Assessment of lower food web in Hamilton Harbour, Lake Ontario, 2002 - 2004.

R. Dermott, O. Johannsson, M. Munawar, R. Bonnell, K. Bowen, M. Burley, M. Fitzpatrick, J. Gerlofsma, and H. Niblock

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ABSTRACT

Dermott, R., Johannsson, O., Munawar, M., Bonnell, R., Bowen, K., Burley, M., Fitzpatrick, M., Gerlofsma, J., Niblock, H. 2007. Assessment of lower food web in Hamilton Harbour, Lake Ontario, 2002 -2004.

As an Area of Concern on the Great Lakes, Hamilton Harbour is an extremely stressed environment with high nutrient levels and contaminated sediments. Remediation efforts have greatly improved water quality and encouraged habitat restoration. Fisheries and Oceans undertook a comprehensive program to examine components of the harbour's food-web, from microbes up to fish.

Bi-weekly sampling between May and October in 2002, 2003 and 2004 examined environmental and biological components simultaneously at the same sites. Site location changed each year. In 2002 and 2003, a nearshore to offshore gradient was examined in the middle (3 sites), or western end of the harbour (2 sites), between 1.5 to 24 m deep. The two offshore sites were re-examined in 2004. Depth profiles of temperature, oxygen, and light were taken, and samples collected for water chemistry, chlorophyll *a*, seston, total and size fractionated primary production, bacteria, ciliates, phytoplankton, and zooplankton. Benthic samples were collected at 3 seasons only in 2002 and 2003. In 2002, a spatial survey was conducted for most components at a number of sites in the harbour.

Annual average phosphorous concentrations at the sites ranged from 25 $\mu g\ l^{-1}$ to 34 $\mu g\ l^{-1}$, above the Remedial Action Plan goal of 17 $\mu g\ l^{-1}$. Anoxic conditions (dissolved oxygen < 1 mg l^{-1}) were observed at depths > 10 m from late June until October. The phytoplankton community was variable with peaks of Diatomeae, Dinophyceae, Chlorophyta and Chrysophyceae throughout the seasons. Primary productivity was dominated by larger phytoplankton (> 20 µm), except in late spring/early summer when picoplankton (<2 µm) contributed 40% of the size fractionated production. The shallowest site (1.5 m) supported the highest zooplankton production (10,188 mg dry m^{-3}). Zooplankton production ranged between 2211 and 4934 mg m^{-3} at sites >3 m deep. The shallow site supported a diverse cladoceran community dominated by benthic or plant-associated taxa such as *Eurycercus*, *Sida* and *Alona*. Dominant zooplankton offshore were *Bosmina spp., Daphnia retrocurva, Eubosmina coregoni*, and juvenile Cyclopoid copepods. *Dreissena* veligers were common in 2002 and 2003. Rotifer biomass was 1-3% that of zooplankton biomass. Low oxygen restricted the benthic community, so that in mid-harbour, the composition was almost exclusively tubificid worms (18.8 g m^{-2} wet). Chironomidae were common only at shallower sites on the north and west end of the harbour. *Dreissena polymorpha* was common at sites along the north shore, but few were present below 8 m depth. This report provides information about the food web of the harbour.

RÉSUMÉ

Dermott, R., Johannsson, O., Munawar, M., Bonnell, R., Bowen, K., Burley, M., Fitzpatrick, M., Gerlofsma, J., Niblock, H. 2007. Évaluation des maillons inférieurs du réseau alimentaire du havre Hamilton (lac Ontario), 2002 -2004.

Le havre Hamilton, un secteur préoccupant des Grands Lacs, est un milieu hautement perturbé dont les taux en nutriments sont élevés et les sédiments sont contaminés. La qualité de l'eau s'est beaucoup améliorée et la restauration de l'habitat est favorisée grâce aux efforts d'assainissement. Pêches et Océans Canada a entrepris un programme intégré de mise à l'étude des composants du réseau alimentaire du havre Hamilton, du microorganisme jusqu'au poisson.

Un échantillonnage bimensuel, de mai à octobre en 2002, 2003 et 2004 avait pour but d'étudier, simultanément, des composants environnementaux et biologiques aux mêmes sites. Les lieux de sites variaient chaque année. En 2002 et 2003, un gradient côtier - extracôtier au milieu du havre (à 3 sites) ou à l'extrémité ouest du havre (à 2 sites), aux sites où la profondeur était de 1.5 à 24 m était à l'étude. On a réétudié les deux sites côtiers en 2004. Des profils de concentration pour la température, l'oxygène et la lumière ont été faits et des échantillons ont été pris pour la composition chimique de l'eau, la chlorophylle *a*, le seston, la production primaire totale et fractionnée en fonction de la taille, les bactéries, les ciliés, le phytoplancton et le zooplancton. En 2002 et en 2003, des échantillons benthiques ont été pris pendant trois saisons seulement. En 2002, on a fait une évaluation spatiale à plusieurs sites portant sur certains des composants nommés ci-dessus.

La concentration moyenne en phosphore aux sites variait de 25 $\mu g\ l^{-1}$ à 34 $\mu g\ l^{-1}$, donc au-dessus de l'objectif de 17 $\mu g\ l^{-1}$ du Plan de Mesures Correctives. On a observé des conditions anoxiques (oxygène dissous de < 1 mg l^{-1}) à des profondeurs de > 10 m, à partir de la fin dejuin jusqu'en octobre. La composition de la population phytoplanctonique était variable : le nombre de diatomées, dinophycées, chlorophycées et chrysophyées montait à la hausse en toute saison. Les grandes cellules phytoplanctoniques (> 20 µm) dominaient la production primaire, sauf vers la fin du printemps et le début de l'été lorsque le picoplancton (< 2 µm) faisait 40 % de la production fractionnée en fonction de la taille. Le site le moins profond (1,5 m) soutenait la production la plus importante en zooplancton (10,188 mg secs m^{-3}). La production en zooplancton variait de 2 211 à 4934 mg m^{-3} aux sites de >3 m de profondeur. Le site peu profond soutenait des cladocères divers dont les taxa benthiques ou végétaux *Eurycerus*, *Sida* et *Alona* dominaient, entre autres. Au large, *Bosmina spp., Daphnia retrocurva, Eubosmina coregoni* et des copépodes cyclopoides juvéniles étaient le zooplancton dominant. Des *Dreissena* au stade veligère étaient communes en 2002 et 2003. La biomasse en rotifères était de 1 à 3 % de la biomasse en zooplancton. Un bas niveau d'oxygène compromettait la population benthique du havre et donc, au milieu du havre, elle était composée presque exclusivement de tubificidés (18.8 g m^{-2} humides). Des chironomes étaient communs uniquement aux sites moins profonds des extrémités nord et ouest du havre. Le dreissena polymorphe était commune aux sites du littoral nord, mais peu présent au-dessous de 8 m de profondeur. Le présent rapport est au sujet du réseau d'alimentation du havre.

ASSESSMENT OF HAMILTON HARBOUR; BACKGROUND

Ronald Dermott

INTRODUCTION

Hamilton Harbour is a major industrial port at the west end of Lake Ontario. It has had a long history of pollution from industrial and municipal wastes, culminating in its description as the largest and most beautiful septic tank in the world (Matheson 1958). Early concerns were high concentrations of bacteria and phenolic substances in the water. Hypolimnetic oxygen depletion occurs during thermal stratification from June until late September. Total nitrogen levels had been between 1 and 2 mg l^{-1} in 1949 and up to 3 mg l^{-1} in 1976 (Matheson 1958, Piccinin 1977). In 1950, total phosphate was 0.04 with levels up to 0.08 mg l^{-1} measured during the summer (Matheson 1958). The water quality and biological community of the harbour have been impaired both from excessive nutrient and ammonia loadings from municipal sewage from the surrounding cities and toxic contaminants from heavy industries. The harbour was designated in 1987 by the International Joint Commission as one of the Areas of Concern on the Great Lakes under the terms of the Great Lake Water Quality agreement (IJC 1988).

For 90 years, wastes from the steel industry and associated coking facilities contaminated the sediments with iron-manganese oxides, heavy metals such as cadmium, copper, and zinc; coal dust and numerous polyaromatic hydrocarbons (PAHs) and polychlorinated biphenyls (PCBs) (Poulton 1987, Mayer and Johnson 1994, Fox et al. 1996). In several areas of the harbour, concentrations in the sediments exceed Ontario's guidelines for dredged sediment disposal (OME 1985). Randle Reef, situated along the industrial south shore, and the Windermere basin are the areas with greatest contamination of coal tar-contaminated sediments, and resuspension of these materials is a major source of genotoxic PAH in the water column (Marvin et al. 2000).

A Remedial Action Plan (RAP) has been developed and several beneficial impairments were identified including high nutrient levels, contaminated sediments and impaired planktonic and benthic communities. Following financial expenditures of over 600 million dollars by industry and municipalities, remediation efforts including improvements to wastewater treatment have considerably reduced nutrient and contaminant loadings (Hamilton Harbour RAP 1992). Improvements to water quality, habitat recovery and the establishment of waterfront parks have increased both wildlife and public use of the harbour. However, phosphate levels are still above initial RAP target goals of 0.034 mg l^{-1} (Charlton and Le Sage 1996), and total nitrogen and ammonia levels still often exceed provincial guidelines (Barica 1990, OMEE 1994). Concentrations of unionized ammonia approach the provincial guideline of 0.02 mg l^{-1} during winter. Water transparency measured as Secchi depth averaged 2.1 m during 1994, still above the target Secchi depth goal of 3 m (Hamilton Harbour RAP 1992). Continuing progress on load reductions from municipal sewage plants is ongoing and will take further infrastructure funding and time before final target goals can be reached.

In addition to the improving conditions in the harbour and increased wildlife habitat areas, several new introduced species have become established in the harbour. These include the zebra mussels (*Dreissena polymorpha* and recently *D. bugensis*), round goby (*Neogobius melastoma*), and the predatory zooplankton (*Cercopagis pengoi*). These new species are now living together with the native species and with long established introduced species including the common carp (*Cyprinus carpio*) and faucet snails (*Bithynia tentaculat*a). The present biological communities have had to adapt to both the chemical and biological pollution in the harbour.

Basin Description

Hamilton Harbour, sometimes called Burlington Bay, is separated from Lake Ontario by a natural sand bar (Fig. 1). The bay, which covers an area of 21 km^2, is bowl shaped with a maximum depth of 24 m and mean depth of 13 m, yet 12 % of the area is less than 2 m deep (Table 1). Re-development of both industrial docks and parklands has reduced the area from 28 km^2 and a maximum depth of 28 m since 1964, when 15 % of the area was less than 2 m deep (Barica 1990, Johnson and Matheson 1968). The watershed surrounding the bay is small, tributaries supply about 1.6 $m^3\ s^{-1}$ of the inflow. During summer, wastewater can represent 67 % of the inflow to the bay (4.9 $m^3\ s^{-1}$), with wastewater from the city of Hamilton responsible for 75 % of the Sewage Treatment Plant (STP) inflows (Hamblin and He 2003). Industrial users recycle up to 27 $m^3\ s^{-1}$ mainly for cooling purposes. A shipping canal of 9.5 m depth connects the harbour to Lake Ontario. The west end of Lake Ontario is subject to cold upwellings and flow through the canal can be multi-layered and bi-directional depending on lake level, wind direction and internal seiches (Poulton et al. 1988, Hamblin and He 2003). Inflow from Lake Ontario through the canal can exceed 11.4 $m^3\ s^{-1}$ (Hamblin and He 2003).

The shape and depth of the harbour results in summer stratification leading to oxygen depletion in the hypolimnion for much of the summer. The oxygen demand of the sediment surface can exceed 0.65 mg $O_2\ l^{-1}\ day^{-1}$. A demonstration oxygenation experiment required an aeration rate of 8.5 m^{-3} (300 cubic feet) compressed air per minute to prevent deoxygenation of the hypolimnion in summer (Piccinin 1977). Sediment anoxia is responsible for increasing metal bioavailability in fall-collected sediments (Krantzberg 1995). Cold oxygenated water from Lake Ontario often enters the hypolimnion or metalimnion via the canal, resulting in complex temperature and oxygen profiles in the harbour. Internal seiches within the harbour can also force the low oxygen water up to above the 8 m depth zone (Matheson 1958).

The north shore is mostly residential, whereas the south shore is mostly industrial which accounts for 46% of the total shoreline (BARC 2006). A secondary sandbar at the west end of the bay, remnant of the higher Lake Iroquois stage of 12,000 years ago separates the harbour from Cootes Paradise (Rukavina and Versteeg 1996). The northwest end of the harbour and Cootes Paradise are part of undeveloped Royal Botanical Gardens lands which supports a variety of wildlife habitats, which are used by a number of rare species including Blanding's turtles (*Emydoidea blandingii*) and Prothonotary warblers (*Protonotaria citrea*).

OBJECTIVES

Less is known about the biological condition of the harbour than is known about the nutrient and contaminant levels. Fisheries and Oceans has undertaken an ecosystem approach and developed a comprehensive program to examine all components of the food-web, from microbes up to fish, as well as information on habitat and aquatic plants in the harbour. This is the first time such a comprehensive study has been done in the harbour. The aim of this work was to add to the knowledge of the biology of Hamilton Harbour over all lower trophic levels from microbial to non-vertebrate predators. This included sampling both for inter-annual changes, and spatial variability in microbial, phytoplankton, zooplankton, and benthic organisms, their species composition and biomasses as related to the depth, temperature and oxygen layers in the water column.

Bi-weekly sampling conducted between May and October in 2002, 2003 and 2004 examined environmental and several biological components simultaneously at the same sites. The site identifications used follow a number of previously established sampling sites including: water quality and sediment sites (200 series) of the Ontario Ministry of Environment (Poulton 1987); Environment Canada's water quality sites (900 series) used by Murray Charlton (Charlton et al. 1992, Halfon 1996); locations used by Fisheries and Oceans electrofishing transects (Brousseau et al. 2005), as well some of the benthic sampling locations established by Johnson and Matheson (1968). Locations and depths of the biweekly sample sites are listed in Table 2.

The location and number of sites changed each year. In 2002, three sites were sampled along a nearshore to offshore gradient, the two nearshore sites 17 (1.5 m depth) and 6 (6 m) were close to La Salle Park, and the offshore site 258 was at mid-harbour at 23.5 m depth. During 2003, sampling was done in the west-end of the harbour; the nearshore site at Willow Cove in 3.8 m and the other site situated halfway between Willow Point and the south shore at a depth of 14 m (site 908). In 2004, sampling was repeated at the two offshore sites, site 908 and the mid-harbour site 258 (Fig 2).

On each sampling date, all the physical variables and biological components except bottom fauna were sampled. Depth profiles of temperature, oxygen, and light profiles were taken. Samples were collected for water chemistry, chlorophyll *a*, seston, total and size fractionated primary production, bacteria, ciliates, phytoplankton, and zooplankton. Bottom samples were collected on 3 seasons only in 2002 and 2003.

A more extensive spatial survey was also conducted during the summer of 2002 at a number of sites between the canal and Willow Cove. Only some of the biological components were sampled in the spatial survey, including: microbial loop including bacteria, ciliates, and size fractionated algae; composition and biomass of the phytoplankton; zooplankton; and bottom fauna.

This research was supported under the Great Lakes Action Plan. The work falls under DFO's priority of an ecosystem approach to management of human activities in the harbour, using several rate processes as ecosystem indicators of human perturbation. DFO priority research on ecosystem modelling and linkages to habitat productive capacity are also addressed. The data produced will eventually contribute toward an ecosystem model (ECOPATH) of the harbour's food web to assess sustainability of the fisheries targets for the harbour. The data will also be used to evaluate the Beneficial Use Assessments for the plankton and benthic communities under the Remedial Acrion Plan for Hamilton Harbour.

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Table 1. Hamilton Harbour hypsometric data from GIS polygons of the depth contours (from C. Bakelaar, DFO).

Depth range (m)	Area (km^2)	Total area (m^2)
0 - 2	2.49	11.9
2 - 5	1.24	5.9
5 - 10	4.27	20.3
10 - 15	5.11	24.4
15 - 20	5.67	27.0
20 +	2.20	10.5
Sum	**20.97**	

Table 2. Depth and location of sites used for the intensive bi-weekly sampling in Hamilton Harbour during the years 2002, 2003 and 2004.

Site	Depth (m)	Latitude	Longitude
2002			
17	1.5	43° 18.201'	079° 50.354'
6	5.7	43° 18.133'	079° 50.300'
258	23.8	43° 17.241'	079° 50.446'
2003			
WC	3.2	43° 17.183'	079° 52.268'
908	14.8	43° 16.768'	079° 52.443'
2004			
908	14.8	43° 16.768'	079° 52.443'
258	23.5	43° 17.241'	079° 50.446'

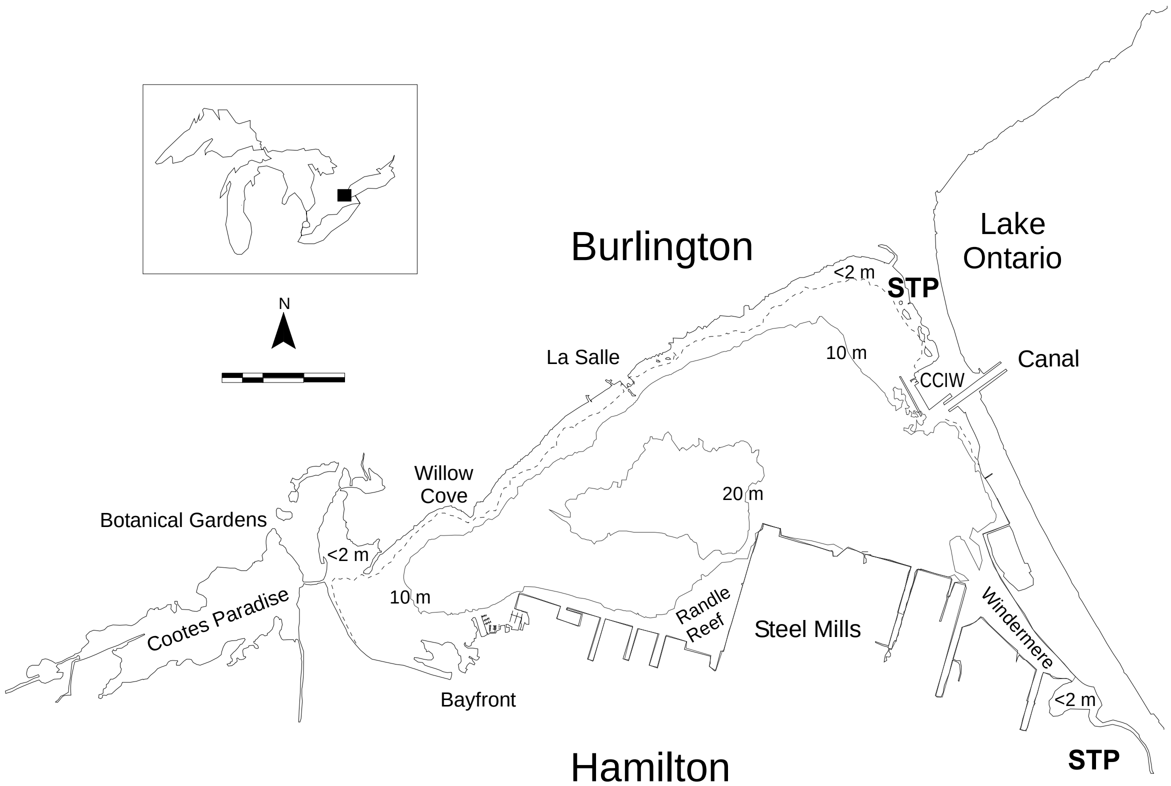

Figure 1. Hamilton Harbour with depth contours and surrounding features, scale bar = 1800 m.

