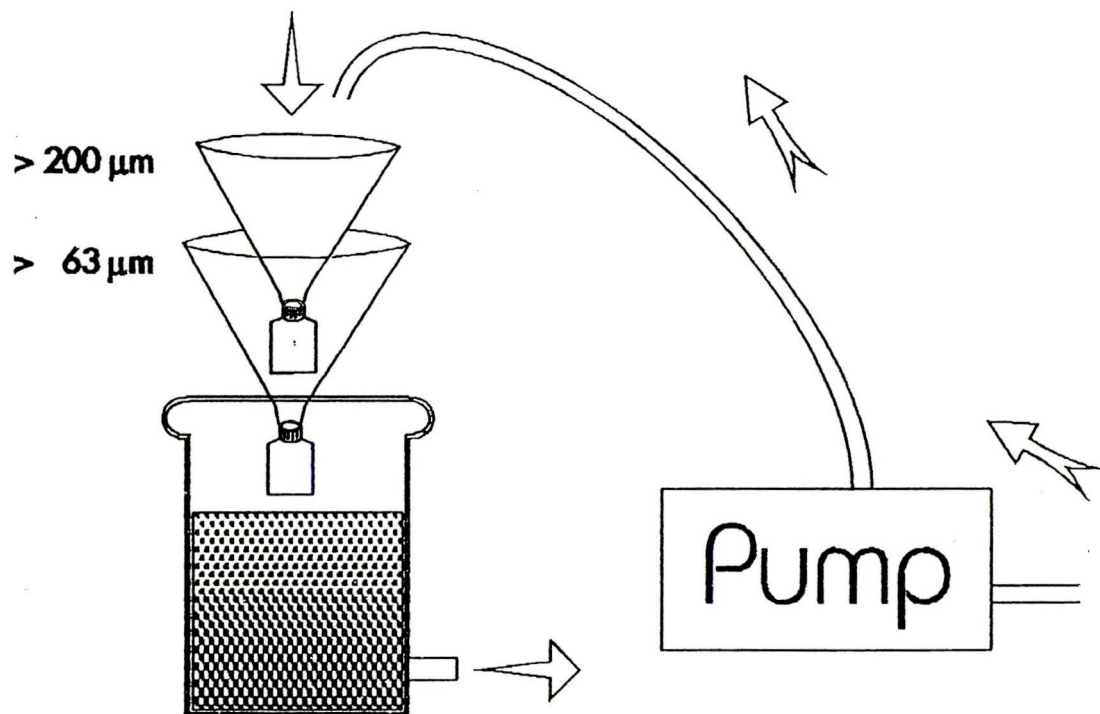
Fig. 4:   Variation in several measured parameters
          between sampling stations and sampling
          periods. **A**:Chlorophyll <u>a</u>, **B**:Secchi depth.
          Sampling period: J(A)=June A, A(B)=August B,
          N(C)=November C etc.   See Appendix - Table 5
          for exact dates of sampling.

LEGEND

Stat. 1 —— Stat. 2 ······
Stat. 3 ······ Stat. 4 —·— Stat. 5 ······
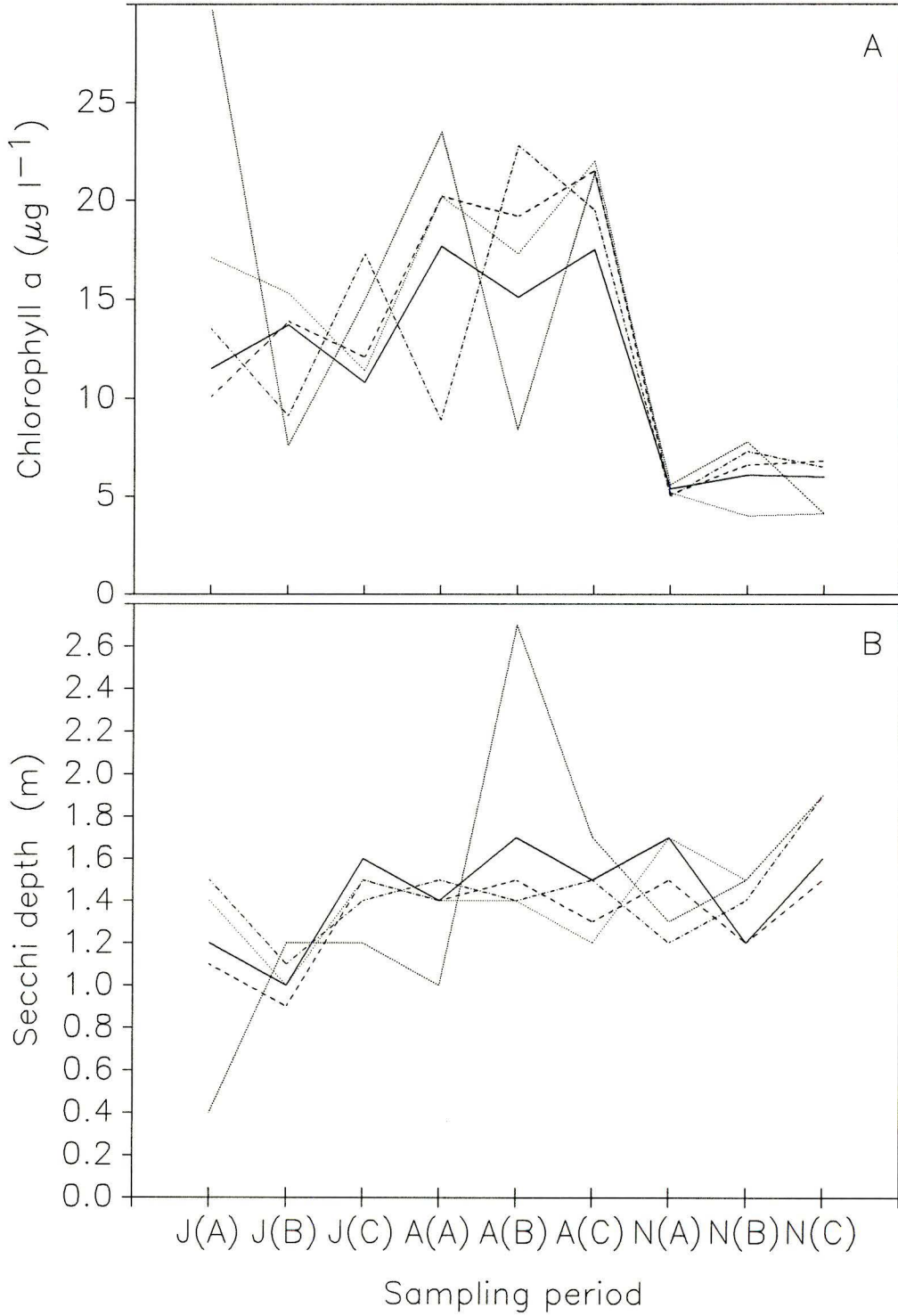
Fig. 5:   Water column temperature profiles for all
           sampling stations. **A**:Station 1, **B**:Station 2,
           C:Station 3, **D**:Station 4, **E**:Station 5.
           Sampling period: J(A)=June A, A(B)=August B,
           N(C)=November C etc. See Appendix - Table 5
           for exact dates of sampling.