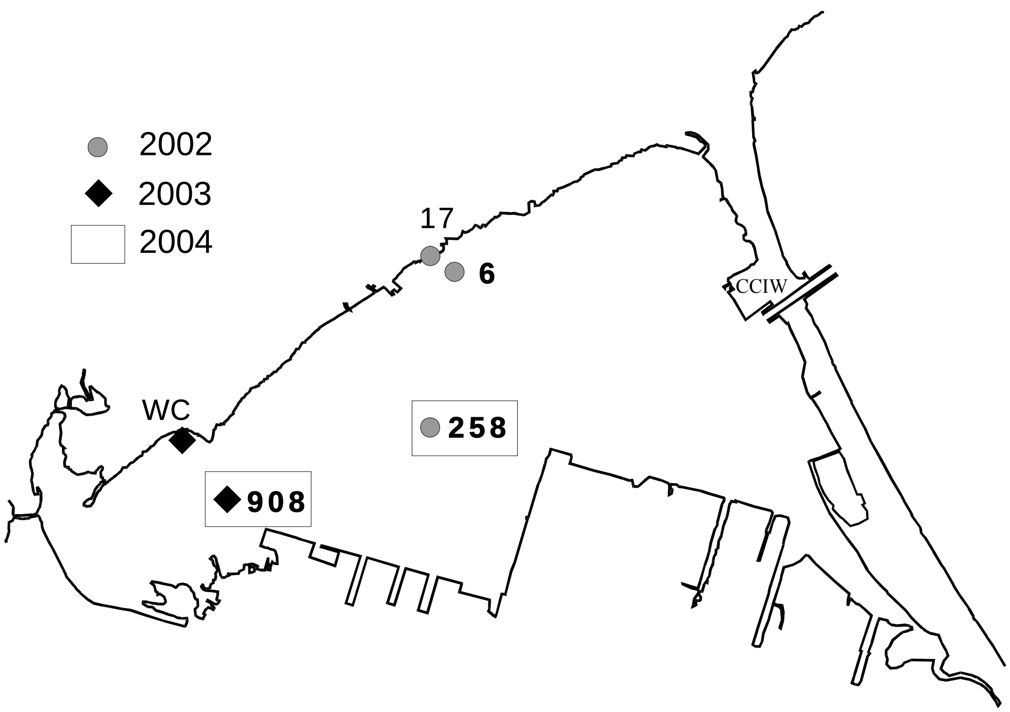

Figure 2. Locations of sites used for intensive biweekly sampling in 2002, 2003 and 2004.

WATER QUALITY AND PHYTOPLANKTON PHOTOSYNTHESIS

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INTRODUCTION

Hamilton Harbour is an embayment with a total area of 2150 hectares, connected to Lake Ontario by the Burlington Ship Canal across the sandbar that forms the harbour's eastern edge. Significant and highly variable exchanges of water occur between Hamilton Harbour and Lake Ontario via the canal. A 250 hectare area of marsh and shallow open water (Cootes Paradise) discharges into the Harbour's western end via the Desjardins Canal. The Harbour's watershed covers an area of 49,400 hectares with three major tributaries flowing into the Harbour. Spencer Creek drains the north-west and west portion of the watershed and feeds into the Harbour via Cootes Paradise; it accounts for the largest tributary inflow of 54%. Redhill Creek (15%) drains the south-east area of the basin and Grindstone Creek (14%) drains the north central portion (Hamilton Harbour Remedial Action Plan (RAP) 1992).

Water quality issues plaguing Hamilton Harbour stem from its long history as an industrial port and receiver of industrial and municipal wastes and watershed runoff. In the 1850s the harbour was deemed unfit as the drinking water supply due to raw sewage contamination from the City of Hamilton (RAP 1992). Starting in the early 1900s steel and iron industry wastes contaminated the sediments with heavy metals, coal tar containing polyaromatic hydrocarbons (PAHs) and later polychlorinated biphenyls (PCBs) (Poulton 1987). Sewage treatment has greatly improved over the years, but still results in excessive eutrophication from nutrient loading of phosphorus, ammonia and suspended solids, as well as a host of other contaminants. Eutrophication symptoms include offensive algal growths, poor water clarity and depleted oxygen. Hypolimnetic oxygen concentrations in the summer reach critical lows (0.5-1 mg l^{-1}) due to the oxidation of ammonia to nitrate by nitrifying bacteria which rapidly depletes the water column of dissolved oxygen. In summer, about half of all the water entering the harbour is waste water effluent (Charlton and Milne 2005). Combined sewer overflow (CSO) effluent from the City of Hamilton and runoff discharged from the eastern part of the harbour contribute to the harbour's loading of nutrients, suspended solids and other contaminants (e.g. lead, zinc, PAHs). The inflow of Lake Ontario water from the Burlington Ship Canal is also a contributor of persistent contaminants to

Hamilton Harbour, but also provides essential oxygenated water to the hypolimnion during the summer months (Hamilton Harbour Remedial Action Plan (RAP) 2003).

The south and east shores of the harbour have been infilled for industrial and marine activities as well as for railway or highway construction. Together with other developments, 75% of the wetlands and fish nursery habitat have been eliminated from the Harbour (RAP 1992). Nearly 3.6 million m^3 of material had been removed from the south shore between 1951-1962 (Whillans 1979), illustrating the magnitude of wetland removed from this portion of the harbour. Sustained high water levels, poor water clarity and plant disturbance by carp have contributed to the disappearance of the formerly extensive marshes in Cootes Paradise and Grindstone Creek (RAP 1992). The wetland at the mouth of Redhill Creek is only a remnant of a considerable marsh in this area (Holmes and Whillans 1984).

The International Joint Commission in 1987 designated Hamilton Harbour an Area of Concern with the aim to restore and protect beneficial uses (IJC 1988). The Remedial Action Plan (RAP) for the harbour set a number of targets for water clarity, phosphorus, ammonia and chlorophyll concentration (RAP 1992). High nutrient concentrations, algal blooms, suspended solids, reduced water transparency and low dissolved oxygen are impairments to the beneficial uses of the harbour. This includes degraded fish and wildlife (iii), degradation of benthos (vi), eutrophication (viii), degradation of aesthetics (xi), added cost to agriculture or industry (xii) and degradation of phytoplankton and zooplankton (xiii). In the past two decades, industry and regional municipalities have spent an estimated $600 million to improve the harbour's water quality (RAP 1992).

A large reduction in nutrient load from recent wastewater upgrades has improved water clarity and chlorophyll levels. The concentration of metals has met provincial guidelines for some time (Charlton and Milne 2005). The Hamilton RAP recommends that nutrient reduction be sufficient to result in final concentrations of 0.017 mg l^{-1} for total phosphorus, <0.02 mg l^{-1} for un-ionized ammonia and 5-10 μg l^{-1} for chlorophyll; no final target has been set for suspended solids for the harbour. The final target for water clarity (Secchi disc depth) is 3 m and the final target for minimum dissolved oxygen has been set to > 4 mg l^{-1} (RAP 1992). This chapter reports on the physical-chemical limnology and present status of phytoplankton productivity based on research conducted in 2002 through 2004. For the purpose of assessing impairment, comparisons are made with the Bay of Quinte, another Area of Concern which also has eutrophication issues.

MATERIALS AND METHODS

The 2002-2004 field sampling was conducted by scientific personnel from Fisheries and Oceans Canada (Burlington) using Boston whalers docked at the Canada Centre for Inland Waters (CCIW). In all years, thirteen cruises were conducted on alternate weeks starting in early May and ending in late October (Table 1). In 2002, stations 17, 6 and 258 were sampled. Stations WC and 908 were sampled in 2003 followed by stations 908 and 258 in 2004 (Table 1).

All laboratory work including photosynthesis experiments, chlorophyll filtration and nutrient processing were conducted at CCIW. The following measurements and collections occurred at each station for all dates. Depth profiles of temperature, dissolved oxygen, pH and specific conductivity were obtained using a Hydrolab minisonde 4a (Hach Environmental, Loveland, Colorado). Stratification status and mixing depth were determined utilizing specialized density gradient analysis software (DENS). Secchi depth was recorded to determine water transparency. Light extinction coefficients (400-700 nm - εpar) were determined using a Li-Cor underwater quantum sensor (Li-Cor, Lincoln, Nebraska). Composite water samples were collected for phytoplankton production, nutrients, seston and chlorophyll *a*. A Van Dorn sampler was used to collect water at evenly spaced depths within the water column. At stratified stations, composite water sampling was limited to the epilimnion. For unstratified or shallow stations whole water column samples were taken. For a detailed description of the methods see MacDougall et al. 2001.

Phytoplankton production was determined using a ^{14}C-uptake technique (Fee 1990). The photosynthetic parameters derived were used as input for the FEE program that computes depth and time-integrated photosynthetic rates. Seasonal (May 1-Oct 31) areal (g C m^{-2}) and volumetric (g C·m^{-3}) rates of phytoplankton photosynthesis were calculated.

Composite water samples were processed for water chemistry following the National Laboratory for Environmental Testing (NLET) guidelines (Environment Canada 1995). This lab analyzed all nutrient samples for the study. Chlorophyll *a* was processed as per MacDougall et al. (2001) following methodology of Strickland and Parsons (1972). Analysis for seston parameters was determined by filtering aliquots of sample water on pre weighed GF/C filters (Whatman Co.). Total suspended solids were determined by weighing samples after drying at 60° C. Ash content was determined after processing in a muffle furnace at 450° C. Two filters were processed per station and averaged to determine final weights.

Eutrophication issues also occur in the elongated Bay of Quinte, another Area of Concern located in the north eastern section of Lake Ontario. The same sample collection and analysis techniques were conducted in both Hamilton Harbour and the Bay of Quinte. As a result, data for both locations are directly comparable. The upper Bay of Quinte station is the most heavily eutrophied and shallow, averaging 5 m deep. The mid bay station HB (Fig. 1) has an average depth of 12 m and stratifies; thus it is similar to the Hamilton station 908 which has an average depth of 14.6 m. Conway, in the Bay of Quinte, is a deep station with average bottom depth of 31 m, which is deeper than the mid harbour station 258 (mean depth 23.4 m; Table 2).

RESULTS

Hydrolab Profiles

Stations 258 (sampled in 2002, 2004) and 908 (sampled in 2003, 2004) were the only harbour stations in the study deep enough for persistent stratification to occur. The mixed (epilimnetic) depths at stations 258 and 908 are shown in Fig. 2. For the purpose of analysis, sampling events were divided seasonally according to stratification status and mixing depth temperatures. The first three cruises from early May through mid June (Table 1) were designated as spring cruises because they were unstratified with a mean mixing depth (epilimnetic) temperature ranging from 10.3 to 17.0 °C. Cruises from late June through September (cruises 4-11) were designated as summer cruises because they were stratified with mean mixing depth temperatures ranging from 16.5 – 24.1 °C. Fall cruises encompassed the last two sampling dates in mid October until early November (cruises 12 and 13) and were unstratified with mean mixing depth temperatures ranging from 10.2 – 15.7 °C.

The mean summer epilimnetic depth at station 258 was 6.3 m in 2002 and 8.0 m in 2004; at station 908 the mean summer epilimnetic depth in 2002 was 5.2 m compared to 8.2 m in 2004. Summer epilimnetic temperatures at station 258 in 2002 averaged 21.9 °C and 21.5 °C in 2004; at station 908 mean summer temperature in 2003 was 19.8 °C and in 2004 was 20.3 °C. At the offshore stations mean spring temperatures ranged from 12.2 to 14.6 °C and fall temperatures ranged from 12.6 to 14.2 °C for all study years (Fig. 3). Epilimnetic temperatures were similar to that at Hay Bay (HB; Fig. 3).

The Hamilton Harbour Remedial Action Plan (RAP) has set an initial goal for minimum oxygen concentration in the harbour to be greater than 1 mg l^{-1} and a final goal of greater than 4 mg l^{-1} oxygen. Oxygen profiles were analyzed for occurrence of concentrations below these targets. Oxygen depletion occurred consistently at station 258 and 908 (Fig. 4) during the stratified period from late June until September. At station 258 dissolved oxygen values at or below 4 and 1 mg l^{-1} occurred at 69% and 54% of the cruises respectively for both study years. At station 908, the average occurrence of dissolved oxygen at or below 4 mg l^{-1} was 69% and 61.5% of the cruises, and 38% and 31% for the 1 mg l^{-1} concentration in 2003 and 2004 respectively.

At station 258 in 2002, the spring and fall whole water column oxygen levels averaged 9.9 mg l^{-1} and 6.2 mg l^{-1} respectively. During stratification, mean hypolimnetic oxygen ranged from a high of 3.8 mg l^{-1} in late June to four occurrences of mean hypolimnetic oxygen below 1.0 mg l^{-1} from August to early October. At station 258 in 2002 the overall mean hypolimnetic oxygen concentration was 1.3 mg l^{-1}. Similar dissolved oxygen trends were observed at station 258 in 2004.

Dissolved oxygen and temperature at station 258 were plotted against depth for sampling dates in 2002 (Fig. 5), to illustrate the irregularity of the oxygen pattern that can occur at this station. Surface oxygen levels were high but could decrease throughout the epilimnion. Oxygen concentrations plunge in the metalimnion and hypolimnion, but upward surges in oxygen concentration could occur in the hypolimnion. At station 908 the oxygen concentrations were typically acceptable in the epilimnion but plunged throughout the metalimnion and hypolimnion. Dissolved oxygen concentrations and temperatures are plotted against depth for station 908 for sampling dates in 2004 (Fig. 6). At station 908, the mean whole-water column dissolved oxygen levels were range from 9.0 to 9.1 mg l^{-1} in spring and 7.7 to 8.2 mg l^{-1} in the fall (2003, 2004). Epilimnion dissolved oxygen concentrations for both years (summer) ranged from 6.0 to 6.4 mg l^{-1}. Hypolimnetic mean dissolved oxygen ranged from 0.25 to 2.9 mg l^{-1} with an overall stratified hypolimentic mean of 1.6 mg l^{-1} in 2004. For both study years (2003, 2004) hypolimnetic oxygen concentrations at 908 were as high as 5.6 and as low as 0.17 mg l^{-1}. Oxygen depletion in the harbour was typically associated with stratification but there were incidents of low oxygen during unstratified conditions mainly in the early fall (Fig. 5).

There were also occurrences of oxygen depletion at the nearshore stations. Station 6 had four occurrences where oxygen dipped below 4 mg l^{-1}. These were during summer between early July until early August. There was only one cruise (July 8, 2002) where oxygen decreased below 1 mg l^{-1} and this was at the station bottom (6.7 m). Oxygen depletion also occurred at the nearshore Willow Cove (WC) station with one occurrence of oxygen below 4 mg l^{-1} on July 8, 2003, when a low of 1 mg l^{-1} was also measured at the station bottom (3.2 m). Unlike Hamilton Harbour, the Bay of Quinte does not experience significant depleted oxygen at any of the sampled stations.

Nutrients

Seasonal data is defined as that data for the spring, summer or fall following the conventions previously outlined in regards to stratification. Whole season data represents the mean of all data collected over the year of sampling (i.e. May through October). Whole season water chemistry results are summarized in Table 3. Nutrient levels exhibited a high degree of variation both spatially and temporally in the harbour (Figures 7-10). With the exception of station 258 during spring 2002, the seasonal trend for the offshore stations was high total phosphorus levels in the spring followed by stepwise decreases in the summer and fall (Fig. 10). In 2002 seasonal means of total phosphorus peaked in the summer at all stations sampled. Comparing all stations, total phosphorus levels were highest in the spring at station 908 in 2003 (42.6 ug l^{-1}) and lowest in the fall at station 258 (2004= 21.8 ug l^{-1}; Fig. 10). A gradient from nearshore to offshore was also observed with the annual mean total phosphorus at the nearshore stations ranging from 35.8 - 37.9 ug l^{-1} compared to 25.5-30.8 ug l^{-1} for those offshore (Table 3).

The Hamilton Harbour RAP has set an initial harbour goal for total phosphorus concentration to be 34 ug l^{-1}; the final goal is 17 ug l^{-1}. Total phosphorus values were below the initial target on all but one occasion at station 258 in 2004 (May 26; Fig. 7); in 2004 there were also three cruises where the total phosphorus values were below the final target listing. On a seasonal basis the initial goal was met 57% of the time over the course of the study, mainly at the offshore stations (Fig. 10). The final goal of 17 ug l^{-1} for the seasonal means was not met at any station. Total phosphorus levels less than 17 ug l^{-1} were measured on seven dates during the study at the offshore stations. In comparison, mean whole season total phosphorus in the offshore of Hamilton Harbour during 2004 (25.5, 28.5 ug l^{-1}) are comparable to the means observed in the upper and mid Bay of Quinte stations B (26.5 ug l^{-1}) and HB (24.4 ug l^{-1}) over the same time period.

The whole season mean total phosphorus versus chlorophyll ratios for the harbour was plotted in relation to the Bay of Quinte ratios for 2002-2004 (Fig. 11). As the Bay of Quinte data were collected with the same methodology as for Hamilton Harbour, direct comparisons between these two compromised ecosystems is valid. Nearshore station 17 exhibited the highest phosphorus concentration and resulting chlorophyll density. The harbour's offshore areas cluster mainly with the Belleville (B) and Hay Bay (HB) stations in the Bay of Quinte. The deep station in the Bay of Quinte (Conway (C) – 23.0 m) had the lowest total phosphorus versus chlorophyll ratio (Fig. 11)

Concentrations of soluble reactive phosphorus (SRP, orthophosphate) exhibited a high degree of spatial and temporal variation throughout the year (Fig. 7), reaching a maximum of 6 ug l^{-1} at station 258 (2002) in mid September. At all stations the whole season mean SRP ranged from 1.0 to 2.4 ug l^{-1} (Table 3a). The maximum seasonal mean of SRP occurred at station 258 (2002) in the spring (2.9 ug l^{-1}). Spring and fall peaks were observed at stations 258 (2004), 908 (2004) and 17 (2002).

Ammonia concentrations reported in this study are dissolved values and not total. Ammonia levels peaked in the spring and began to drop in late June (Fig. 8). Concentrations declined considerably by late July and reached their lowest values in mid-August to early September. The highest concentration observed was at Willow Cove in mid May (2003 = 1150 ug l^{-1}). Average ammonia levels in the spring were greater than 600 ug l^{-1} for all stations (Fig. 10). At all stations there was a considerable decrease from the spring to summer ammonia levels. Summer ammonia levels were up to eight times lower than those occurring in the spring (stations 258, 908, WC). Both the highest seasonal mean (spring 2003 = 1060 ug l^{-1}) and lowest (summer 2004 = 94 ug l^{-1}) occurred at station 908. The Bay of Quinte ammonia levels ranged from 12 to 14 ug l^{-1}, drastically less than the harbour values. Ammonia concentrations were calculated to un-ionized ammonia using pH and temperature conversion factors from Emerson et al. (1975). This involved calculation of seasonal means of pH and whole water column temperatures for each station for all years. Conversion factors were determined from these means and the data converted accordingly (Fig. 10). The Hamilton Harbour RAP has set a target for the un-ionized ammonia concentration to be less than 20 ug l^{-1}.

The un-ionized ammonia values are derived from dissolved values and not total. As a result, the occurrence of exceedance of this contaminant above target levels is likely higher than reported in this study. Seasonally, un-ionized ammonia concentrations exceeded the target concentration at four stations in the spring. These were in 2003 (908 = 22.3 and WC = 20.4 ug l^{-1}) and in 2004 (258 = 60.9 ug l^{-1} and 908 =57.5 ug l^{-1}). Un-ionized ammonia levels in summer ranged from 1.2 to 2.4 ug l^{-1} and in the fall 0.8 to 1.7 ug l^{-1}. As expected, seasonal means of un-ionized ammonia followed the same pattern as measured ammonia with high levels in spring followed by a swift drop in the summer and fall.

Nitrate-nitrite levels (dissolved) were high throughout the harbour and at all seasons (Fig. 10). Values reported indicate both nitrate and nitrite but generally the nitrate component represents the bulk of the reported value as nitrite is rare in aquatic systems (Keeney 1972). However Barica (1990) reported that nitrite levels in the harbour were a concern and therefore their portion in the nitrate-nitrite results cannot be discounted. Levels over 2900 ug l^{-1} were observed at station WC (2003) in June and again in early July. The lowest observed value occurred at station 17 (2002) at 854 ug l^{-1} (Fig. 8). With the exception of station 908 in 2004, the nitrate-nitrite levels for spring versus summer were similar (spring mean range 1863–2343 ug l^{-1}; summer mean range 1646-2243 ug l^{-1}. For all stations nitrate-nitrite levels dropped considerably in the fall (mean range 1390–1776 ug l^{-1}). Highest seasonal mean of nitrate-nitrite was in the nearshore (WC-2003 spring; 2343 ug l^{-1}) and the lowest was at station 258 in the fall (2002-1390 ug l^{-1}). The highest mean nitrate-nitrite concentration for 2002 - 2004 in the Bay of Quinte occurred at Conway (250 ug l^{-1}); this is considerably less than the mean concentrations observed in Hamilton Harbour for the same timeframe (1673-2147 ug l^{-1}).

As with other nutrients, silica concentrations were variable between stations and sampling periods (Fig. 9) ranging from 0.06 to 1.6 mg l^{-1}. Whole season silica means for the nearshore stations (range 0.52-0.80 mg l^{-1}) were comparable to those offshore (range 0.55-0.82 mg l^{-1}; Table 3). Highest silica concentration (1.6 mg l^{-1}) occurred at station 258 (2002) in mid October. On a seasonal basis, lowest mean silica concentrations were observed in the spring (Fig. 10). The lowest spring values occurred at all stations in 2002 (6 = 0.08, 17 = 0.12, 258 = 0.12 mg l^{-1}). The only time silica concentrations exceeded 1.0 mg l^{-1} was in the fall. This fall upsurge of silica occurred at all stations but remained below 1.0 mg l^{-1} at station 908 in 2003 (0.89 mg l^{-1}). In the Bay of Quinte, station B had higher whole season (2002 through 2004) silica concentrations ranging from 2.7 to 3.3 mg l^{-1}. Conway, the deepest and least enriched Bay of Quinte station had annual silica concentrations ranging from 1.2 to 1.3 mg l^{-1} for the same timeframe. The spring mean at Conway for 2002-2004 was 1.9 mg l^{-1}.

Mean annual total suspended solids (TSS) ranged from 3.4 to 4.7 mg l^{-1} at the offshore stations and 4.4 to 8.0 at nearshore stations (Table 2). Comparing seasonal means, the highest level of 9.8 mg l^{-1} occurred in the summer at station 17 (2002) and a low of 2.7 mg l^{-1} was observed at stations 258 (2004) and WC (2003) in the fall (Fig. 12).

Chlorophyll *a*

The highest whole season mean chlorophyll *a* concentration in the study (22.9 ug l^{-1}) was observed at the shallowest station (17-depth 1.3 m; Table 2). At the offshore stations, lower whole season concentrations were observed in 2004 (258 = 11.4 ug l^{-1}, 908 = 11.7 ug l^{-1}) than in 2002 (15.7 ug l^{-1}) or in 2003 (16.4 ug l^{-1}). These latter means are comparable to the whole season means observed at stations 6 (16.4 ug l^{-1}) or WC (13.4 ug $l^{-1)}$. For the most part chlorophyll *a* concentrations peaked in the summer. Station 17 had the highest seasonal means for the study in the summer (25.0 ug l^{-1}) and fall (28.2 ug l^{-1}) (Fig. 13).

The RAP has set an initial goal of 15-20 ug l^{-1} chlorophyll *a* concentration and a final goal of 5-10 ug l^{-1}. At the central station 258, chlorophyll *a* concentrations were below 20 ug l^{-1} at all sampling events in 2004 and all but three dates in 2002. Station 908 had lower chlorophyll *a* concentrations in 2004 when all but two dates had levels below 20 ug l^{-1}. Stations 6 and 17 exceeded 20 ug l^{-1} chlorophyll *a* concentrations on four occasions and WC was in exceedance of 20 ug l^{-1} on two occasions (Fig. 9). For all stations, chlorophyll *a* concentrations were less than 20 ug l^{-1} at 77% of the time over the duration of the study. Chlorophyll *a* concentrations below the final target of 10 ug l^{-1} occurred 35% of the time during the study. The station which most consistently met this target was 258 in 2004 when 9 of the 13 samples were below 10 ug l^{-1}.

In the Bay of Quinte, whole season mean chlorophyll *a* concentrations from 2002-2004 for B, HB and C were 12.9, 12.0 and 3.2 ug l^{-1} respectively (Table 2). In 2003, HB had unusually high chlorophyll *a* levels, excluding 2003 the mean for this station was 9.6 ug l^{-1}, which was more typical for this station (Burley and Millard 2006). These levels reflect the gradient in nutrient enrichment which is high in the upper bay (Belleville) and decreases downstream to Conway. The whole season means for the offshore harbour stations in 2004 (258=11.4 ug l^{-1}, 908= 11.7 ug l^{-1}) and WC (2003= 13.4 ug l^{-1}) are comparable to the chlorophyll levels observed at B and HB for the same timeframe.

Light extinction

The Hamilton Harbour RAP set initial and final goals for Secchi disk transparency to be 2 and 3 m respectively. Station 17 in 2002 was excluded from the comparisons as its station depth was only 1.3 m; so the bottom was always visible at this station. On a whole season basis all but two stations met the initial goal criteria of 2.0 m; stations WC (2003) and 908 (2003) were close at 1.8 and 1.9 m respectively (Table 2). None of the station's annual mean Secchi depth met the final 3 m goal. Mid harbour station 258 had the highest whole season mean of 2.4 m in both 2002 and 2004; station WC (2003) had the lowest at 1.8 m (Table 2).

There were occurrences when Secchi depth met the final goal of 3 m. In 2003, Secchi depths were at or greater than the final RAP target once at both station 908 and WC (Fig. 14). There were three occurrences when Secchi depth was at or greater than 3 m at station 258 in both sampling years, and at station 908 in 2004.

Seasonally lowest transparencies occurred in the spring where all but one station (258-2002; 2.5 m) were below the 2.0 m target. Generally transparency increased stepwise from spring to summer to fall. The fall mean Secchi depths exceeded the 2.0 m target at all of the stations. The lowest seasonal mean Secchi depth occurred at station 908 in spring 2003 at 1.5 m. This station also had the maximum Secchi depth mean for the study at 2.9 m in the fall of 2004 (Fig. 12).

The light extinction coefficients (ε_{par}) increase as transparency decreases. Lowest observed whole season ε_{par} mean (highest transparency) occurred at station 258 in 2004 (0.709 m^{-1}; Table 2). At the offshore stations higher ε_{par}s were associated with the spring. A drop in ε_{par} from summer to fall was observed at stations 258 in 2002 and 908 in 2003 and at all the nearshore stations (Fig. 12).

Both the light extinction coefficients and Secchi depth indicated that greatest water clarity on a whole season basis occurred at station 258. Station 908 and nearshore stations 6 (2002) and WC (2003) were similar to each other with ε_{par}s ranging from 0.76 to 0.77 m^{-1} (Table 2). Despite visually being able to identify the bottom at shallow station 17, it had the highest whole season ε_{par} mean of 1.06 m^{-1}. While the Secchi disk data indicates lowest water transparency occurred in the spring, the ε_{par} data indicates that the spring or summer season could have the lowest water transparency. Spring highs in ε_{par} (lowest water transparency) were observed at stations WC, 258 in 2004 and 908 in both study years which is congruent with Secchi depths which were lowest in spring at these stations (Fig. 12). The light extinction coefficient was most notable at 908 in 2003 where the spring ε_{par} (1.264 m^{-1}) was almost double the summer mean (0.656 m^{-1}). Summer highs in ε_{par} were observed nearshore at stations 6 and 17 (2002) and offshore at station 258 in 2002.

In the Bay of Quinte, comparative whole season means for Secchi depth for 2002, 2003 and 2004 at the B, HB and C were 1.9, 2.2 and 6.0 respectively. Whole season mean light extinction coefficients for B, HB and C for 2002 through 2004 were 0.957, 0.749 and 0.359 m^{-1}) respectively (Table 2, Fig. 12). This reveals that transparency increases from the uppermost shallow station Belleville to the deep station Conway near the mouth of the bay which exchanges water with Lake Ontario.

Phytoplankton Photosynthesis

Seasonal areal phytoplankton photosynthesis (SAPP) was calculated for all stations and years. The lowest SAPP observed in the study was at station 17 at 156 g C m^{-2}. Offshore stations 258 and 908 had lower SAPP values in 2004 (190, 215 g C m^{-2} respectively) compared to 2002 (258= 282 g C m^{-2}) and 2003 (908= 346 g C m^{-2}) (Table 4, Fig. 13). This latter SAPP value at 908 was the highest observed in the study. The SAPP percentage of cloudless for 2004 (258-65%, 908-66%) was low in comparison to the other years in the study (70-77%). In the Bay of Quinte, average SAPP for Quinte station B ranged from 189-289 g C m^{-2} and 136-359 g C m^{-2} at HB. Conway's SAPP values were much lower at 119-188 g C m^{-2} from 2002-2004 (Table 4). Total phosphorus concentrations versus SAPP values were analyzed comparing Hamilton Harbour and the Bay of Quinte (Fig. 11).

DISCUSSION

During the study period (2002 through 2004), the Harbour was stratified by late June and remained so until early to mid October. At station 258 the mixed depth became deeper in mid-September and conditions were isothermal by mid October. Station 908 is not as deep (14.6 m) as 258 (23.3 m) and was isothermal by mid September in both study years. The Ministry of Environment (1985) reported isothermal conditions persisting in the harbour from mid-October through May. There are complex and seasonally variable exchanges of harbour and Lake Ontario water via the Burlington Ship Canal. During the stratified period, cold well oxygenated Lake Ontario water can travel along the bottom of the harbour providing oxygen to the hypolimnion. These lake water intrusions are reported to spread up to two-thirds of the way to the Stelco property and up to 1.5 km laterally (MOE 1985). Despite this periodic influx, stratification results in oxygen depletion in the hypolimnion because epilimnetic oxygen remains trapped in the upper layer resulting in oxygen depletion in the metalimnion and hypolimnion.

Oxygen depletion in the harbour remains a persistent issue and is a serious barrier to long term ecosystem recovery. The main factor causing oxygen depletion is bacterial oxidation of the ammonia from sewage effluent to nitrate by nitrifying bacteria in the water column, which accounts for 35 to 45 percent of the oxygen demand (RAP 1992, Charlton and Milne 2005). Oxygen is further depleted through bacterial oxidation in the water column and sediments of reduced carbon, nitrogen and sulphur present in the effluents and oxygen demand from phytoplankton decay (RAP 2003). Decomposing algae may represent 30 to 35 percent of the oxygen demand during the summer (RAP 1992). Reaeration, photosynthesis and inflows from Lake Ontario all contribute to the oxygen present in the harbour (MOE 1985).

The Ministry of Environment (1985) reported that from 1976 to 1980 the harbour exhibited a rapid decrease in oxygen in May and June and hypolimnetic concentrations were close to zero. In the current study oxygen levels remained sufficient (9-10 mg l^{-1}) until stratification set up in mid to late June. In the 1976 to 1980 time period, hypolimnetic means during stratification ranged from 1 to 2 mg l^{-1} , which are comparable to the overall hypolimnetic means of 1.3 to 1.6 mg l^{-1} observed in the current study. Epilimnetic dissolved oxygen in 1976 to 1980 averaged between 6.0 to 8.0 mg l^{-1}; the current study had epilimnetic means lower than 6.0 mg l^{-1} at station 258 but they were still above the 4.0 mg l^{-1} RAP target. The study sites sampled were likely beyond the range of the main exchange of Lake Ontario water via the Burlington Canal (RAP 2003).

Water quality goals for the harbour were adopted by the Hamilton Harbour RAP committee in 1992 (Charlton and Milne 2005). The final RAP goal of a minimum of 4 mg l^{-1} dissolved oxygen concentration is the provincial water quality objective set by the Ministry of Environment for warm water biota living at 20 to 25 degrees Celsius. The objectives for cold water biota at zero degrees Celsius range from 4 to 8 mg l^{-1} dissolved oxygen. The initial RAP goal for dissolved oxygen is a minimum of 1 mg l^{-1} dissolved oxygen concentration. Below 1 mg l^{-1} conditions are hypoxic and unable to sustain most biota. The persistence of hypoxia at the stratified stations is a critical issue limiting biota from these areas. The periodic occurrence of depleted oxygen at the nearshore stations puts additional stress on the biota of the harbour. Currently the harbour fish populations are dominated by species that tolerate low oxygen such as carp, bullhead and gizzard shad. The return of viable cold water populations of lake trout, whitefish, herring and sturgeon is unlikely and remediation efforts are focusing on restoring warm water species such as pike, bass, perch, crappies, as well as rainbow and brown trout and Pacific salmon (RAP 1992).

Davis (1975) reported negative physiological and/or behavioural responses on the desired warm water species such as largemouth bass, rainbow trout and Pacific salmon at dissolved oxygen levels less than 4.5 mg l^{-1}. Seager et al. (2000) reported greater than 95 percent mortality of rainbow trout at 1.6 mg l^{-1} dissolved oxygen over a short term exposure and implicated duration of the exposure as a critical factor. The high occurrence of oxygen levels below 4 mg l^{-1} in the deep stations will hamper establishment of healthy population of these warm water fishes in the harbour. Dermott (*this volume*) indicates that the benthic invertebrate community living beyond nine metres is also restricted due to the oxygen depletion across the harbour bottom.

Nutrient loading to the harbour is a causative agent of the oxygen depletion and eutrophication occurring in the harbour. To promote the reduction of the nutrients, the RAP committee has set initial (34 ug l^{-1}) and final (17 ug l^{-1}) targets for total phosphorus concentration in the harbour. Phosphorus is considered to be a major factor controlling phytoplankton biomass and resulting trophic status (Dillon and Rigler 1974, Schindler, D.W. 1974). Charlton and Milne (2005) report total phosphorus concentrations peaking in summer which corresponds to the observed seasonal means at the nearshore stations and at station 258 in 2002. Spring peaks of total phosphorus were also observed at the offshore stations 258 (2004) and 908 (2003, 2004) in this study and this pattern has also been reported by Charlton and Milne (2006). There is variability in the nutrient dynamics of the harbour which is evident in the biweekly sampling values.

Phosphorus concentrations are high due to inputs from four sewage treatment plants, combined sewer overflows, agricultural and urban runoff and the steel industry. Loadings to the harbour have been reduced from 1,200 kg/day in 1967 to less than 10 kg/day in 1989 (RAP 1992). The MOE (1985) reported that in 1975 to 1983 annual means of total phosphorus ranged from 56 to 104 ug l^{-1}. During 2002 to 2004, the harbour whole season means for total phosphorus ranged from 25.5 to 37.9 ug l^{-1} which represents a two and a half fold decrease from the previous study. Values in the current study did not exceed 72.9 ug l^{-1} and Charlton and Milne (2005) reported annual means below 80 ug l^{-1} since 1990. Reducing loads to the harbour have been effective and in this study the initial target was met on a seasonal basis over half of the time at the stations studied. This fact combined with the occurrence of total phosphorus values below the final target on some dates over the course of the study is encouraging. Continued reductions of this nutrient are an important element in further limiting phytoplankton growth and increasing water transparency.

Seasonal means for total phosphorus versus chlorophyll were plotted to compare Hamilton Harbour data with the Bay of Quinte. Both areas are embayments of Lake Ontario requiring rigorous nutrient enrichment controls to abate eutrophication. The Bay of Quinte has undergone anthropogenic eutrophication beginning with European colonization; remediation efforts have been undertaken since the early 1970s. The linear regression of total phosphorus to chlorophyll indicates that for the timeframe of 2002 through 2004 there is no difference in response to total phosphorus inputs in terms of chlorophyll production between the two Areas of Concern. Comparing both systems the nearshore stations are the most nutrient enriched; station 17 in Hamilton Harbour exhibits the highest chlorophyll to total phosphorus ratio. The offshore areas of Hamilton Harbour closely resemble the enriched upper (B) and mid bay (HB) area of the Bay of Quinte and are far removed from the low chlorophyll to total phosphorus ratios observed in the deep station in the Bay of Quinte (station C).