LEGEND

Jun A ———   Jun B △——△   Jun C ▲——▲

Aug A ········   Aug B ◇····◇   Aug C ◆····◆

Nov A – – –   Nov B ············   Nov C –·–·–
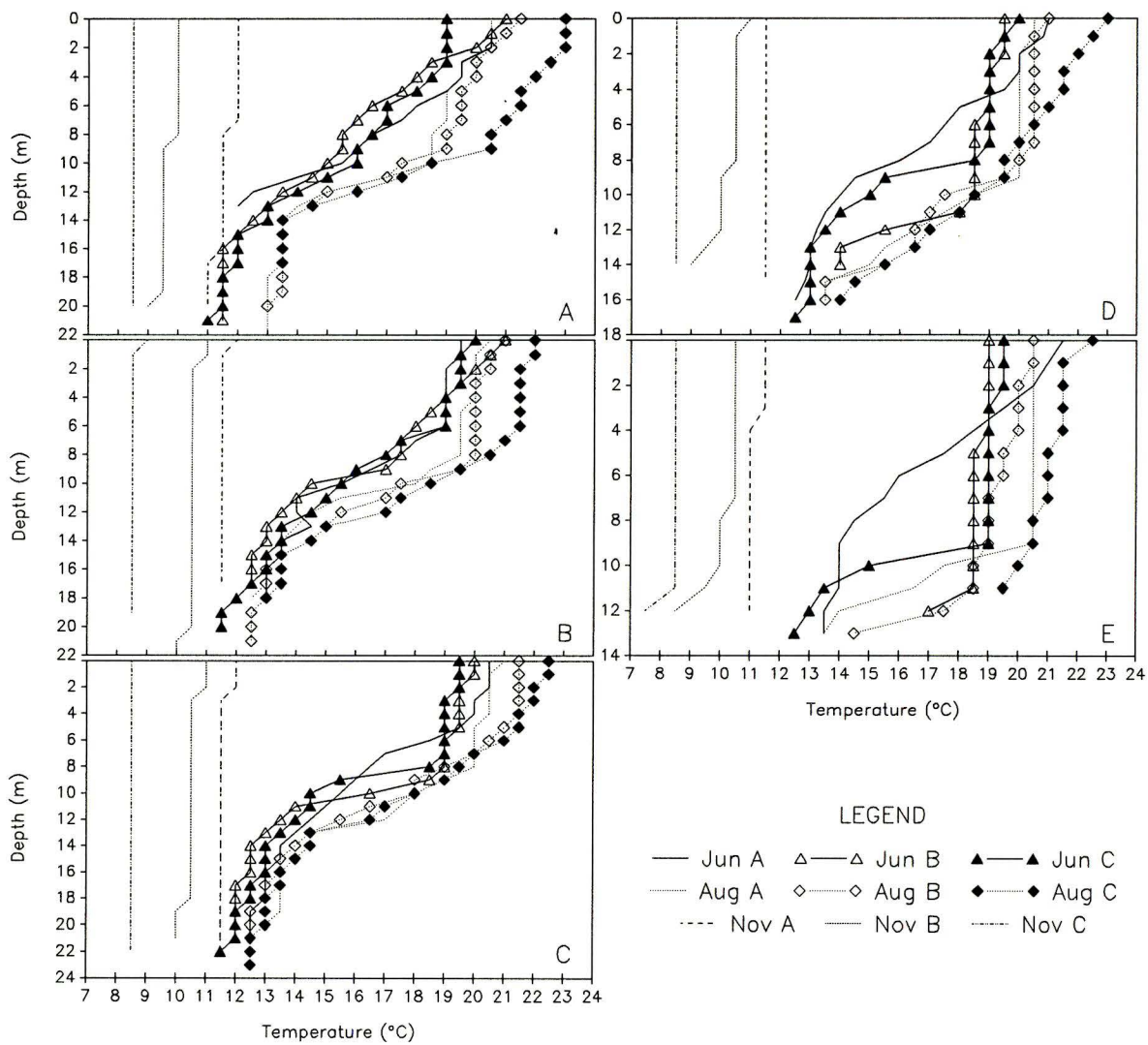
Fig. 6:   Water column dissolved oxygen profiles for all
           sampling stations. **A**:Station 1, **B**:Station 2,
           C:Station 3, **D**:Station 4, **E**:Station 5.
           Sampling period: J(A)=June A, A(B)=August B,
           N(C)=November C etc. See Appendix - Table 5
           for exact dates of sampling.
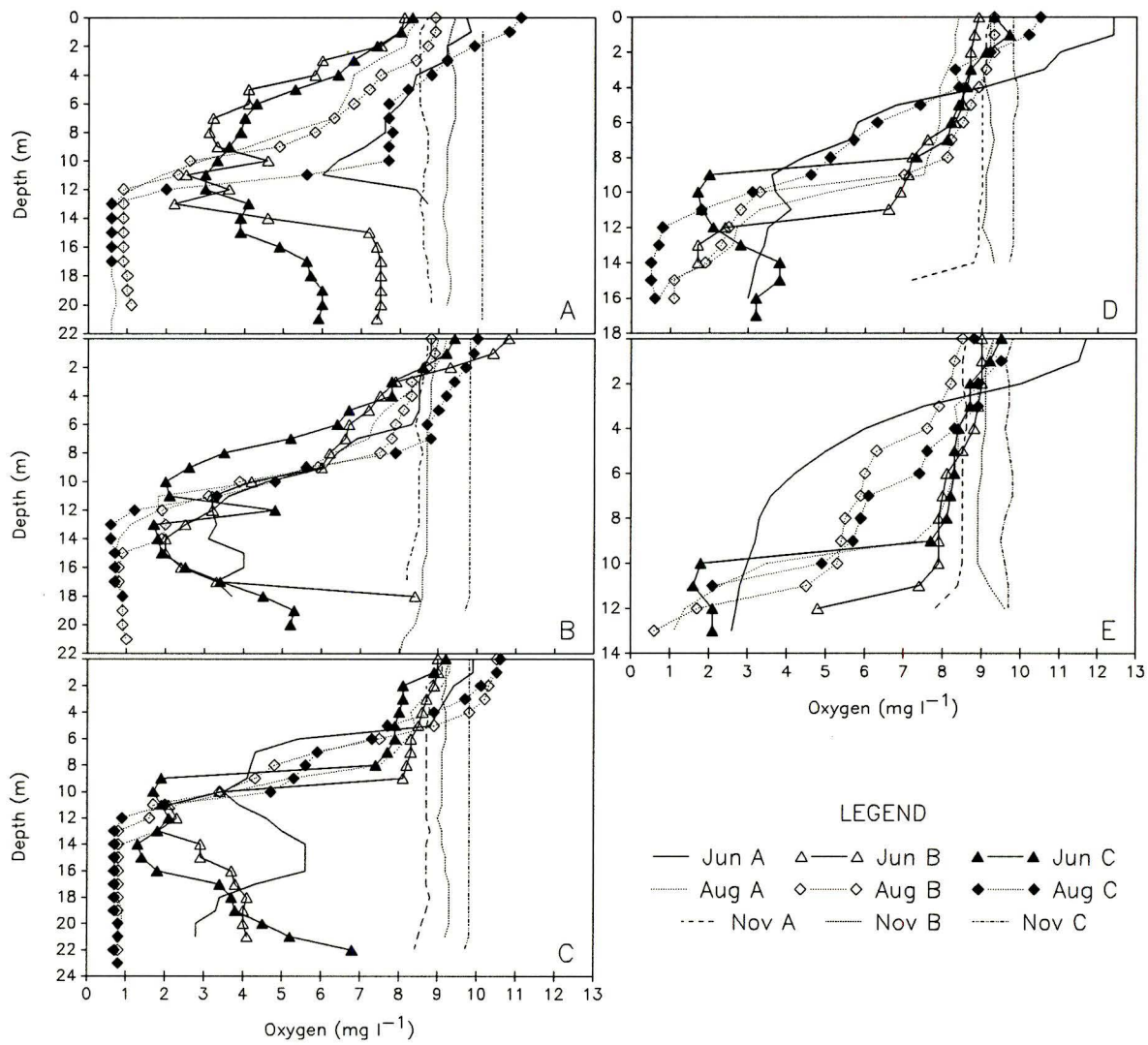
Fig. 7:   Estimated biomass (from length-weight
          regression equations) of the various
          zooplankton groups during each sampling
          period. **A**:Station 1, **B**:Station 2, **C**:Station 3,
          D:Station 4, **E**:Station 5. Sampling period:
          J(A)=June A, A(B)=August B, N(C)=November C
          etc. See Appendix - Table 5 for exact dates
          of sampling.