Soluble reactive phosphorus (SRP, orthophosphate) is the inorganic form of phosphate and is a measure of biologically available phosphorus readily uptaken by bacteria and phytoplankton during photosynthesis. The current results concur with Harris et al. (1980) who reported peak concentrations of SRP associated with spring and fall overturn. Summer lows are likely associated with depletion of SRP under high algal densities. Charlton and Milne (2005) reported possible regeneration of SRP from the sediments triggered by extended periods of anoxic conditions.
The phosphorous loading to Hamilton Harbour has dramatically been reduced but continuing to reduce loading is required in order to further shift the system from its eutrophic state. From 1984 to

1986 SRP averaged 13 ug l^{-1}; improvements to the current levels in the harbour (1.0 - 2.4 ug l^{-1}) is a factor controlling algal abundance (RAP 2003).

The 1992 Remedial Action Plan (RAP 1992) stated that while industrial loadings of ammonia have decreased substantially in the last forty years there has been little change in the ammonia loadings from municipal sewage treatment plants. The latter contributes 80 % of the ammonia to the harbour. Annual ammonia concentrations of 1130 to 1950 ug l^{-1} during 1975 to 1983 were reported by MOE (1985). The current concentrations of 218 to 320 ug l^{-1} ammonia indicate a drastic reduction over the last twenty five years but these values are dissolved, not total ammonia. Compared with station B, the most enriched station in the Bay of Quinte study, ammonia levels are sixteen to twenty three times higher in Hamilton Harbour. Ammonia concentrations are highest in the spring because of a build-up of ammonia over winter. During these months sewage effluent enters the harbour but nitrifying bacteria are inactive in the cold temperatures (Fletcher 1979). As water temperature rises, oxidation of ammonia to nitrate by nitrifying bacteria increases to the point where hypolimnetic oxygen concentrations in the summer reach critical lows (0.5-1 mg l^{-1}) as nitrification rapidly depletes the dissolved oxygen from the bottom water layer.

Another issue with ammonia is its un-ionized form which is the most toxic form to aquatic life; the RAP committee has adopted the Provincial guideline of 20 ug l^{-1} as the target for un-ionized ammonia in the harbour. The percent of ammonia present in the un-ionized form is dependent on the pH and water temperature, and the subsequent conversion of the ammonia data indicated levels exceeded the target in the spring both at the offshore stations as well as at Willow Cove. The highest concentration un-ionized ammonia in the current study was 60.9 ug l^{-1} as a result of increased pH. The Harbour RAP also reported concentrations exceeding the target in the late spring and summer with a high over 120 ug l^{-1} reported in the 1987 and 1988 data (RAP 1992). Barica (1990) reported values in 1987 to 1988 that exceeded the targets for most of the spring and summer. The current study suggests a possible reduction in un-ionized ammonia concentrations, but the number of sampling sites was limited and results were based on dissolved ammonia, not total. Charlton and Milne (2005) reported fewer occurrences of un-ionized ammonia concentrations above the target in 2000 to 2005, due to greatly improved removal at the Burlington Skyway wastewater treatment plant. Further controls to reduce ammonia in the harbour are an essential component for improving the oxygen regime in the harbour.

In 1977, the Ministry of the Environment (MOE 1981) indicated the peak in seasonal nitrate coinciding with the time of maximum harbour BOD; maximum surface nitrate values (in July) averaged 2610 ug l^{-1}. This relates to the nitrification of ammonia in the spring which requires oxygen. A further review (MOE 1985) listed seasonal means for nitrates from 1975-1983 ranging from 1410 to 2180 ug l^{-1}; this is similar to the current concentrations from 2002 through 2004 which ranged from 1673 to 2147 ug l^{-1}. In the current study, peaks occurred in a similar timeframe generally in early July. The maximum of 2960 ug l^{-1} in early June indicates high levels of nitrates-nitrites still occur in the harbour despite reductions in ammonia loadings (RAP 1992). Seasonal means over 2000 ug l^{-1} were observed at both offshore stations (908-2003, 258-2004) and nearshore at station WC (2003). Generally nitrate-nitrite levels dropped by early September and lowest means occurred in the fall. This is likely related to denitrification of nitrates during the anoxic summer period and gassing off as atmospheric nitrogen (MOE 1981). Harris et al. (1980) found rapid nitrification of ammonia in June and July but the higher nitrate values they reported in the fall did not occur in this study. It does not appear there has been a meaningful reduction in nitrates-nitrites in Hamilton Harbour since 1980.

The emphasis is on reductions in un-ionized ammonia, and there is currently no target set for nitrates or nitrites. Harris et al. (1980) reported nitrite levels in 1976 and 1977 and Barica (1990) reported nitrite concentrations frequently exceeding acute toxic levels of 250 ug l^{-1} and chronic toxicity thresholds (30 ug l^{-1}) in 1987-1988. Scott and Crunkilton (2000) reported nitrate toxicity in fish fry and zooplankton but exposure levels far exceeding what was observed in the harbour. The lowest toxic concentration of nitrates reported was 6250 ug l^{-1} which resulted in sublethal effects on fry of lake trout and lake whitefish (McGurk et al. 2006). The nitrate-nitrite levels in the harbour are considerably higher than in the Bay of Quinte. Concurrent reductions of nitrates and nitrites associated with ammonia reductions would be positive for Hamilton Harbour to reduce potential toxicity and overall nutrient enrichment to the ecosystem.

The RAP report (1992) indicated about one third of the suspended solid loadings enter the harbour via Cootes Paradise; combined sewer overflows are the second greatest contributor at 19 percent. Industry and sewage treatment plants have reduced suspended solids loading in conjunction with pollutant reductions. The Ministry of the Environment (MOE 1985) found suspend solid concentrations in the harbour of 4.7 to 5.8 mg l^{-1} from 1975 to 1983 which are comparable to the current findings of 3.4 to 8.0 mg l^{-1}. With the exception of station 17, the whole season concentrations of suspended solid in Hamilton Harbour are similar to that in the upper (B) and mid bay (HB) station in the Bay of Quinte. Further reductions in suspended solids will be beneficial to water clarity and reduced oxygen demand in the harbour.

Silica can be a limiting nutrient for phytoplankton growth since diatoms require large quantities of silica for their cell walls (Goldman and Horne 1983). Levels were low at all stations in Hamilton Harbour over the duration of the study compared to the Bay of Quinte. Station B had whole season means four to six times higher than Hamilton Harbour for the same timeframe. The spring seasonal means at station C, the least enriched and deepest station in the study were sixteen times greater than spring means in Hamilton Harbour. The difference between the two Areas of Concern is unclear but the lows in the spring are related to depletion by spring diatom blooms and silica may be a limiting nutrient for diatoms in the harbour at this time of year.

Harris et al. (1980) reported silica in the Harbour ranging from 0.2 to 1.5 mg l^{-1} in 1975 which is comparable to the range in the current study (0.06 to 1.6 mg l^{-1}). Thus current silica levels in the harbour are comparable to mid 1970s values. Goldman and Horne (1983) indicate a release of silica under anoxic sediments. In the harbour silica may be released from sediments into the oxygen depleted hypolimnion during summer stratification and mixed into the whole water column after fall turnover. In combination with the low diatom levels observed in the fall, this may partially explain the high silica levels observed in the fall at all stations. Harris et al. (1980) reported mid and bottom water silica concentrations rising during the stratified period but found decreases in silica during the fall.

Water transparency measured as Secchi depth was lowest in the spring and typically highest in the fall. Transparency in the harbour has been mainly related to chlorophyll *a* levels, suspended silt and dissolved substances (RAP 2003, Harris et al. 1980). The spring transparency may be related to turbidity from suspended solids entering the harbour during spring runoff. Total suspended solids were highest in the spring at stations 908 and WC. These stations are located in the western portion of the harbour near Cootes Paradise and Grindstone Creek. This turbidity may reach as far as the central station 258 which exhibited high spring turbidity in 2004. The light extinction coefficient is a direct measure of photosynthetically available radiation and consequently is a more accurate tool to assess water transparency than Secchi depth. The light extinction coefficients revealed that lowest transparency occurred in summer at nearshore stations 6 and 17. Harris (1980) reported that chlorophyll *a* was a significant factor controlling water transparency in the water column. Chlorophyll *a* levels were high at stations 6 and 17 in the summer and are likely related to the seasonally low water transparency (high εpars) during this season. In 1986 the average Secchi disc transparency in the harbour was 1.4 m (RAP 2003); the current overall mean of 2.1 m indicates a 50 % increase in transparency in the past two decades.

The mean whole season Secchi depth for the harbour met or came very close to the initial RAP committee target for Secchi depth of 2.0 m. Charlton and Milne (2005) reported increased Secchi depth since the late 1990s and better than usual Secchi disc transparency readings in 2005 (Charlton and Milne 2006). With the exception of station 17, the εpar in the current study ranged from 0.71 to 0.77 which indicates a marked improvement in water clarity from a 1975 study by Harris et al. (1980) who reported extinction coefficients ranging from 0.8 to 1.6 m^{-1}.

Offshore station 258 had the highest water transparency in the study considering both εpar and Secchi depth, but its εpar was still two times greater than that observed in the deep station Conway in the Bay of Quinte. On a positive note, both stations 258 and 908 had occurrences where Secchi depth exceeded the final goal of 3.0 m. Comparing light extinction coefficients, nearshore station 17 is similar to the upper Bay of Quinte (station B) while the remaining nearshore stations and station 908 most closely resemble the middle Bay of Quinte (station HB). The offshore station 258 has slightly greater water transparency than the mid bay (HB) station.

Chlorophyll *a* concentrations are an indicator of phytoplankton density and were highest at the shallowest station 17. At the offshore stations in 2004, chlorophyll *a* levels were lower than in 2002 and 2003, likely related to the lower total phosphorus concentrations observed in 2004 Charlton and Milne (2006) also reported lower than usual chlorophyll *a* concentrations at station 258 in 2005. In the offshore, chlorophyll *a* concentrations peaked in summer; this also occurred at station 6 which was the deepest nearshore station at 7 m. Charlton and Milne (2005) reported high summer chlorophyll *a* in the harbour which loosely paralleled phosphorus concentrations; it is notable that spring highs in total phosphorus were also observed in the current study. The RAP targets for chlorophyll *a* concentration are 15-20 ug l^{-1} for the initial phase and 5-10 ug l^{-1} as the final goal. On a whole season basis, this initial target was met at all stations except 17 which was slightly higher at 22.9 ug l^{-1}. In 2004, central station 258 was compliant with the initial goal on every sampling occasion. Charlton and Milne (2005) reported a decrease in peak chlorophyll *a* concentrations since 2000 and a trend for more values are within the range of the final goal.

The offshore whole season chlorophyll *a* levels for 2004 were comparable to the upper bay station B (12.9 ug l^{-1}). The offshore whole season chlorophyll *a* concentrations for 2002 as well as nearshore stations 6 and WC were elevated compared to station B, ranging from 15.7 to 16.4 ug l^{-1}; station 17 has considerably higher chlorophyll *a* levels than B. Charlton and LeSage (1996) reported only a few occurrences of compliance to the final chlorophyll *a* target for transects along the north shore of the harbour in 1994. The highest seasonal mean observed in the current study was 28.2 ug l^{-1} which was a marked reduction in chlorophyll *a* since 1975 when Harris et al. (1980) found chlorophyll *a* fluctuated between 30 to 60 ug l^{-1} in the harbour. The current 77% and 35% compliance rate for all sampling occurrences to initial and final goal concentration targets respectively is encouraging.; but further reductions in phosphorus loading to the harbour would be beneficial in limiting algal growth.

Seasonal areal phytoplankton photosynthesis (SAPP) in the offshore harbour stations was approximately one and a half times higher in 2002 and 2003 than in 2004. This relates to higher total phosphorus and chlorophyll *a* levels observed in the earlier years. The low percentage of cloudless in 2004 (Table 4) was atypical and a result of the low incident solar irradiance that year. This same trend was noted in the Bay of Quinte for 2004 resulting in low SAPP values in both the Bay of Quinte and Hamilton Harbour in 2004 (Burley and Millard 2006). Atypically, low solar irradiance in 2004 was mainly responsible for the lower production values observed at both locations in that year. Station 17 had the lowest SAPP in the study despite having high chlorophyll *a* levels; depth truncation of the calculated rates explains in part the low value observed at this site; reduced water transparency is another cause for the low SAPP. The highest SAPP was observed in 2003 at station 908; although water transparency was lower at 908 in 2003 than 2004 it was not to the same degree as was observed at station 17. The nearshore stations 6 and WC had SAPP values comparable to the central station 258.

Both the Ministry of the Environment (MOE 1985) and Harris et al. (1980) reported lower than predicted phytoplankton production in Hamilton Harbour in the mid to late 1970s. This was attributed to light limitation due to low water transparency caused by high levels of suspended solids and dissolved organics in the harbour (MOE 1985, Harris et al. 1980, Piccinin 1976). Light extinction coefficients in the current study indicate improved water transparency to a range similar to that in the Bay of Quinte. The euphotic zone is the depth range in which solar irradiance penetrates to permits photosynthesis. It is generally considered to be from the surface to a depth with 1 % of the surface solar irradiance. The mean euphotic depth at station 258, 908 and Quinte station HB were 6.8 m, 6.7 m and 6.7 m respectively, indicating water clarity is now comparable between these two locations.

The Ministry of the Environment (MOE 1985) implicated variable mixing depths as another major factor limiting algal production. They reported that due to inputs from Lake Ontario, wind and other effects, the thermocline depth, and hence the mixing depth, is variable. Phytoplankton are often mixed below the euphotic zone dampening production. In the current study stratification appeared stable and the mixing depth did not alter radically over the course of the summer at either offshore station. The stability may be related to the location of the offshore stations, which were far enough from the canal to limit the influence of intruding Lake Ontario water. Mixing depth did exceed the euphotic zone depth during isothermal conditions. This occurred at station HB as well but mainly in the latter part of the field season. Toxicity of chemical contaminants to phytoplankton in the harbour has also been suggested as having an inhibitory effect on production (RAP 2003) but MOE (1985) reported this was secondary when compared with constraints due to physical factors discussed previously.

The range of SAPP observed in Hamilton Harbour are comparable to those observed in the Bay of Quinte. A linear regression of total phosphorus concentrations versus SAPP indicates that for the 2002 to 2004 time period, the ecosystem response of SAPP to total phosphorus inputs was comparable for the two study locations. Phytoplankton production in the harbour is not dampened in comparison to the Bay of Quinte. Increased water transparency in the harbour and thermal stability are contributing factors to the equivalency of phytoplankton production observed in these two Areas of Concern. This suggests that continuing to reduce phosphorus loading to the harbour is a key factor controlling algal growth, as water quality continues to improve through the remediation efforts.

Hamilton Harbour's long history as an industrial port and dumping site for industrial and municipal wastes makes it a complicated Area of Concern to successfully remediate and restore the beneficial uses. In addition to chemical contamination of its waters and sediment, the eutrophication of the harbour has compounded the water quality issues. Great strides have been made to improve water quality. Concentration of metals currently meet provincial guidelines and reductions in nutrient loads have improved water clarity and reduced phytoplankton biomass. A focus to further reduce nutrient loading, in particular ammonia and phosphorus, will improve the oxygen regime in the harbour. The current gains in water quality observed since the 1970s is testimony to the commitment and resolve of the Hamilton Harbour RAP and community.

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Table 1. Hamilton Harbour sampling dates for all stations and years sampled.

Cruise	**17** 2002	**6** 2002	**WC** 2003	**258** 2002	**258** 2004	**908** 2003	**908** 2004
1	15-May	15-May	14-May	15-May	13-May	14-May	13-May
2	28-May	28-May	27-May	28-May	26-May	27-May	26-May
3	13-June	13-June	10-June	12-June	8-June	10-June	8-June
4	26-June	26-June	24-June	25-June	24-June	24-June	24-June
5	8-July	8-July	8-July	9-July	6-July	8-July	6-July
6	24-July	24-July	22-July	23-July	19-July	22-July	19-July
7	7-Aug	7-Aug	6-Aug	6-Aug	3-Aug	6-Aug	3-Aug
8	21-Aug	21-Aug	20-Aug	20-Aug	17-Aug	20-Aug	17-Aug
9	4-Sep	3-Sep	3-Sep	4-Sep	31-Aug	3-Sep	31-Aug
10	18-Sep	18-Sep	16-Sep	17-Sep	14-Sep	16-Sep	14-Sep
11	1-Oct	1-Oct	29-Sep	1-Oct	27-Sep	29-Sep	27-Sep
12	16-Oct	16-Oct	14-Oct	15-Oct	15-Oct	14-Oct	15-Oct
13	6-Nov	6-Nov	27-Oct	6-Nov	27-Oct	27-Oct	27-Oct

Table 2. Comparison of whole season means for station depth, epilimnetic mixed depth, total suspended solids (TSS), light extinction ε_{par} (m^{-1}), Secchi depth disc (m), and chlorophyll ($\mu g\ l^{-1}$).

Station	Year	Station Depth	Epi Depth*	TSS	ε_{par}	Secchi	Chl
Nearshore							
17	2002	1.3	1.3	8.0	1.060	bottom	22.9
6	2002	6.7	6.7	4.5	0.759	2.0	16.4
WC	2003	3.4	3.4	4.4	0.772	1.8	13.4
Offshore							
258	2002	23.5	6.3	4.0	0.727	2.4	15.7
258	2004	23.2	8.0	3.4	0.709	2.4	11.4
908	2003	14.6	5.2	4.7	0.773	1.9	16.4
908	2004	14.6	8.2	3.9	0.758	2.3	11.7
Bay of Quinte							
B	2002-2004	5.2	5.2	4.41	0.957	1.9	12.9
HB	2002-2004	12.3	5.1, 5.4, 6.3	3.83	0.749	2.2	12.0
C	2002-2004	31.1	13.0	1.04	0.359	6.0	3.2

* for HH nearshore stations and B epi depth = mean bottom depth, for remaining stations epi depth = mean summer epilimnion depth

Table 3a. Whole season means for water chemistry including unfiltered phosphorus P-total (µg l^{-1}), dissolved phosphorus P-flt (µg l^{-1}), particulate phosphorus P-part (µg l^{-1}), soluble reactive phosphorus SRP (µg l^{-1}), dissolved total nitrogen N-flt (µg l^{-1}), dissolved ammonia NH_3 (µg l^{-1}) and dissolved nitrate/nitrite NO_3/NO_2 (µg l^{-1}). A dash indicates no samples processed for these parameters.

Station	Year	P-total	P-flt	P-part	SRP	N-flt	NH_3	NO_3/NO_2
Nearshore								
17	2002	36.8	14.8	-	1.7	-	279	1673
6	2002	37.9	13.1	-	1.8	-	301	1854
WC	2003	35.8	17.0	28.7	1.4	2649	320	2147
Offshore								
258	2002	30.8	13.2	27.3	2.4	2437	304	1751
258	2004	25.5	11.1	20.5	1.0	2881	310	2115
908	2003	35.5	13.6	31.6	1.5	2684	218	2063
908	2004	28.5	12.6	24.1	1.3	2752	283	1987
Bay of Quinte								
B	2002-04	26.5	11.4	22.9	2.1	437	14	56
HB	2002-04	24.4	11.9	20.3	2.3	413	14	74
C	2002-04	11.4	7.6	9.2	1.7	476	12	250

Table 3b. Whole season means for water chemistry including dissolved inorganic carbon DIC (mg l^{-1}), dissolved organic carbon DOC (mg l^{-1}), dissolved chloride CL (mg l^{-1}), dissolved silica SIO_2 (mg l^{-1}), dissolved sulphate SO_4 (mg l^{-1}), particulate organic carbon POC (mg l^{-1}) and particulate organic nitrogen PON (mg l^{-1}). A dash indicates no samples processed for these parameters.

Station	Year	DIC	DOC	CL	SIO_2	SO_4	POC	PON
Nearshore								
17	2002	25.0	5.6	77.7	0.60	49.4	-	-
6	2002	26.0	5.2	73.8	0.80	47.3	-	-
WC	2003	25.7	4.9	-	0.52	-	1.17	0.19
Offshore								
258	2002	26.0	5.0	73.4	0.75	48.0	1.18	0.18
258	2004	26.5	4.8	-	0.74	-	0.95	0.15
908	2003	26.0	4.8	106.4	0.55	53.1	1.25	0.22
908	2004	26.6	4.9	-	0.82	-	0.97	0.16
Bay of Quinte								
B	2002-04	24.1	7.6	12.3	3.10	10.8	0.98	0.16
HB	2002-04	23.7	6.8	14.0	2.22	14.2	0.95	0.15
C	2002-04	22.2	4.2	19.3	1.24	21.9	0.28	0.05

Table 4. Seasonal areal phytoplankton photosynthesis (g C m^{-2}) in Hamilton Harbour 2002-2004.

Location	Year	Empirical Irradiance	Cloudless Irradiance	% Cloudless
Nearshore				
17	2002	156	202	77
6	2002	263	378	70
WC	2003	252	343	74
Offshore				
258	2002	282	398	71
	2004	190	290	65
908	2003	346	470	74
	2004	215	324	66
Bay of Quinte				
B	2002	248	312	80
	2003	289	386	75
	2004	189	282	67
HB	2002	282	358	79
	2003	359	469	77
	2004	136	200	68
C	2002	124	154	81
	2003	188	245	77
	2004	119	165	72

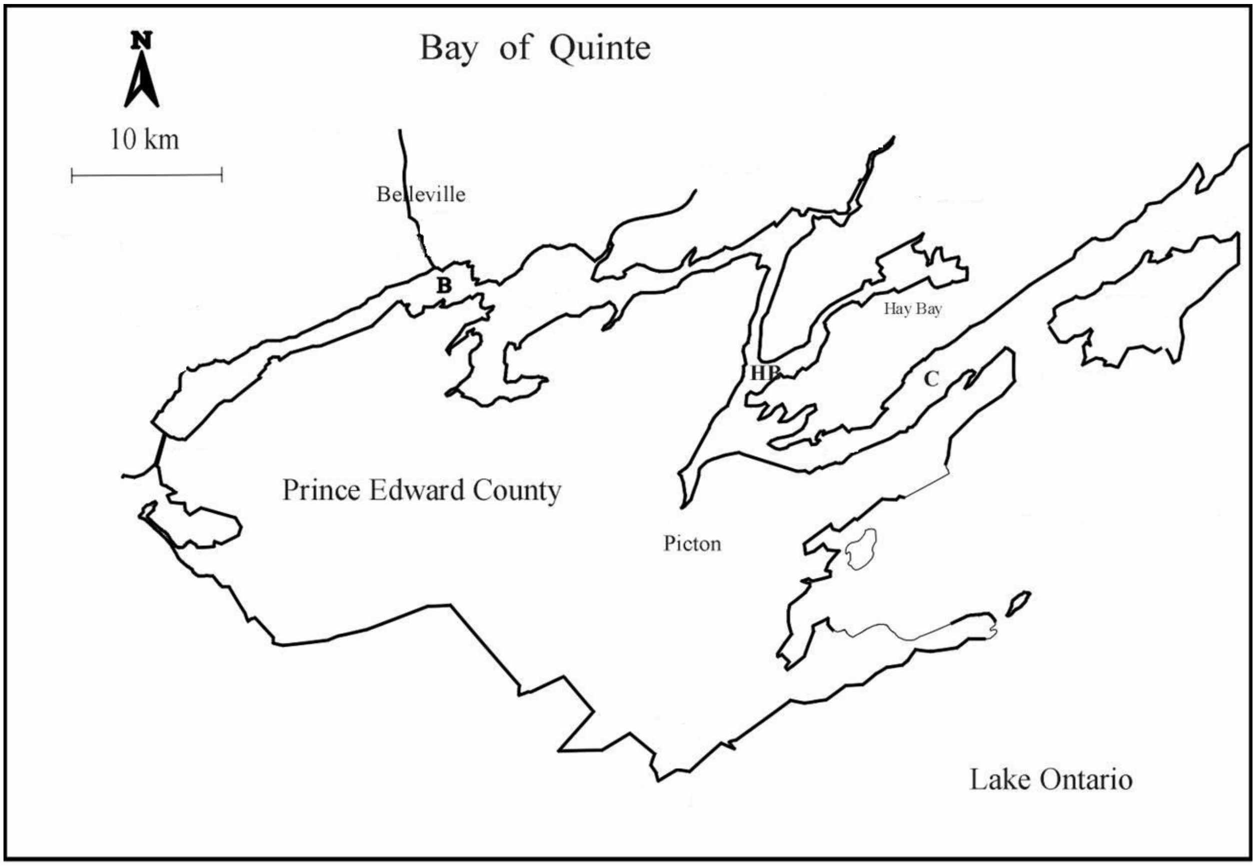


Fig. 1. Bay of Quinte and location of sample stations: Belleville (B), Hay Bay (HB), Conway (C).

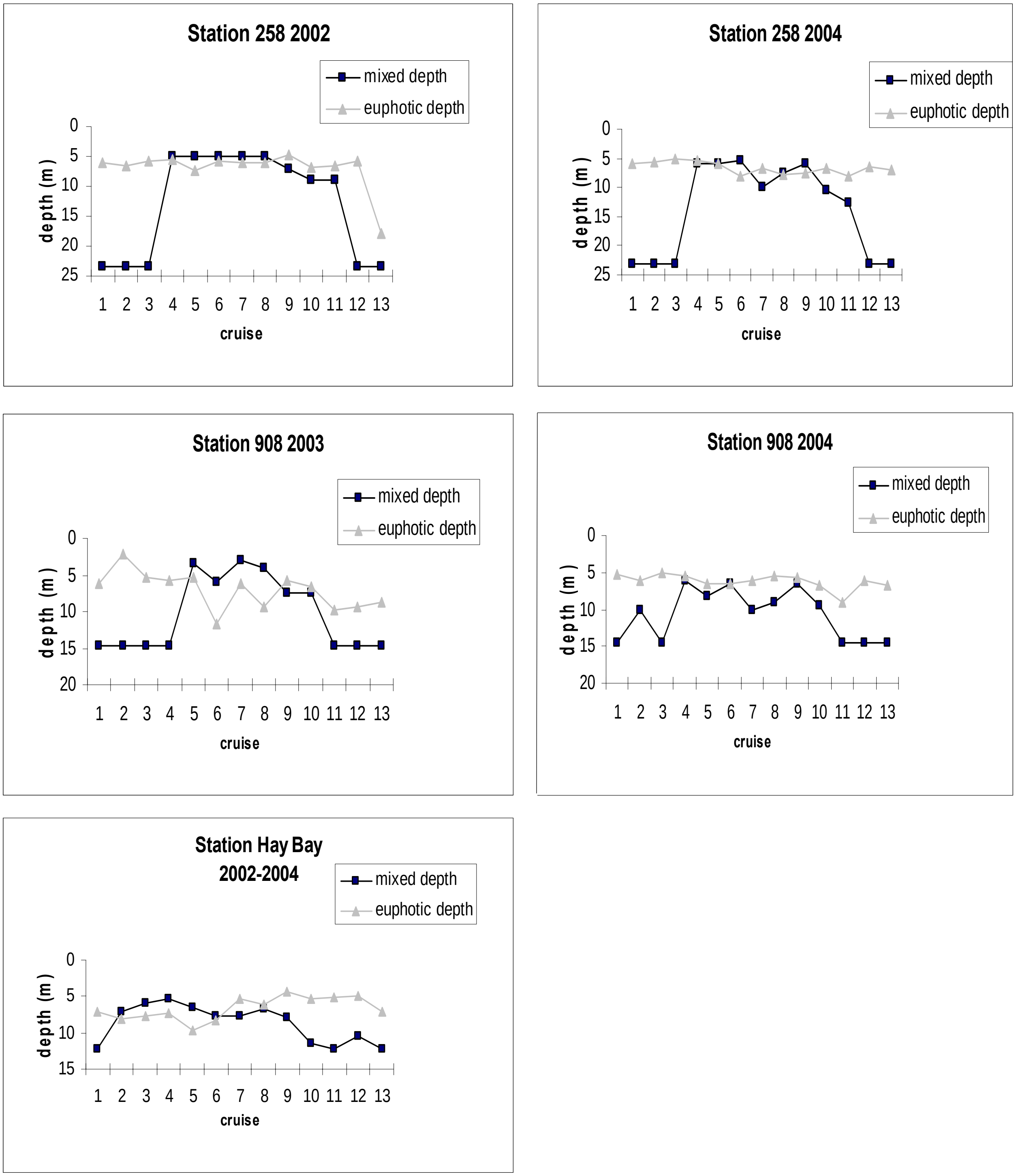


Fig. 2. Mixed epilimnetic and euphotic zone depths for Hamilton Harbour and Hay Bay in 2002-2004.

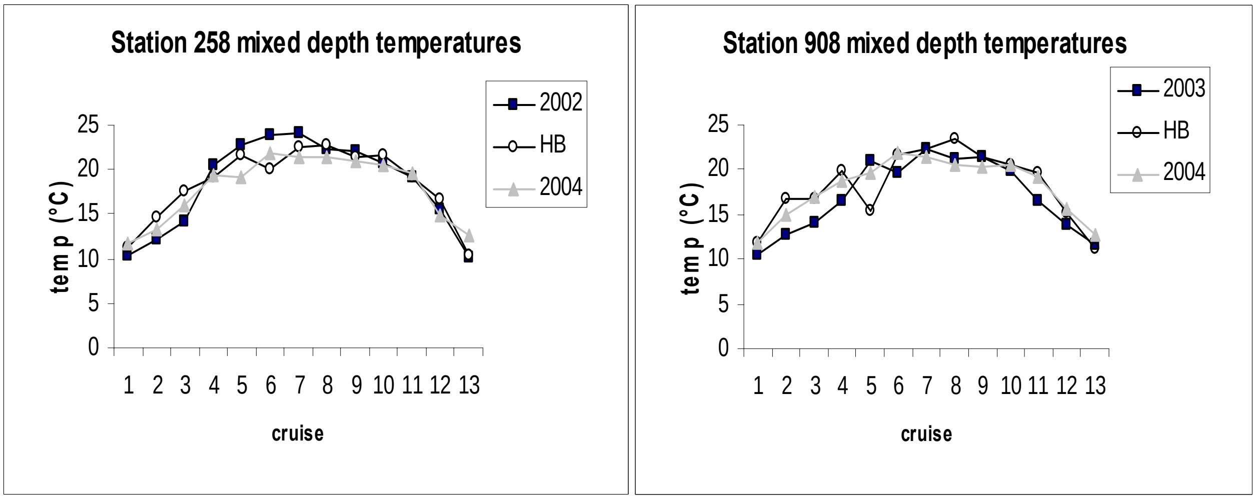


Fig. 3. Mixed epilimnetic temperatures for station 258 (2002, 2004) and station 908 (2003, 2004) compared to that at Hay Bay.

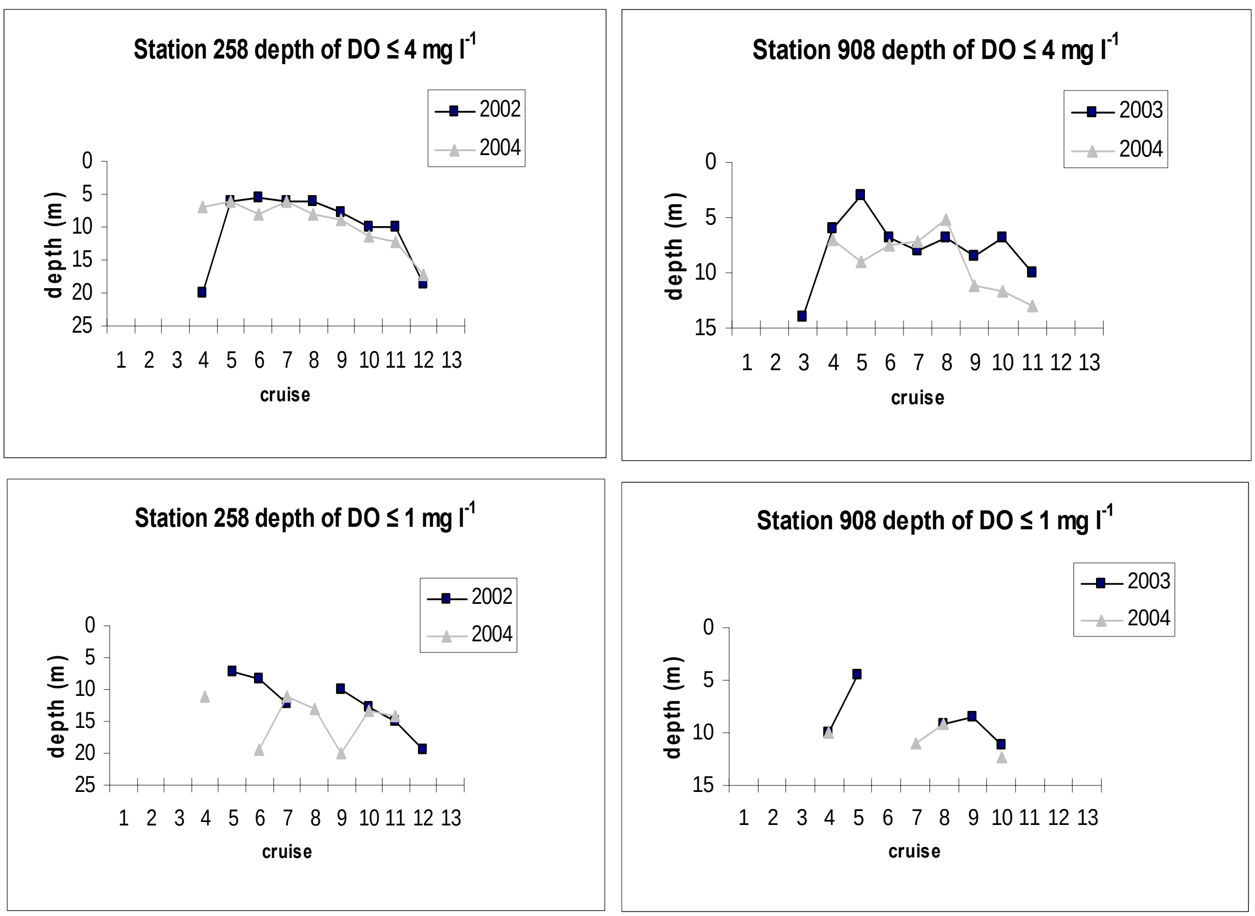


Fig. 4. Initial depth where low dissolved oxygen concentrations of 4 or 1 mg l^{-1} were recorded in Hamilton Harbour 2002 – 2004.

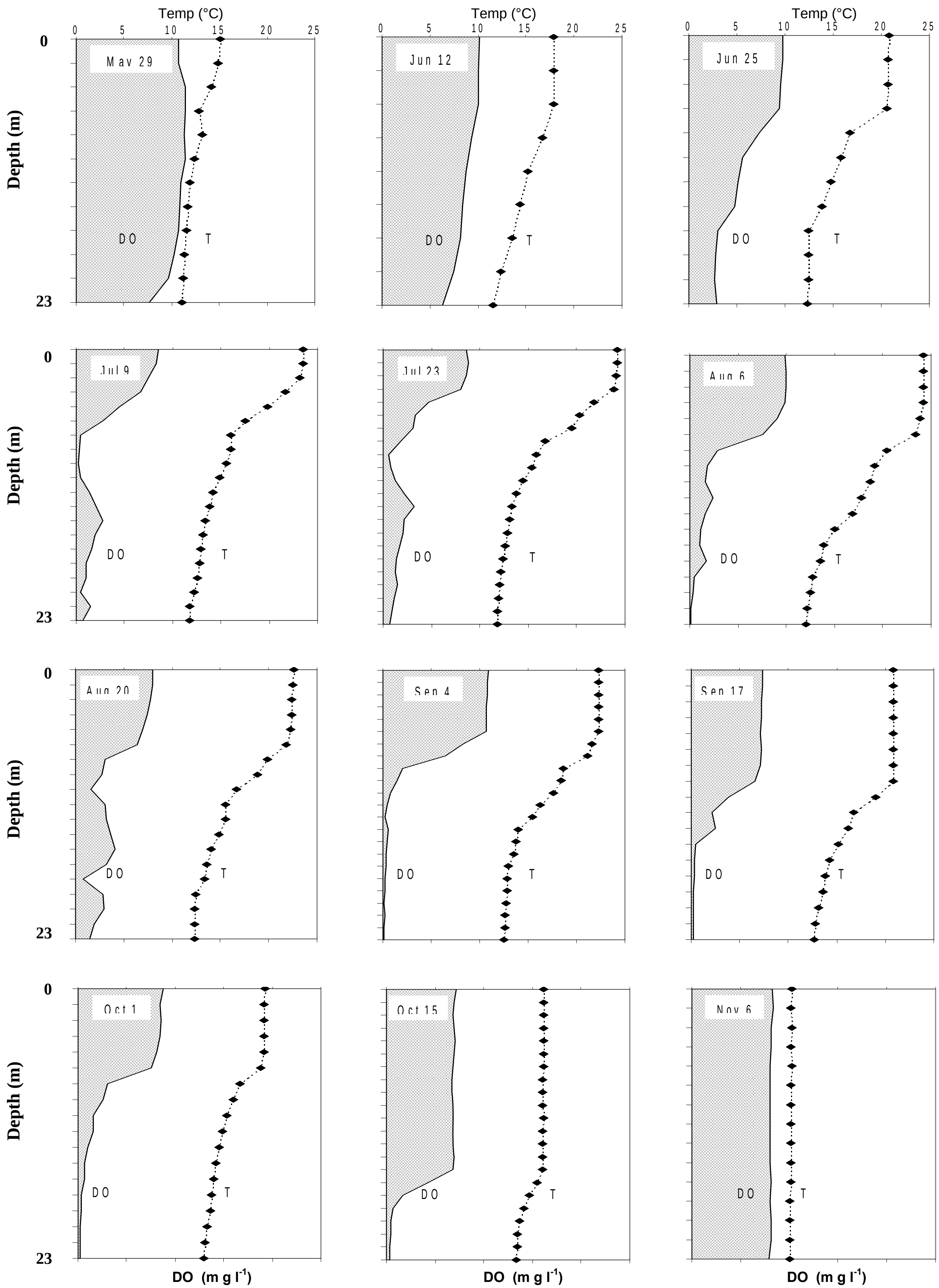


Fig. 5. Temperature and dissolved O_2 concentration profiles versus depth for station 258 in 2002.