LEGEND

Harpacticoids   Rotifers

Cladocerans   Cyclopoids

Fig. 8:   Copper concentrations in water and biological
            tissues at each station and sampling period.
            A:Total dissolved in water, **B**:Phytoplankton,
            C:Zooplankton. Sampling period: J(A)=June A,
            A(B)=August B, N(C)=November C etc. See
            Appendix - Table 5 for exact dates of
            sampling.

LEGEND
Stat. 1 ----- Stat. 2
Stat. 3 ----- Stat. 4 ----- Stat.5
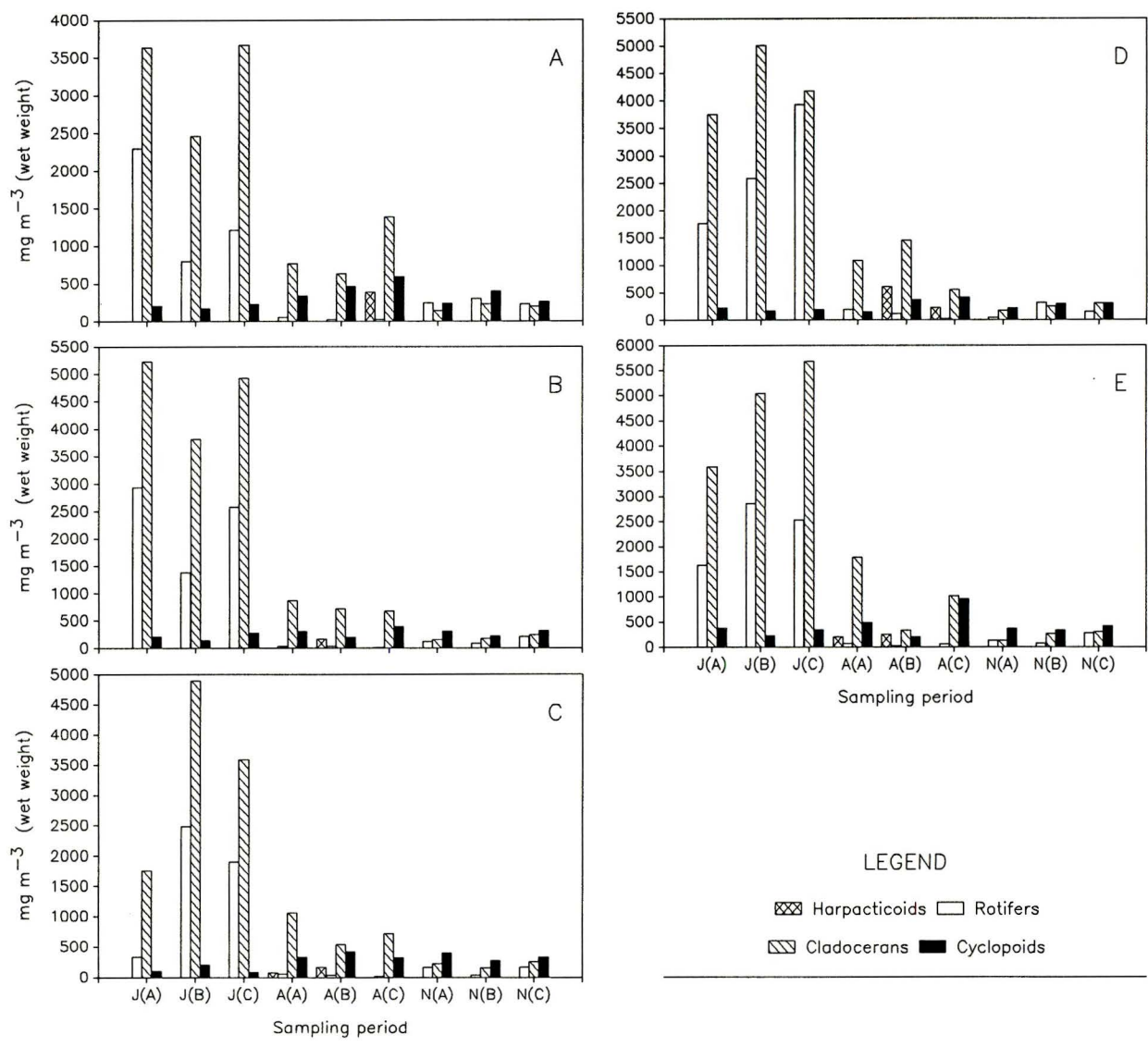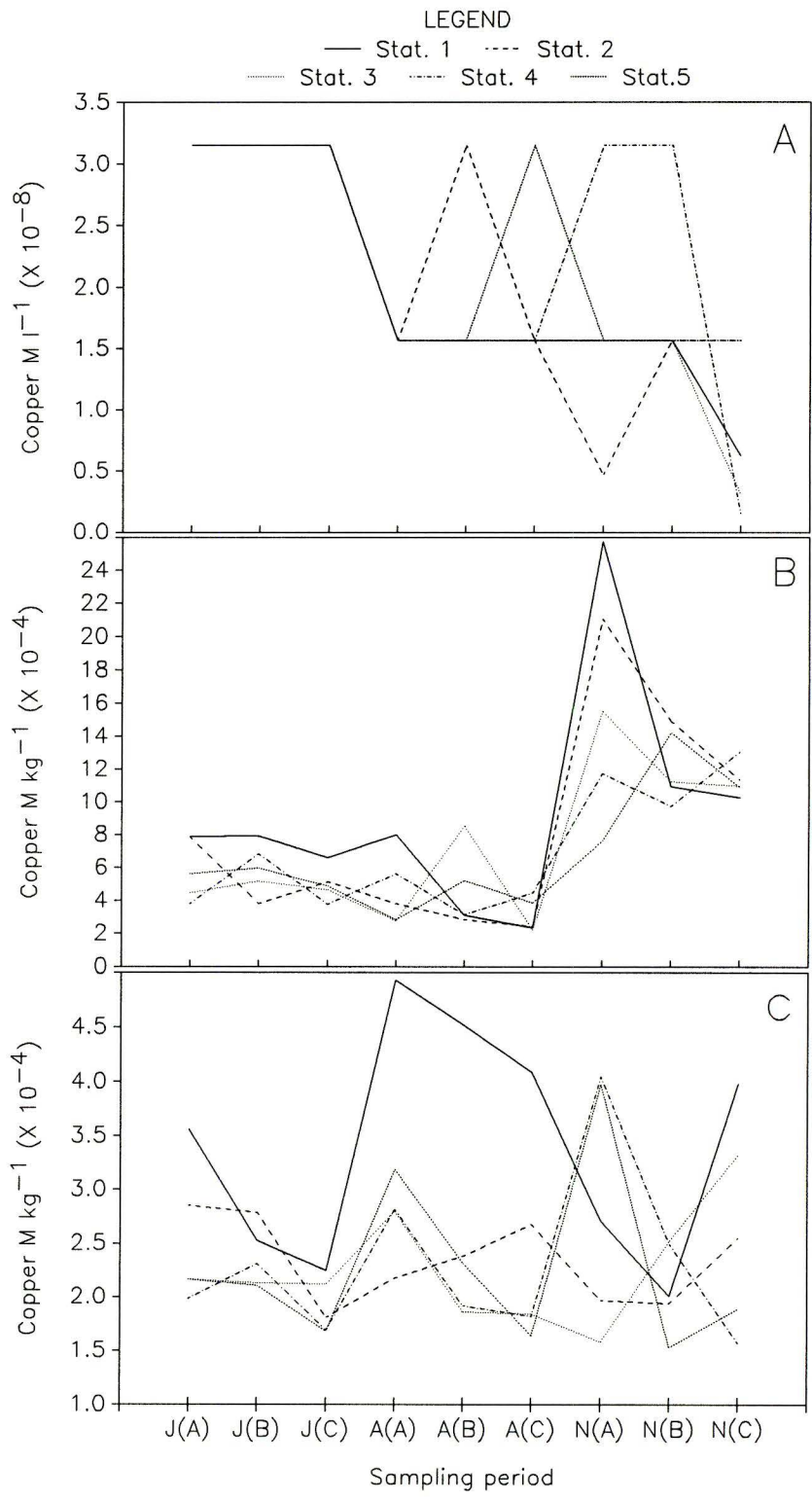
Fig. 9:   Cadmium concentrations in water and
           biological tissues at each station and
           sampling period. **A**:Total dissolved in water,
           B:Phytoplankton, **C**:Zooplankton. Sampling
           period: J(A)=June A, A(B)=August B,
           N(C)=November C etc. See Appendix - Table 5
           for exact dates of sampling.

LEGEND
Stat. 1   Stat. 2
Stat. 3   Stat. 4   Stat. 5
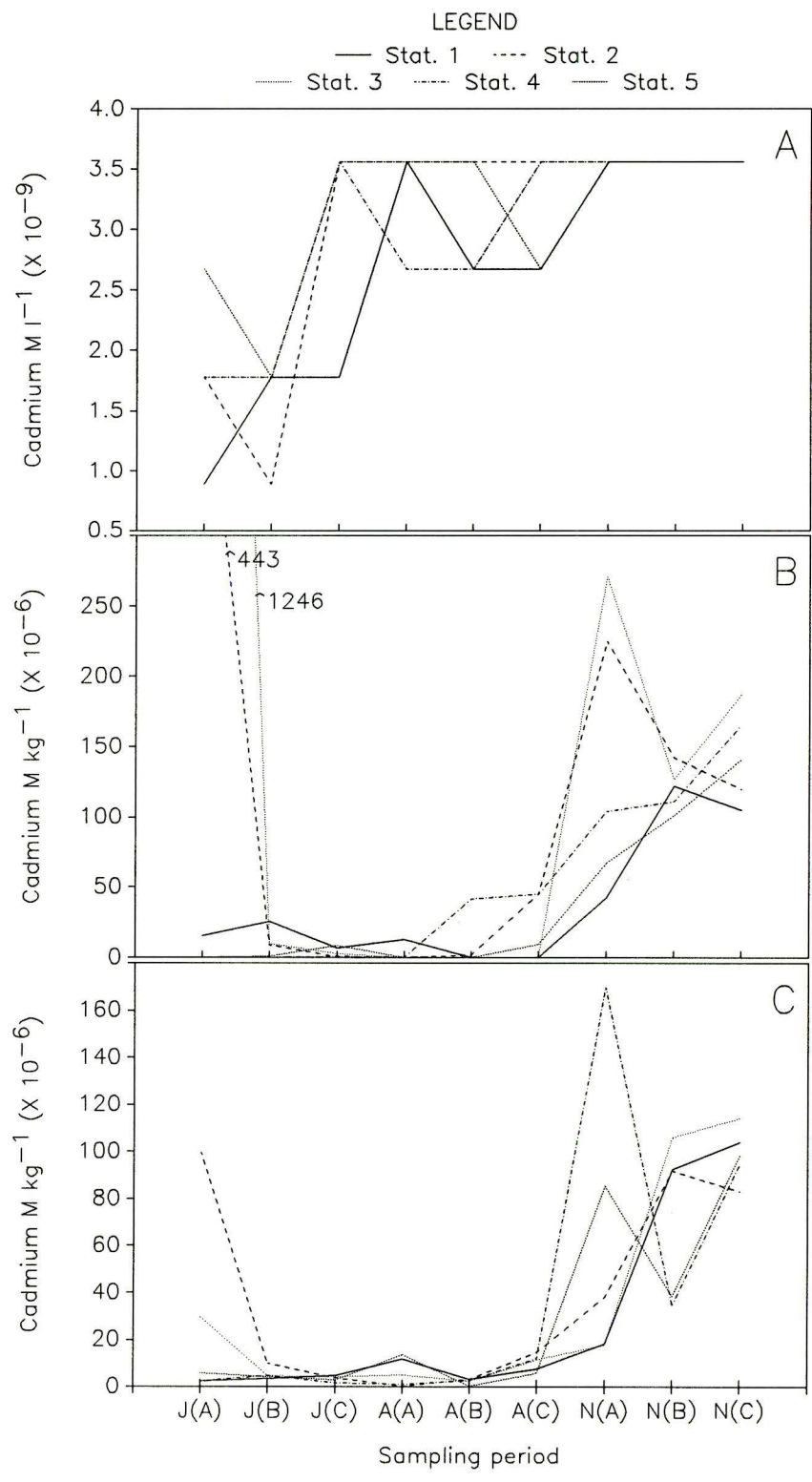
APPENDIX

Table 1:   Main body lengths (i.e. not including mucrons, spines, tails, antennae, bristles, toes, feet, helmets, or paddles) of the zooplankton found Hamilton Harbour (June, August and November 1990). All values are in μm and represent the average of n=100 for all cladocerans and cyclopoids and n=20 for all rotifers. Lengths in **bold** type are average values estimated from size ranges found in Ruttner-Kolisko 1974 and also 10 measured values, or in the case of *Leptodora kindti*, literature values obtained from Balcer et al. 1984.