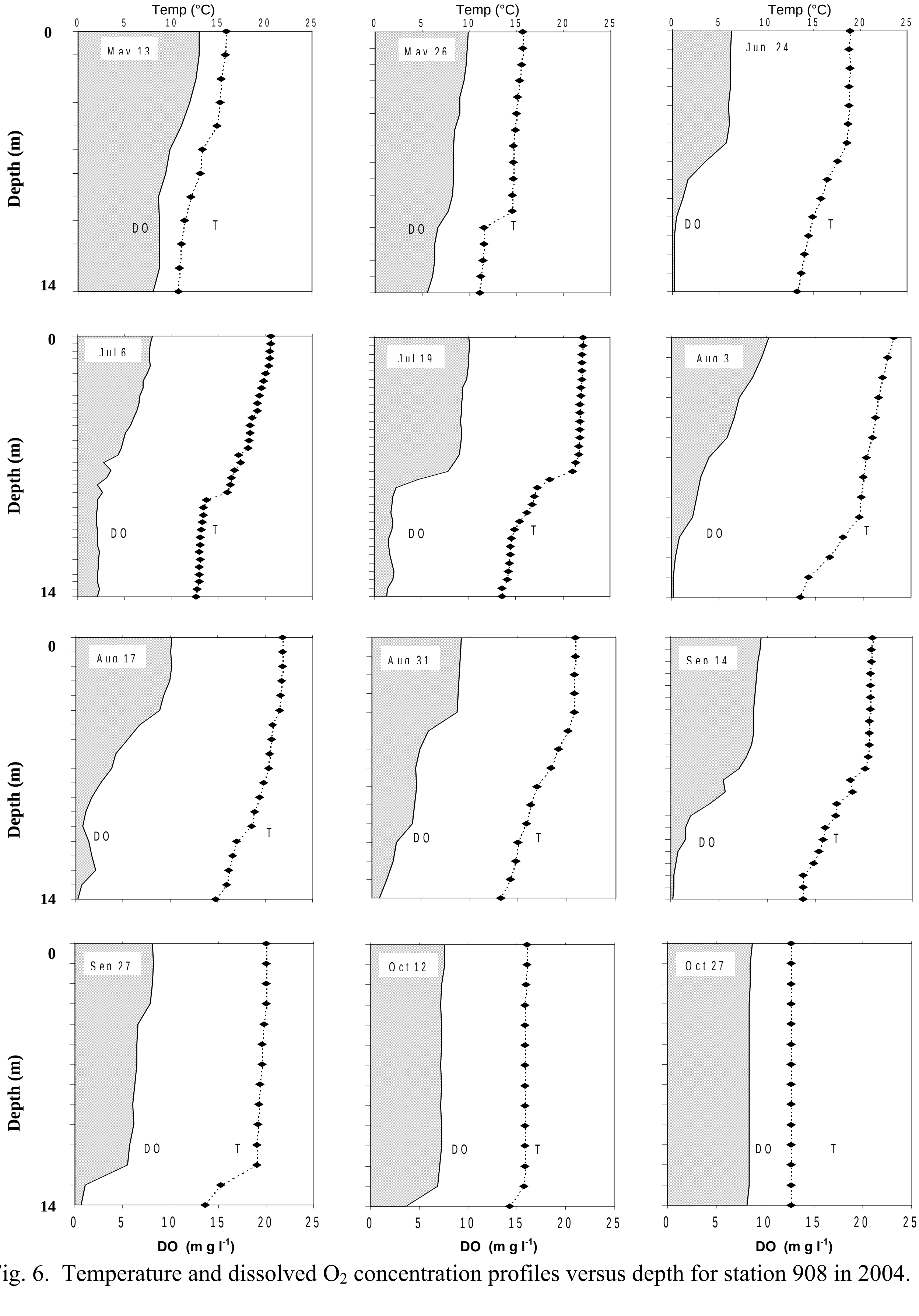


Fig. 6. Temperature and dissolved O_2 concentration profiles versus depth for station 908 in 2004.

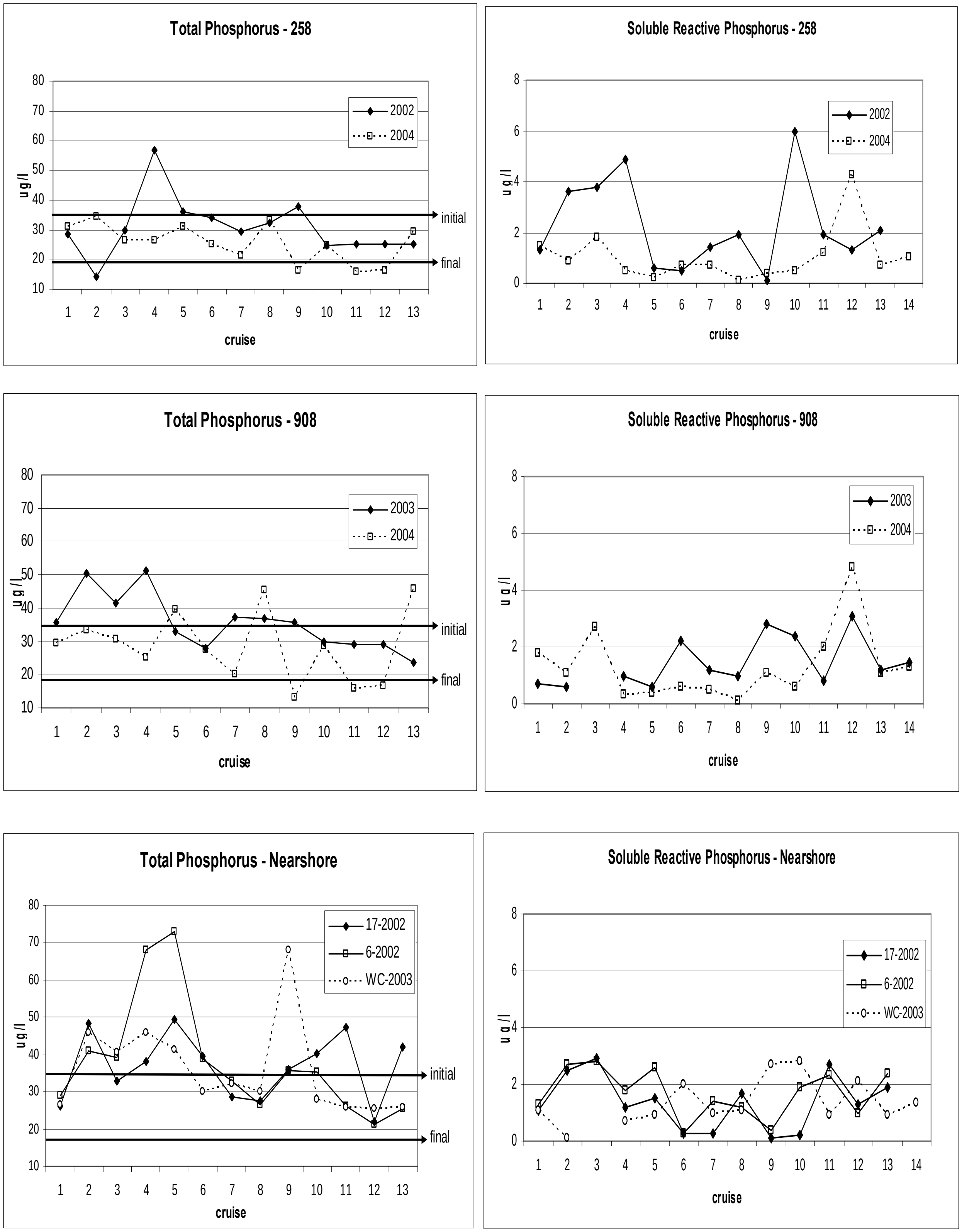


Fig. 7. Total phosphorus and soluble reactive phosphorus concentrations for Hamilton Harbour 2002 – 2004 dark lines indicating RAP initial and final targets for total phosphorus.

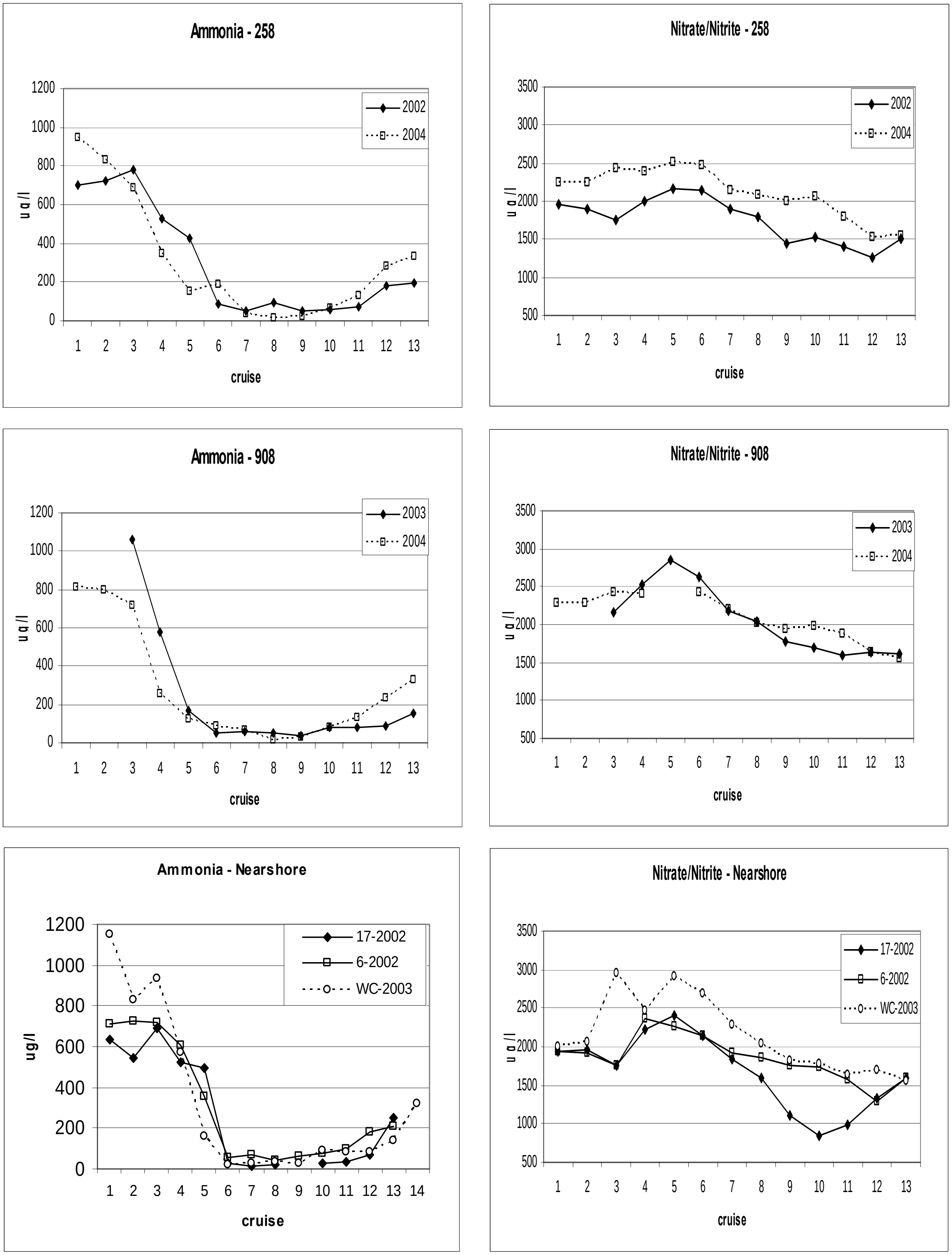


Fig. 8. Dissolved ammonia and dissolved nitrate/nitrites concentrations for Hamilton Harbour 2002 – 2004.

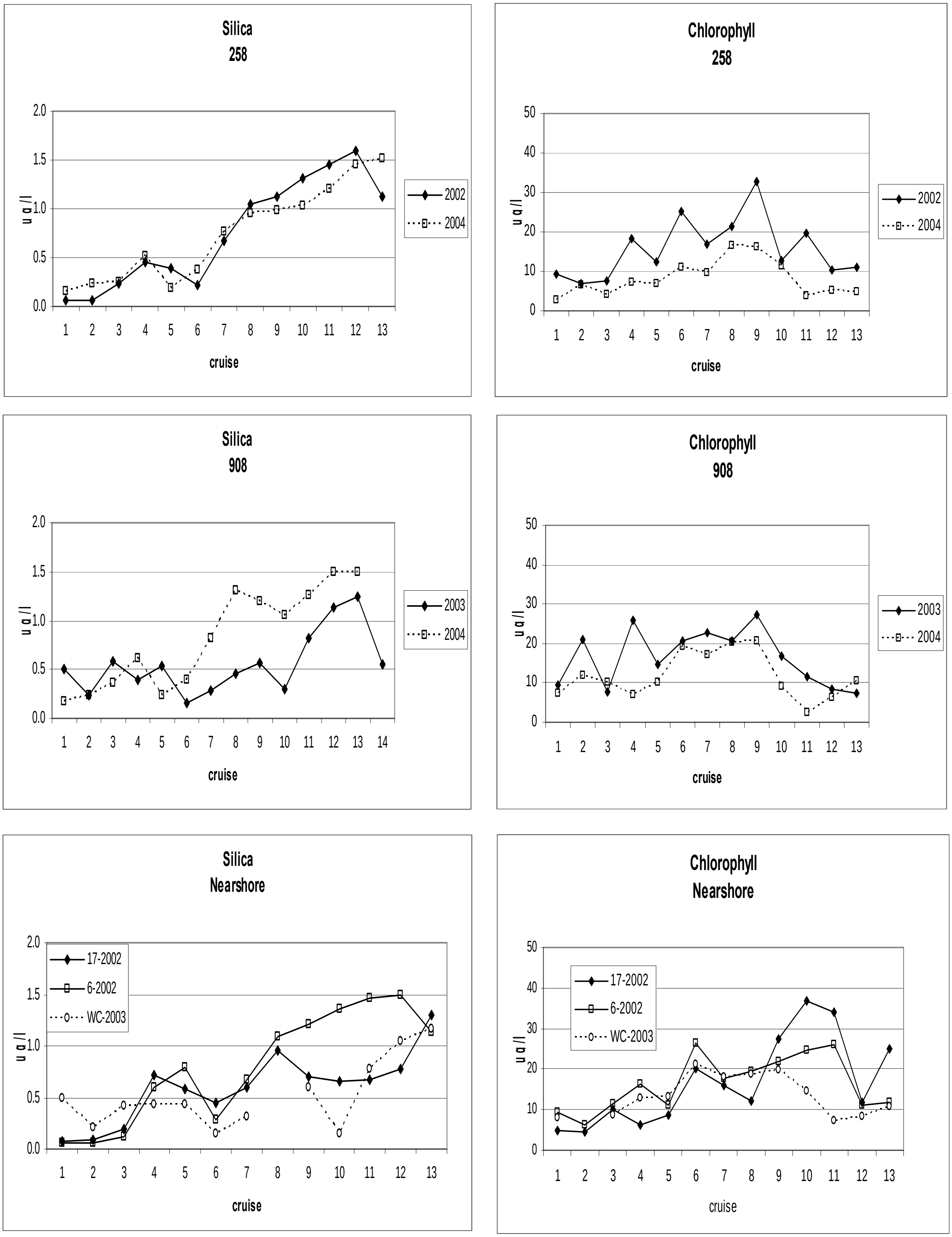


Fig. 9. Soluble reactive silica and chlorophyll concentrations for Hamilton Harbour 2002 – 2004.