Taxon	Length
ROTIFERS	
Asplanchna sp.	413
Brachionus angularis	72
Brachionus caudata	**146**
Brachionus calyciflorus	**228**
Brachionus diversicornus	**192**
Brachionus urceolaris	**208**
Filinia sp.	135
Kellicottia longispina	**137**
Keratella cochlearis	85
Keratella quadrata	129
Keratella serrulata	**144**
Lecane sp.	**165**
Notholca sp.	**142**
Ploesoma sp.	**179**
Polyarthra sp.	122
Pompholyx suleata	96
Synchaeta sp.	192
Trichocerca sp.	166
CLADOCERANS	
Bosmina sp.	323
Daphnia sp.	605
CYCLOPOIDS	
Cyclops sp.	503
Nauplii	163
HARPACTICOIDS	
Leptodora kindti	**6950**

Table 2: General form of the equation and specific values used to calculate biomass of the zooplankton, courtesy of Chow-Fraser (pers. comm. 1991).

BIOMASS = a(length)^b

(μg wet weight) = a(mm)^b

Taxon	a	b
Globular rotifers (i.e. *Asplanchna sp.*)	254	3.00
Linear rotifers (i.e. herbivorous rotifers)	25.3	3.00
Bosmina	266	3.13
Daphnia	50	2.84
Cyclopoids	55	2.46
Nauplii	42	2.48
Leptodora	4.4	2.67

Table 3:Abundances of the zooplankton found in the >200 μm fraction. Units are number of organisms/m^3.

Stat.	Time		Asplan.	B.ang.	B.cal.	B.div.	B.urc.
1	Jun	A	1518	72	0	72	0
1	Jun	B	7415	0	0	0	0
1	Jun	C	10797	0	0	0	0
2	Jun	A	17140	0	459	0	0
2	Jun	B	4387	0	0	0	0
2	Jun	C	24571	0	0	0	0
3	Jun	A	5854	0	520	0	0
3	Jun	B	6894	0	0	0	0
3	Jun	C	7516	0	0	0	0
4	Jun	A	8325	0	867	0	0
4	Jun	B	15076	0	0	0	133
4	Jun	C	23704	0	289	0	0
5	Jun	A	16911	0	4770	0	0
5	Jun	B	6622	0	0	0	0
5	Jun	C	33604	434	217	0	0
1	Aug	A	108	12466	0	0	0
1	Aug	B	325	13550	0	0	0
1	Aug	C	100	1101	0	0	0
2	Aug	A	118	10761	0	0	118
2	Aug	B	0	14959	0	0	0
2	Aug	C	0	300	0	0	0
3	Aug	A	0	15210	0	0	0
3	Aug	B	108	12141	0	0	0
3	Aug	C	0	400	0	200	0
4	Aug	A	0	12748	130	130	0
4	Aug	B	390	9886	0	0	0
4	Aug	C	200	1101	0	0	0
5	Aug	A	0	8455	0	390	0
5	Aug	B	108	3794	0	0	0
5	Aug	C	0	260	0	0	0
1	Nov	A	743	0	0	0	0
1	Nov	B	1095	0	0	0	0
1	Nov	C	411	0	0	0	0
2	Nov	A	1301	0	0	0	0
2	Nov	B	496	0	0	0	0
2	Nov	C	145	0	0	0	0
3	Nov	A	991	0	0	0	0
3	Nov	B	130	0	0	0	0
3	Nov	C	496	0	0	0	0
4	Nov	A	0	0	0	0	0
4	Nov	B	0	0	0	0	0
4	Nov	C	1601	0	0	0	0
5	Nov	A	946	0	0	0	0
5	Nov	B	315	0	0	0	0
5	Nov	C	631	0	0	0	0

Table 3: (cont'd)

Stat.	Time		Filin.	Ke.lon.	K.coc.	K.qua.	K.ser.
1	Jun	A	939	0	72	578	0
1	Jun	B	390	130	0	1171	0
1	Jun	C	260	390	585	1691	0
2	Jun	A	3826	306	918	2143	306
2	Jun	B	1632	0	408	612	102
2	Jun	C	1734	578	0	1445	0
3	Jun	A	1171	260	1171	3252	260
3	Jun	B	1041	0	130	130	0
3	Jun	C	1156	0	578	0	0
4	Jun	A	173	0	1214	2602	0
4	Jun	B	1067	133	667	1334	0
4	Jun	C	1734	0	289	289	0
5	Jun	A	2168	0	5203	14743	217
5	Jun	B	1419	0	237	1183	0
5	Jun	C	217	0	217	650	0
1	Aug	A	0	217	759	0	0
1	Aug	B	0	0	325	0	108
1	Aug	C	200	0	500	0	0
2	Aug	A	0	0	355	0	0
2	Aug	B	0	0	434	108	0
2	Aug	C	0	100	600	0	0
3	Aug	A	0	300	1001	150	100
3	Aug	B	325	0	108	0	0
3	Aug	C	200	400	1001	0	100
4	Aug	A	0	0	260	0	0
4	Aug	B	130	130	780	130	0
4	Aug	C	0	100	500	100	0
5	Aug	A	0	0	650	0	0
5	Aug	B	217	108	108	108	0
5	Aug	C	0	0	650	130	0
1	Nov	A	124	0	496	0	0
1	Nov	B	0	0	137	0	0
1	Nov	C	0	0	0	0	0
2	Nov	A	0	0	0	0	0
2	Nov	B	0	0	0	0	0
2	Nov	C	0	0	0	0	0
3	Nov	A	0	0	0	124	0
3	Nov	B	0	0	0	0	0
3	Nov	C	0	0	0	0	0
4	Nov	A	0	0	0	0	0
4	Nov	B	0	0	0	0	0
4	Nov	C	0	0	320	0	0
5	Nov	A	0	237	0	0	0
5	Nov	B	0	0	0	0	0
5	Nov	C	0	0	158	0	0

Table 3: (cont'd)

Stat.	Time		Nothol.	Ploe.	Poly.	P.sul.	Synch.
1	Jun	A	72	72	1807	72	506
1	Jun	B	0	130	260	0	0
1	Jun	C	0	0	0	0	0
2	Jun	A	0	0	3826	612	918
2	Jun	B	0	0	5305	1326	1836
2	Jun	C	0	0	2602	1734	3469
3	Jun	A	0	0	650	0	0
3	Jun	B	0	0	390	260	390
3	Jun	C	0	0	2602	1734	2891
4	Jun	A	0	0	0	0	0
4	Jun	B	0	267	8806	6137	1468
4	Jun	C	0	0	2891	1301	2602
5	Jun	A	0	0	1301	434	0
5	Jun	B	0	237	1183	2129	591
5	Jun	C	0	0	434	0	1301
1	Aug	A	0	217	542	108	0
1	Aug	B	0	0	108	0	0
1	Aug	C	0	100	200	0	0
2	Aug	A	0	118	1064	0	0
2	Aug	B	0	0	2276	650	0
2	Aug	C	0	0	100	100	0
3	Aug	A	0	300	6704	0	100
3	Aug	B	0	0	1626	108	0
3	Aug	C	0	0	801	0	0
4	Aug	A	0	0	2472	0	390
4	Aug	B	0	260	4683	130	0
4	Aug	C	0	0	200	0	0
5	Aug	A	0	260	4553	0	0
5	Aug	B	0	217	1843	0	0
5	Aug	C	0	0	1171	0	0
1	Nov	A	0	0	124	124	0
1	Nov	B	0	0	0	0	0
1	Nov	C	0	0	0	0	0
2	Nov	A	0	0	0	0	0
2	Nov	B	0	0	0	0	0
2	Nov	C	0	0	0	0	0
3	Nov	A	0	0	124	0	0
3	Nov	B	0	0	0	0	0
3	Nov	C	0	0	0	0	0
4	Nov	A	0	0	0	0	0
4	Nov	B	0	0	0	0	0
4	Nov	C	0	0	0	0	0
5	Nov	A	0	0	0	0	0
5	Nov	B	0	0	0	0	0
5	Nov	C	0	0	0	0	0

Table 3: (cont'd)

Stat.	Time		Tricoc.	Bosmin.	Daphn.	Cyclop.	Naup.	L.kin.
1	Jun	A	145	19512	0	72	0	0
1	Jun	B	0	75577	390	5203	130	0
1	Jun	C	260	128260	130	9886	0	0
2	Jun	A	1224	94730	459	5509	765	0
2	Jun	B	714	47033	102	1836	816	0
2	Jun	C	3180	223162	578	3758	0	0
3	Jun	A	0	167675	390	8846	390	0
3	Jun	B	650	114341	260	6114	0	0
3	Jun	C	867	134995	0	4336	289	0
4	Jun	A	0	173442	0	8152	1214	0
4	Jun	B	2001	149026	667	5870	667	0
4	Jun	C	1445	182114	289	4625	289	0
5	Jun	A	0	206179	650	11924	1734	0
5	Jun	B	1301	134339	0	5085	710	0
5	Jun	C	1084	289214	650	8672	1518	0
1	Aug	A	217	12575	26667	9756	0	0
1	Aug	B	0	4770	23306	14092	0	0
1	Aug	C	0	11407	66542	17711	200	500
2	Aug	A	237	13126	27672	10052	0	0
2	Aug	B	650	9648	28943	6938	0	217
2	Aug	C	0	10006	32520	17011	300	0
3	Aug	A	600	13809	37423	8705	100	100
3	Aug	B	325	8130	29485	13659	542	217
3	Aug	C	700	11807	32520	16610	600	0
4	Aug	A	911	23285	23675	3512	0	0
4	Aug	B	260	22374	48520	8455	650	780
4	Aug	C	600	8105	31620	19912	100	300
5	Aug	A	520	29138	43707	7024	650	260
5	Aug	B	0	6829	8238	5420	108	325
5	Aug	C	390	22634	27707	21593	390	0
1	Nov	A	0	7557	4832	6938	743	0
1	Nov	B	0	13145	4656	5614	0	0
1	Nov	C	0	6846	2602	3560	0	0
2	Nov	A	0	10569	3577	10407	163	0
2	Nov	B	0	9911	2726	4212	0	0
2	Nov	C	0	13008	3613	3613	0	0
3	Nov	A	0	11398	6194	9168	124	0
3	Nov	B	0	8065	3252	8065	0	0
3	Nov	C	0	13504	6566	8920	0	0
4	Nov	A	0	10778	2478	4336	124	0
4	Nov	B	0	10140	7338	3202	0	0
4	Nov	C	0	20013	6884	7044	0	0
5	Nov	A	0	7805	2838	8278	0	0
5	Nov	B	0	18921	3627	5834	0	0
5	Nov	C	0	21917	5361	8830	0	0

Table 3:   (cont'd)
List of Abbreviations

Stat.:Station
Asplan.:*Asplanchna sp.*
B.ang.:*Brachionus angularis*
B.cal.:*Brachionus calyciflorus*
B.div.:*Brachionus diversicornus*
B.urc.:*Brachionus urceolaris*
Filin.:*Filinia sp.*
Ke.lon.:*Kellicottia longispina*
K.coc.:*Keratella cochlearis*
K.qua.:*Keratella quadrata*
K.ser.:*Keratella serrulata*

Nothol.:*Notholca sp.*
Ploe.:*Ploesoma sp.*
Poly.:*Polyarthra sp.*
P.sul.:*Pompholyx suleata*
Synch.:*Synchaeta sp.*
Tricoc.:*Trichocerca sp.*
Bosmin.:*Bosmina sp.*
Daphn.:*Daphnia sp.*
Cyclop.:*Cyclops sp.*
Naup.:nauplii (various)
L.kin.:*Leptodora kindti*

Table 4: Abundances of the zooplankton found in the >63 μm fraction. Units are number of organisms/m^3.

Stat.	Time		Asplan.	B.ang.	B.cau.	B.cal.	B.div.
1	Jun	A	122338	1549	0	1549	4646
1	Jun	B	36237	1394	0	0	0
1	Jun	C	54355	1394	0	0	1394
2	Jun	A	144292	0	0	0	0
2	Jun	B	68867	0	0	1093	0
2	Jun	C	114595	3097	0	0	0
3	Jun	A	12098	650	0	780	0
3	Jun	B	126829	5575	0	0	0
3	Jun	C	92915	3097	0	0	0
4	Jun	A	88579	1858	619	4955	0
4	Jun	B	125793	0	0	0	1429
4	Jun	C	188928	0	0	0	0
5	Jun	A	71235	2323	0	4646	2323
5	Jun	B	150776	1267	0	0	1267
5	Jun	C	99884	0	0	2323	4646
1	Aug	A	1549	35617	0	0	0
1	Aug	B	0	23229	0	0	0
1	Aug	C	0	7147	0	0	1429
2	Aug	A	0	23651	0	0	1689
2	Aug	B	0	27875	0	0	0
2	Aug	C	0	6433	0	2144	2144
3	Aug	A	0	44313	0	0	0
3	Aug	B	0	30972	0	0	0
3	Aug	C	0	2859	0	0	0
4	Aug	A	5575	66899	1858	1858	0
4	Aug	B	1858	53891	0	0	0
4	Aug	C	0	2859	0	0	1429
5	Aug	A	0	61324	0	0	5575
5	Aug	B	0	0	0	0	1549
5	Aug	C	1858	3717	0	0	1858
1	Nov	A	12530	0	0	0	0
1	Nov	B	15404	734	0	0	0
1	Nov	C	12226	0	0	0	0
2	Nov	A	5226	0	0	0	0
2	Nov	B	4646	0	0	0	0
2	Nov	C	11356	0	0	0	0
3	Nov	A	7964	0	0	0	0
3	Nov	B	1858	465	0	0	0
3	Nov	C	8407	442	0	0	0
4	Nov	A	2212	0	0	0	0
4	Nov	B	17154	0	0	0	0
4	Nov	C	6861	0	0	0	0
5	Nov	A	5913	0	0	0	0
5	Nov	B	3942	0	0	0	0
5	Nov	C	14078	0	0	0	0