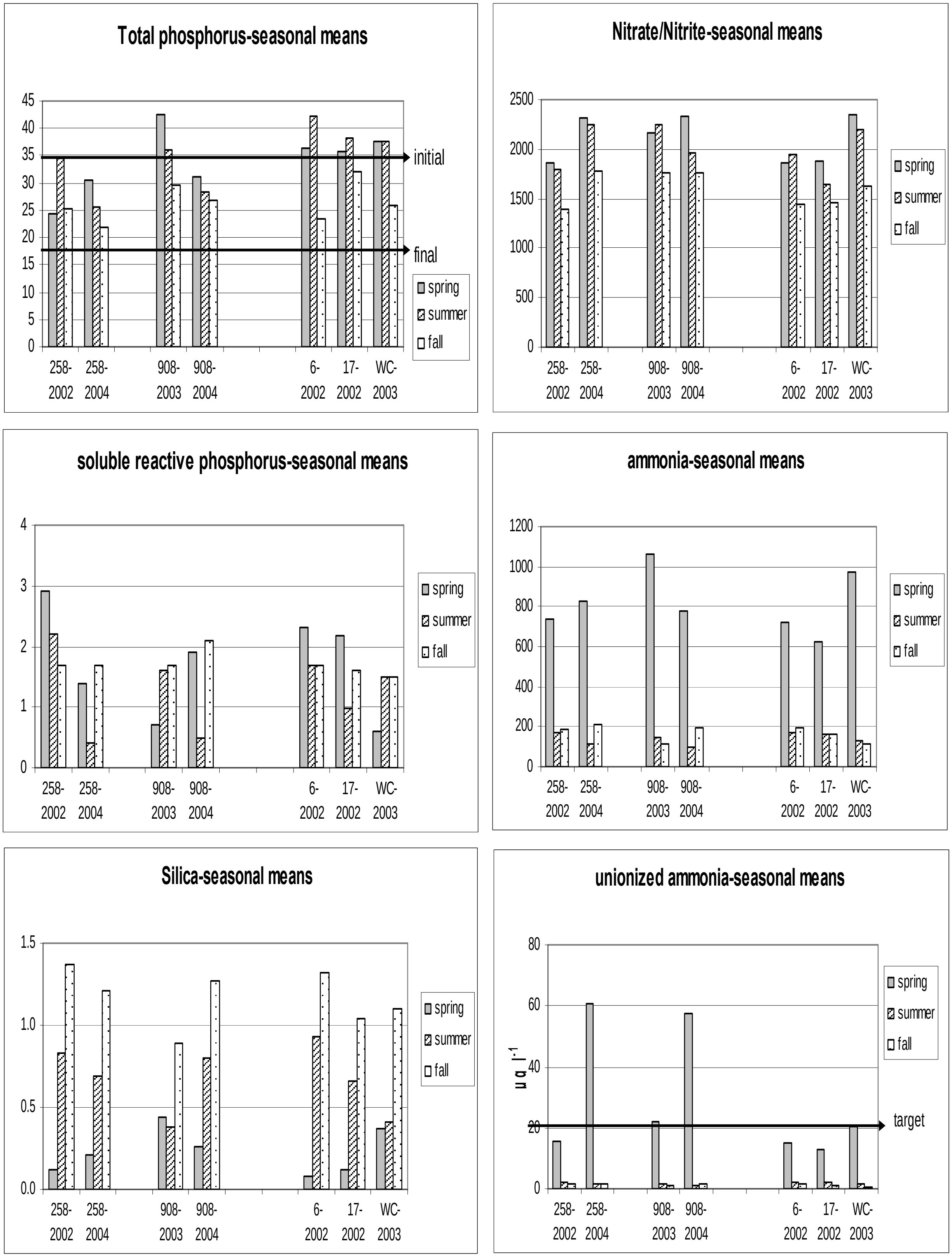
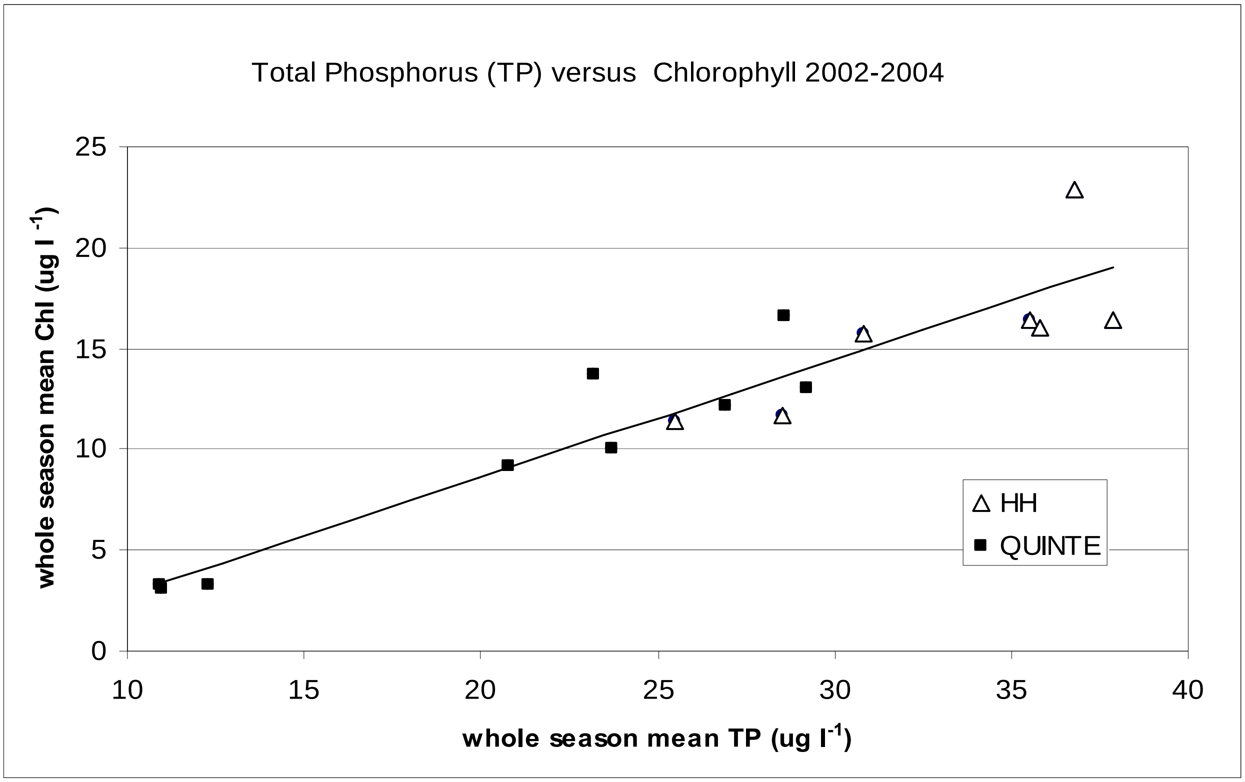


Fig. 10. Seasonal nutrient concentrations ($\mu g\ l^{-1}$) for Hamilton Harbour 2002 – 2004. Dark lines indicate RAP initial and final targets.



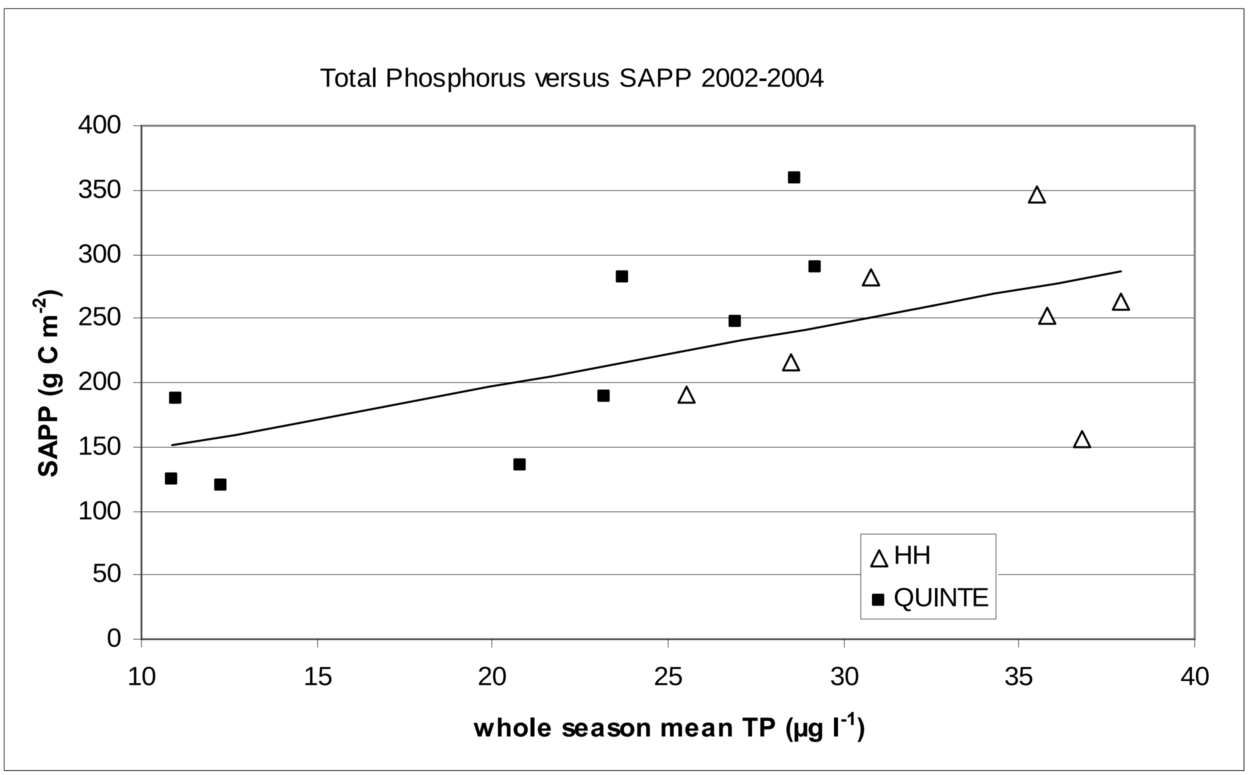


Fig. 11. Whole season mean total phosphorus versus chlorophyll *a* concentrations and SAPP for Hamilton Harbour and the Bay of Quinte (B, HB, C) for 2002 – 2004.

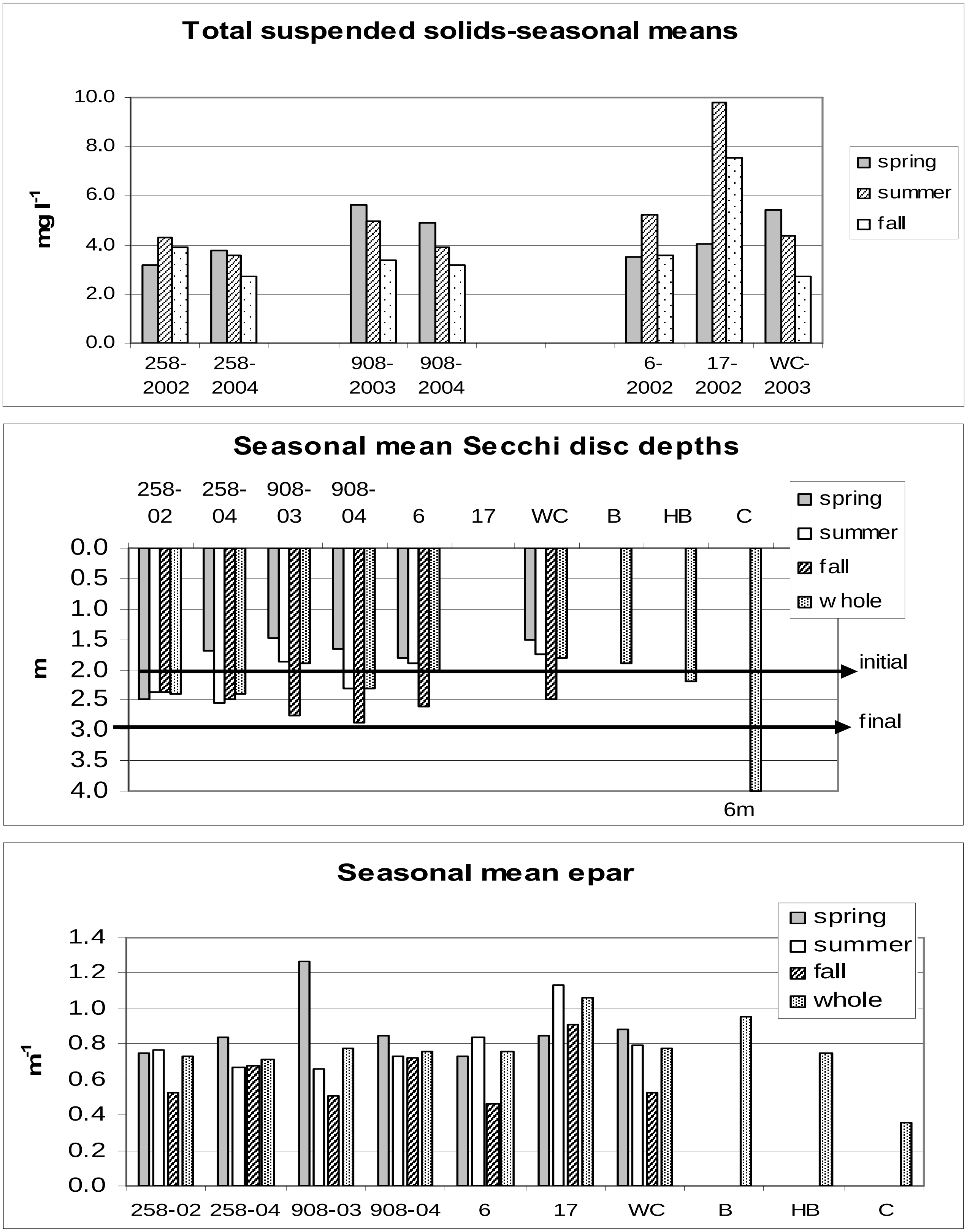
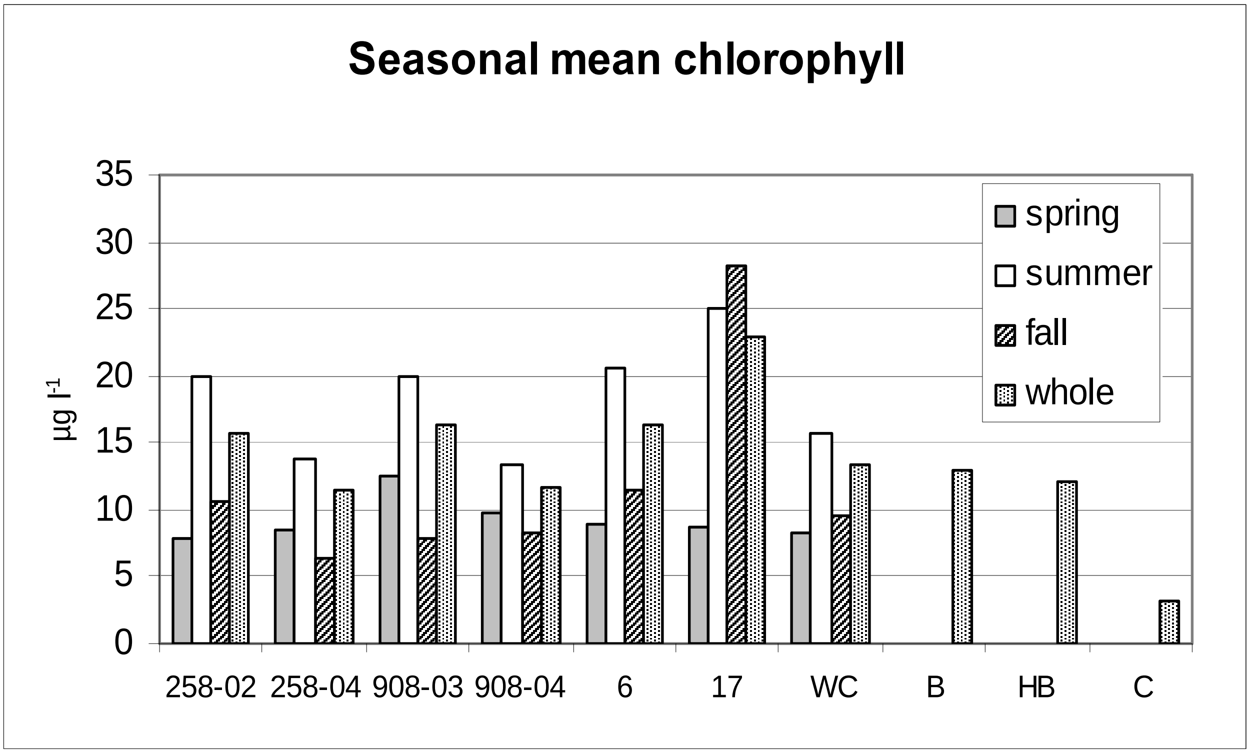


Fig. 12. Seasonal mean total suspended solids, Secchi disc depths and ε_{par} for Hamilton Harbour 2002 – 2004 and the Bay of Quinte. Dark lines indicate RAP initial and final targets.