Table 4:   (cont'd)

Stat.	Time	B.urc.	Filin.	Ke.lon.	K.coc.	K.qua.
1	Jun A	0	72784	0	40263	20132
1	Jun B	1394	12544	1394	16725	19512
1	Jun C	0	8362	6969	27875	15331
2	Jun A	0	54109	0	31154	59028
2	Jun B	0	50284	4919	12024	28421
2	Jun C	0	40263	0	49555	12389
3	Jun A	0	7415	390	21724	22634
3	Jun B	0	83624	4181	13937	5575
3	Jun C	0	27875	3097	27875	3097
4	Jun A	1858	21680	4336	192644	111498
4	Jun B	0	80050	0	30019	15724
4	Jun C	0	34069	0	21680	0
5	Jun A	774	17809	0	192799	182733
5	Jun B	0	50681	2534	21539	11403
5	Jun C	0	44135	0	27875	2323
1	Aug A	0	0	6194	139373	1549
1	Aug B	0	3097	4646	102207	1549
1	Aug C	0	7147	2859	195837	0
2	Aug A	0	1689	3379	114877	0
2	Aug B	0	0	3097	122338	6194
2	Aug C	0	0	2859	165103	0
3	Aug A	0	5718	6433	137229	4288
3	Aug B	0	0	3097	92915	3097
3	Aug C	0	4288	10006	125793	2859
4	Aug A	0	1858	0	0	0
4	Aug B	0	9292	0	157956	1858
4	Aug C	0	0	7147	172965	2859
5	Aug A	0	3717	0	148664	3717
5	Aug B	0	0	0	82075	1549
5	Aug C	0	0	5575	262021	1858
1	Nov A	0	248	1752	92827	0
1	Nov B	0	0	734	88025	0
1	Nov C	0	0	2445	61128	489
2	Nov A	0	0	581	133566	0
2	Nov B	0	0	2655	63050	664
2	Nov C	0	516	1549	80527	516
3	Nov A	0	885	2655	121232	0
3	Nov B	0	0	1394	70151	465
3	Nov C	0	0	1327	72562	0
4	Nov A	0	0	2212	78757	0
4	Nov B	0	0	2382	96251	0
4	Nov C	0	0	1715	87483	0
5	Nov A	0	0	1689	94605	0
5	Nov B	0	0	1126	56312	0
5	Nov C	0	0	1689	83342	1126

Table 4:   (cont'd)

Stat.	Time		K.ser.	Lecane	Nothol.	Ploe.	Poly.
1	Jun	A	1549	0	1549	0	639566
1	Jun	B	1394	0	0	0	54355
1	Jun	C	1394	0	2787	0	221603
2	Jun	A	0	0	0	0	423038
2	Jun	B	2186	0	0	1093	415386
2	Jun	C	9292	0	0	0	442896
3	Jun	A	260	0	130	0	83902
3	Jun	B	1394	0	0	0	440418
3	Jun	C	3097	0	0	0	380952
4	Jun	A	6194	0	4955	2478	265738
4	Jun	B	1429	2859	0	0	397391
4	Jun	C	0	0	0	0	699961
5	Jun	A	12389	774	0	774	528068
5	Jun	B	1267	0	0	0	233133
5	Jun	C	2323	0	0	0	694541
1	Aug	A	0	0	0	35617	356175
1	Aug	B	1549	0	0	0	271003
1	Aug	C	0	0	0	14295	315912
2	Aug	A	0	0	0	27030	586211
2	Aug	B	1549	0	0	40263	453736
2	Aug	C	7147	0	0	1429	198696
3	Aug	A	0	0	0	34307	909140
3	Aug	B	1549	0	0	12389	577623
3	Aug	C	1429	0	0	0	291611
4	Aug	A	0	0	0	65041	1661324
4	Aug	B	7433	0	0	70616	1267364
4	Aug	C	1429	0	0	11436	244438
5	Aug	A	1858	0	0	33449	1347271
5	Aug	B	1549	0	0	9292	396438
5	Aug	C	0	0	0	18583	314053
1	Nov	A	248	0	0	4336	35467
1	Nov	B	1467	0	0	2934	38144
1	Nov	C	0	0	0	0	10270
2	Nov	A	0	0	581	581	30778
2	Nov	B	664	0	0	1327	13937
2	Nov	C	516	0	0	0	24777
3	Nov	A	0	0	0	0	22123
3	Nov	B	0	465	0	1858	13008
3	Nov	C	400	0	0	442	14159
4	Nov	A	0	0	0	885	13716
4	Nov	B	0	0	0	2859	23348
4	Nov	C	572	0	0	1715	24587
5	Nov	A	0	0	0	2534	32098
5	Nov	B	563	0	0	0	7321
5	Nov	C	0	0	0	0	35477

Table 4:   (cont'd)

Stat.	Time		P.sul.	Synch.	Tricoc.	Bosmin.	Daphn.
1	Jun	A	210608	201316	17034	449090	0
1	Jun	B	75261	59930	6969	241115	0
1	Jun	C	68293	167247	34843	344251	0
2	Jun	A	95101	75425	12298	578807	0
2	Jun	B	252511	188017	24049	443807	0
2	Jun	C	520325	291134	40263	411924	0
3	Jun	A	9626	7154	650	58407	0
3	Jun	B	170035	352613	40418	517073	0
3	Jun	C	631823	368564	55749	328300	0
4	Jun	A	28494	22300	1858	310337	0
4	Jun	B	527473	114357	52890	496024	0
4	Jun	C	514131	390244	37166	356175	0
5	Jun	A	30197	14712	774	255517	0
5	Jun	B	401647	69686	35477	515679	0
5	Jun	C	383275	511034	11614	441347	0
1	Aug	A	9292	0	21680	37166	4646
1	Aug	B	18583	0	18583	21680	12389
1	Aug	C	27160	0	10006	47172	11436
2	Aug	A	20272	0	30409	40545	10136
2	Aug	B	60395	0	15486	18583	12389
2	Aug	C	47887	1429	12150	25016	1429
3	Aug	A	17154	0	23586	42884	14295
3	Aug	B	10840	0	1549	13937	1549
3	Aug	C	8577	0	2859	25730	2859
4	Aug	A	29733	0	59466	70616	5575
4	Aug	B	24158	0	13008	66899	14866
4	Aug	C	2859	0	15724	7147	4288
5	Aug	A	26016	5575	20441	118931	9222
5	Aug	B	1549	0	0	15486	4646
5	Aug	C	29733	0	22300	39024	16725
1	Nov	A	0	0	619	3256	0
1	Nov	B	5135	734	1467	8802	0
1	Nov	C	1956	0	489	11737	1956
2	Nov	A	6388	0	1742	4065	0
2	Nov	B	3318	0	1327	7300	1327
2	Nov	C	2065	516	516	11356	516
3	Nov	A	9292	885	1327	6194	1327
3	Nov	B	3252	465	929	6504	0
3	Nov	C	1770	442	0	5752	2212
4	Nov	A	1770	0	0	7079	885
4	Nov	B	2859	0	0	9530	476
4	Nov	C	2859	0	572	8005	572
5	Nov	A	3379	0	0	5068	0
5	Nov	B	4505	0	563	7884	563
5	Nov	C	4505	0	0	8447	0

Table 4:   (cont'd)

Stat.	Time	Cyclop.	Naup.
1	Jun A	18583	29423
1	Jun B	11150	26481
1	Jun C	11150	34843
2	Jun A	13117	40992
2	Jun B	10931	29514
2	Jun C	21680	37166
3	Jun A	1691	2081
3	Jun B	12544	41812
3	Jun C	3097	21680
4	Jun A	13628	25397
4	Jun B	10006	28589
4	Jun C	12389	43360
5	Jun A	21680	78978
5	Jun B	15204	34210
5	Jun C	23229	34843
1	Aug A	21680	49555
1	Aug B	27875	78978
1	Aug C	35737	110069
2	Aug A	16894	65885
2	Aug B	10840	44909
2	Aug C	17868	80765
3	Aug A	21442	52890
3	Aug B	24777	57298
3	Aug C	11436	72903
4	Aug A	9292	55749
4	Aug B	26016	52033
4	Aug C	17154	77191
5	Aug A	37166	76190
5	Aug B	10840	78978
5	Aug C	63182	195122
1	Nov A	14831	32742
1	Nov B	32276	42545
1	Nov C	20539	44012
2	Nov A	18002	37166
2	Nov B	15265	45794
2	Nov C	25294	48522
3	Nov A	28317	41591
3	Nov B	17189	40883
3	Nov C	22123	44245
4	Nov A	15928	36724
4	Nov B	23824	45266
4	Nov C	21156	44599
5	Nov A	25341	42234
5	Nov B	24214	48992
5	Nov C	29282	48992

Table 4:   (cont'd)

List of Abbreviations

Stat.:Station
Asplan.:*Asplanchna sp.*
B.ang.:*Brachionus angularis*
B.cau.:*Brachionus caudata*
B.cal.:Brachionus calyciflorus
B.div.:Brachionus diversicornus
B.urc.:*Brachionus urceolaris*
Filin.:*Filinia sp.*
Ke.lon.:*Kellicottia longispina*
K.coc.:*Keratella cochlearis*
K.qua.:*Keratella quadrata*
K.ser.:*Keratella serrulata*

Lecane:*Lecane sp.*
Nothol.:*Notholca sp.*
Ploe.:*Ploesoma sp.*
Poly.:*Polyarthra sp.*
P.sul.:*Pompholyx suleata*
Synch.:*Synchaeta sp.*
Tricoc.:*Trichocerca sp.*
Bosmin.:*Bosmina sp.*
Daphn.:*Daphnia sp.*
Cyclop.:*Cyclops sp.*
Naup.:nauplii (various)
L.kin.:*Leptodora kindti*

Table 5: Abbreviated and actual sampling dates for all
stations. All sampling took place during 1990.

Station	Abbreviation for Sampling Date	Actual Sampling Date
1	J(A), June A	June 27
1	J(B), June B	June 28
1	J(C), June C	June 29
1	A(A), August A	August 23
1	A(B), August B	August 24
1	A(C), August C	August 28
1	N(A), November A	October 31
1	N(B), November B	November 8
1	N(C), November C	November 14
2	J(A), June A	June 26
2	J(B), June B	June 27
2	J(C), June C	June 29
2	A(A), August A	August 23
2	A(B), August B	August 24
2	A(C), August C	August 29
2	N(A), November A	November 1
2	N(B), November B	November 8
2	N(C), November C	November 14
3	J(A), June A	June 21
3	J(B), June B	June 28
3	J(C), June C	June 29
3	A(A), August A	August 24
3	A(B), August B	August 27
3	A(C), August C	August 29
3	N(A), November A	November 1
3	N(B), November B	November 8
3	N(C), November C	November 14
4	J(A), June A	June 20
4	J(B), June B	June 28
4	J(C), June C	June 29
4	A(A), August A	August 23
4	A(B), August B	August 24
4	A(C), August C	August 29
4	N(A), November A	November 3
4	N(B), November B	November 8
4	N(C), November C	November 14
5	J(A), June A	June 20
5	J(B), June B	June 28
5	J(C), June C	June 29
5	A(A), August A	August 24
5	A(B), August B	August 27
5	A(C), August C	August 30
5	N(A), November A	October 31
5	N(B), November B	November 8
5	N(C), November C	November 14

Table 6: Pearson correlation coefficients for log values of
zooplankton, phytoplankton and water Cu
concentrations, chl <u>a</u> concentrations and percentage
of total biomass contributed by rotifers,
cladocerans, and cyclopoids. Probability levels
are stated in brackets; n=45.

	ZoopCu	WaterCu	PhyCu	Chl <u>a</u>	%Rotif	%Cladoc	%Cyclop
ZoopCu	1.000 (0)	-.046 (.761)	-.071 (.644)	-.061 (.681)	-.164 (.281)	.004 (.978)	.132 (.386)
WaterCu		1.000 (0)	-.356 (.016)	.388 (.008)	.136 (.370)	.462 (.001)	-.601 (.0001)
PhyCu			1.000 (0)	-.817 (.0001)	.564 (.0001)	-.728 (.0001)	.339 (.022)
Chl <u>a</u>				1.000 (0)	-.507 (.0004)	.761 (.0001)	-.345 (.020)
%Rotif					1.000 (0)	-.247 (.100)	-.452 (.001)
%Cladoc						1.000 (0)	-.662 (.0001)
%Cyclop							1.000 (0)

Table 7: Pearson correlation coefficients for log values of
zooplankton, phytoplankton and water Cd
concentrations, chl <u>a</u> concentrations and percentage
of total biomass contributed by rotifers,
cladocerans, and cyclopoids. Probability levels
are stated in brackets; n=41.

	ZoopCd	WaterCd	PhyCd	Chl <u>a</u>	%Rotif	%Cladoc	%Cyclop
ZoopCd	1.000 (0)	.321 (.040)	.766 (.0001)	-.630 (.0001)	.220 (.165)	-.687 (.0001)	.547 (.0002)
WaterCd		1.000 (0)	.166 (.299)	-.158 (.321)	-.272 (.084)	-.430 (.005)	.612 (.0001)
PhyCd			1.000 (0)	-.596 (.0001)	.338 (.030)	-.665 (.0001)	.398 (.009)
Chl <u>a</u>				1.000 (0)	-.510 (.0006)	.740 (.0001)	-.305 (.052)
%Rotif					1.000 (0)	-.257 (.103)	-.473 (.001)
%Cladoc						1.000 (0)	-.628 (.0001)
%Cyclop							1.000 (0)8

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