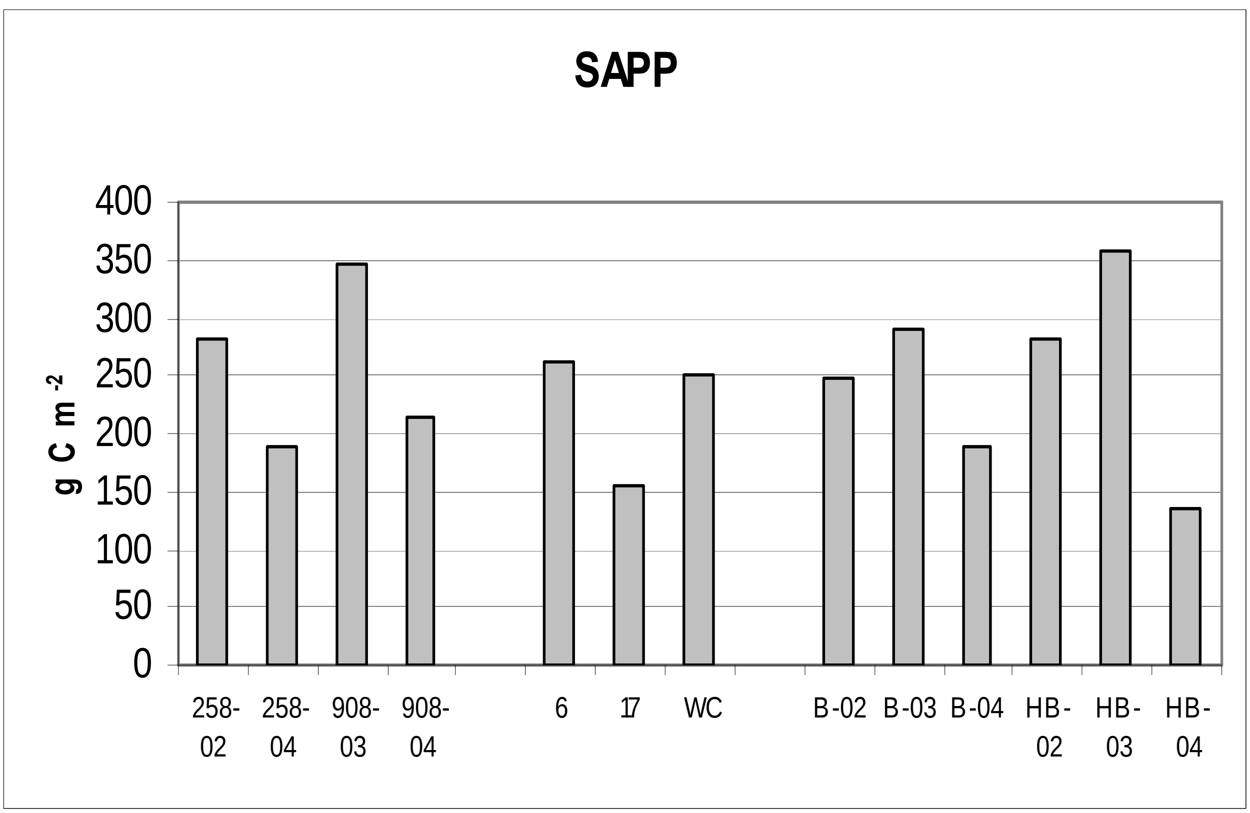
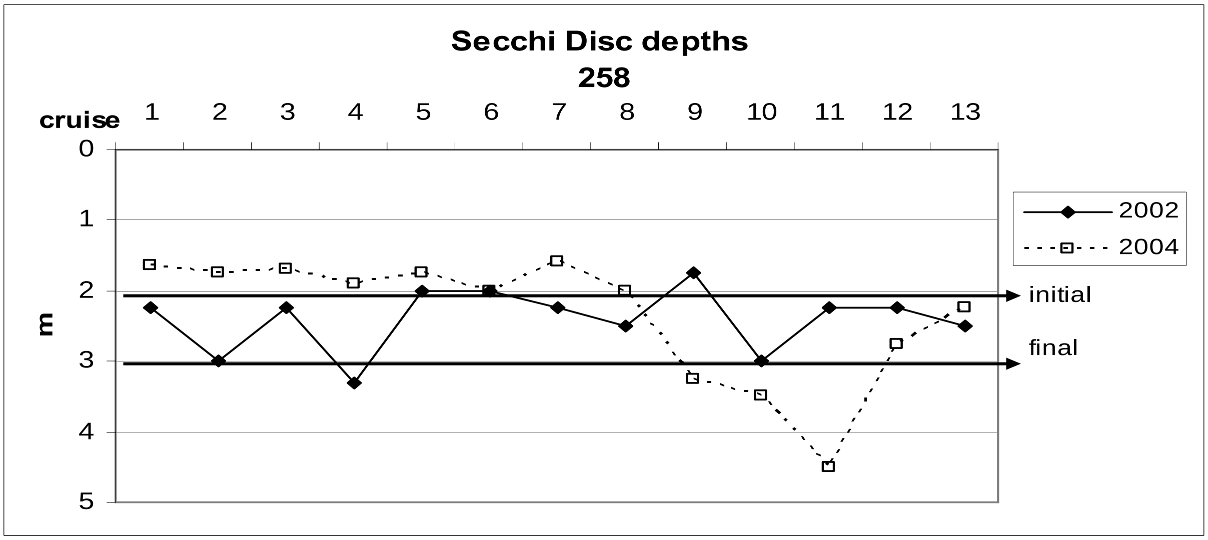
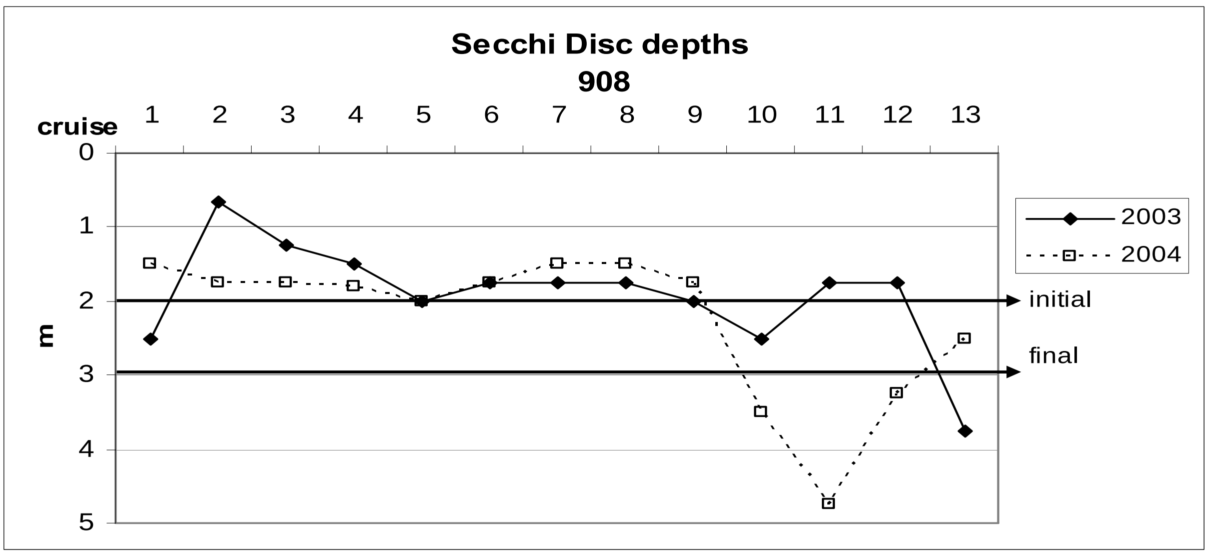


Fig. 13. Seasonal mean chlorophyll concentrations and average areal phytoplankton photosynthesis (SAPP) for Hamilton Harbour, 2002 – 2004.





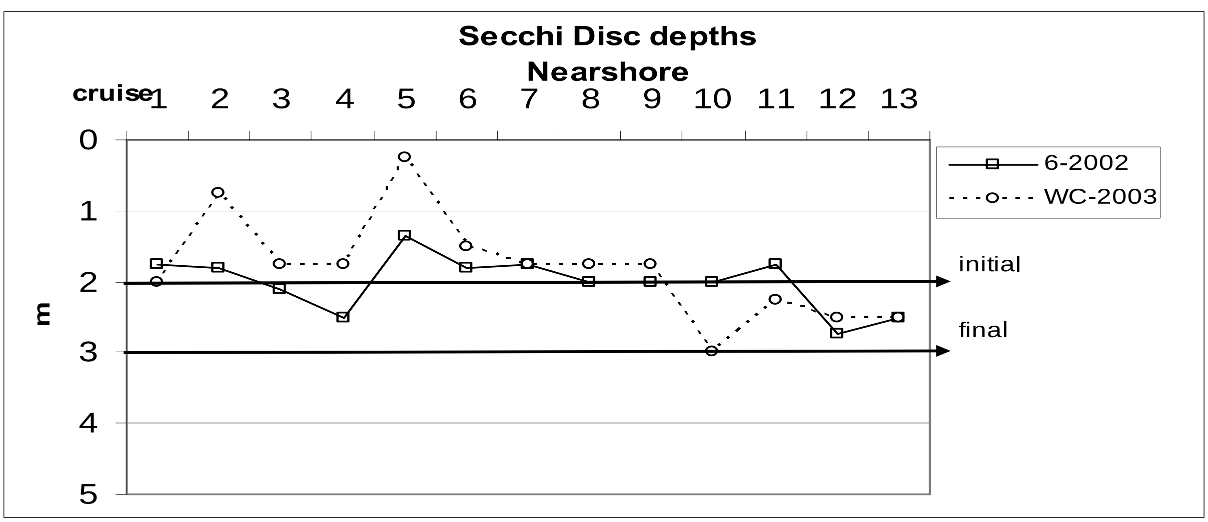


Fig. 14. Secchi disc depths for Hamilton Harbour 2002 – 2004. Dark lines indicate RAP initial and final targets.

AN INTEGRATED ASSESSMENT OF THE MICROBIAL AND PLANKTONIC COMMUNITIES OF HAMILTON HARBOUR

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INTRODUCTION

Studies of the microbial and planktonic food webs of aquatic ecosystems have proven to be important in increasing our understanding of ecosystem structure and function. Taken together, the microbial and planktonic communities form the pelagic component of the lower food web and play an important role in transferring autochthonous energy to higher trophic levels (e.g. Munawar and Weisse 1989, Munawar et al. 2005, Fitzpatrick et al. 2007). Thus, healthy fisheries and healthy ecosystems are dependant upon the relative health of the lower food web.

With respect to Hamilton Harbour, adverse health effects termed "Beneficial Use Impairments", have been known to persist for some time and were responsible for it's designation as an "Area of Concern" (International Joint Commission (IJC) 1989). These Beneficial Use Impairments (BUIs) included "water quality and bacterial contamination" and "fish and wildlife impacts" as a result of municipal and industrial wastes (Environment Canada 2005). Fisheries & Oceans Canada undertook a temporally extensive (May – October) survey of the bay from 2002–2004 in order to provide a baseline assessment of the health of the microbial and planktonic food webs.

The structure of the Hamilton Harbour food web was assessed, including bacteria, autotrophic picoplankton, phytoplankton, heterotrophic nanoflagellates and ciliates, using standard techniques. Size fractionated primary productivity was included as a functional measurement. Furthermore, a preliminary analysis of phytoplankton and zooplankton interactions was undertaken to provide insights into predator – prey interactions and energy transfer at the lowest trophic levels. The results of this study will provide scientists and managers with important information on this highly complex and stressed ecosystem.

MATERIALS AND METHODS

Two stations within Hamilton Harbour were sampled for this study and are shown in Figure 1. Station 258 was sampled from May to October 2002 and 2004, and station 908 was sampled from May to October 2003 and 2004. Microbial loop samples (bacteria, autotrophic picoplankton and heterotrophic nanoflagellates) were fixed with 1.6% formaldehyde and enumerated using DAPI staining (Porter and Feig 1980) under epi-fluorescence microscopy (Munawar and Weisse 1989). Freshweight biomass was estimated as 0.91 pg $cell^{-1}$ for bacteria, 1.82 pg $cell^{-1}$ for autotrophic picoplankton and 127 pg $cell^{-1}$ for heterotrophic nanoflagellates (Sprules et al. 1999). Ciliate samples were preserved with Lugol's iodine and enumerated following the Quantitative Protargol Staining technique (Montagnes and Lynn 1987). Phytoplankton samples were fixed immediately with Lugol's iodine. Identification and enumeration followed the Utermöhl (1958) inverted microscope technique (see Munawar et al. 1987). Zooplankton data from Gerlofsma et al. (*this volume*) was also incorporated in this analysis.

Size fractionated primary productivity was determined for three size categories of phytoplankton (<2 µm, 2-20 µm and >20 µm) following the standard protocol of Munawar and Munawar (1996). Whole water samples were spiked with $Na^{14}CO_3$, incubated for 2 - 4 hours at surface temperatures and irradiance levels equivalent to P_{opt} (Fee 1969). After incubation, size classes were determined via filtration of the sample water through polycarbonate filters and radioactivity was determined by liquid scintillation counting.

RESULTS AND DISCUSSION

Hamilton Harbour is a highly variable physical, chemical and biological system. As with other areas in the Great Lakes, phosphorus abatement was introduced in the 1970s in order to control eutrophication and ultimately improve the health of the ecosystem. Target total phosphorus concentrations of 34 µg l^{-1} (interim) and 17 µg l^{-1} were set under the Great Lakes Water Quality Agreement. Previous reports have indicated that the interim target total phosphorus concentration was met by the late 1980s (e.g. Charlton and Le Sage 1996), however further reductions are still to be realized (see Burley, *this volume*). Fisheries & Oceans Canada implemented a holistic monitoring program of Hamilton Harbour in 2002 in order to assess the current health of the lower trophic levels.

Size Fractionated Primary Production

Size fractionated primary productivity for 2002 – 2004 at both stations is summarized in Figures 2 and 3. At station 258, net plankton (>20 µm) productivity (SWM) declined from 33.5 – 13.5 mg C m^{-3} h^{-1} between 2002 and 2004. Nanoplankton (2 – 20 µm) productivity declined very slightly from 18.7 – 17.1 mg C m^{-3} h^{-1} during the same time frame, and picoplankton (<2 µm) decreased from 22.3 – 5.4 mg C m^{-3} h^{-1}. SWM primary productivity also revealed a decreasing trend at station 908 from 2003 to 2004. Net plankton productivity fell slightly from 15.0 – 12.9 mg C m^{-3} h^{-1}, net plankton from 38.1 – 22.8 mg C m^{-3} h^{-1} and picoplankton from 10.7 – 5.7 mg C m^{-3} h^{-1}. Primary productivity was high and dominated by larger net (>20 µm) and nano (2-20 µm) plankton which is generally characteristic of a eutrophic system.

Microbial Loop

The seasonal distribution of bacteria, autotrophic picoplankton, heterotrophic nanoflagellates and ciliates for stations 258 and 908 are shown in Figures 4 and 5, respectively. The seasonally weighted mean (SWM) bacterial biomass at 258 declined from 451.5 mg m^{-3} in 2002 to 418.6 mg m^{-3} in 2004. Similar trends were apparent in picoplankton, declining from 98.9 to 15.4 mg m^{-3}, heterotrophic nanoflagellates from 155.6 to 104.9 mg m^{-3}, and ciliates from 89.6 to 65.1 mg m^{-3} between years. At station 908, SWM microbial loop biomass also declined between years, with bacteria falling from 545.7 to 452.1 mg m^{-3}, picoplankton from 78.6 to 29.6 mg m^{-3}, nanoflagellates from 212.9 – 166.1 mg m^{-3} and ciliates from 93.2 – 62.8 mg m^{-3}. The microbial loop was dominated by bacteria at both sites, with seasonally weighted mean biomass ranging from 420 – 550 mg m^{-3}, and was typically 4-5 times greater than each of the other components.

Phytoplankton Biomass and Composition

The seasonal distribution of the phytoplankton community including biomass and % composition (by biomass) is shown in Figures 6-9. At station 258, SWM biomass declined from 2034.6 mg m^{-3} in 2002 to 1819.3 mg m^{-3} in 2004. Dominant taxonomic groups (>50% of biomass) included: Diatomeae, Chlorophyta, Cryptophyceae, and Dinophyceae during 2002. During 2004, the dominant phytoplankton groups included Chrysophyceae, Chlorophyta and Dinophyceae. At station 908, SWM phytoplankton biomass showed a very small increase from 1448.9 to 1477.1 mg m^{-3} from 2003 to 2004. Dominant phytoplankton groups throughout 2003 included Diatomeae, Cryptophyceae, Chlorophyta and Dinophyceae. In 2004, dominant taxa included Chrysophyceae, Diatomeae, Chlorophyta and Dinophyceae. A detailed listing of species contributing greater than 5% to the total biomass of any sample is given in Table 1 for station 258 and Table 2 for station 908.

Phytoplankton biomass was quite high in the harbour (SWM: 1450 – 2040 mg m^{-3}) indicating mesotrophic conditions based on the trophic index of Munawar and Munawar (1982). High biological variability was evident in the phytoplankton community at both sites where Diatomeae, Dinophyceae, Chlorophyta and Chrysophyceae were all dominant at different times throughout each year. The rapidly changing phytoplankton community might be a consequence of the physical disturbances (e.g. wind, cargo ships) that Hamilton Harbour is subject to. These physical disturbances have previously been shown to limit the overall size of standing crop (Haffner et al. 1980) and could be expected to have a bottom-up effect on the zooplankton community. A detailed discussion of phytoplankton and zooplankton interactions follows.

TROPHIC INTERACTIONS IN HAMILTON HARBOUR

The extreme variability observed in the phytoplankton community of Hamilton Harbour, both in terms of biomass and composition could be expected to have a bottom-up influence on the zooplankton community. We therefore consider the interactions of zooplankton and phytoplankton in an attempt to gain some understanding of predator-prey interactions at the base of the food web. This is a preliminary analysis of the phytoplankton – zooplankton relationship styled after Munawar et al. 2007. Other factors which could influence this relationship, such as nutrient dynamics affecting phytoplankton growth and planktivore predation on zooplankton are not directly considered here, but are important nonetheless.

Station 258, 2002

Spring

Zooplankton biomass was initially low during spring of 2002 (≈ 0.1-0.6 g m^{-3} fresh weight) before surging to 2.8 g m^{-3} as part of an upward trajectory that would continue into the summer (Fig. 6). The zooplankton community was initially dominated by Cyclopoids, but rapidly shifted towards Cladocera. Phytoplankton biomass was initially high (3.2 g m^{-3}) but declined to 0.9 g m^{-3} by late spring. The composition of the phytoplankton community also showed a rapid shift from Diatomeae to Chlorophyta in this period. Interestingly, almost 50% of the zooplankton biomass was carnivorous during the spring, which would normally be associated with low biomass, however this proportion held even as biomass increased. The phytoplankton community was then dominated by largely edible species of *Stephanodiscus*, *Scenedesmus* and *Coelastrum*, which likely helped sustain the herbivorous zooplankton and would in turn provide food for the carnivorous zooplankton.

Summer

Zooplankton biomass showed a bimodal pattern during summer, soaring to 10 g m^{-3} then falling to 2 g m^{-3} and rising again to 10 g m^{-3} before dampening out in the 2 – 3 g m^{-3} range (Fig. 6). Cladocera dominated the zooplankton biomass until the late summer when Cyclopoids became more prevalent. A peak of dreissenid veligers was also observed in mid summer. Phytoplankton biomass showed considerable variability throughout the summer and peaked at 4.1 g m^{-3} in early September. Phytoplankton composition was quite variable with Chlorophyta, Cryptophyceae and Dinophyceae overwhelmingly dominating the biomass at different periods throughout the season. The proportion of carnivorous zooplankton decreased to 20 – 30% of the biomass as largely edible species of phytoplankton dominated the biomass including *Cryptomonas reflexa, Oocystis lacustris, Scenedesmus braziliensi,* and *Coelastrum reticulum*, although the inedible *Ceratium furcoides* was prevalent in late summer. The large biomass of herbivorous zooplankton observed throughout the summer may have helped create conditions for the observed dinoflagellate bloom by reducing the standing crop of edible algae.

Fall

Zooplankton biomass declined in the fall from its late summer peak of 3.4 g m^{-3} to 0.6 g m^{-3} (Fig. 6). Zooplankton composition was almost equally split between Cyclopoids and Cladocera, along with a smaller component of Calanoids. Phytoplankton biomass also declined during fall from a late summer peak of 4.1 g m^{-3} to 0.5 g m^{-3} and was tightly coupled with zooplankton biomass. Fall biomass was overwhelmingly dominated by species of Chlorophyta including *Coenochloris pyrenoidosa* and *Coelastrum reticulatum*. The close relationship between phytoplankton and zooplankton biomass observed during fall may be a result of the large proportion of herbivorous zooplankton observed and the correspondingly large population of edible algae.

Station 258, 2004

Spring

Zooplankton biomass at station 258 increased during the spring of 2004 from 0.7 to 2.9 g m^{-3} as the composition shifted from Cyclopoids to Cladocera (Fig. 7). Phytoplankton biomass increased from 0.8 g m^{-3} to 2.2 g m^{-3} but then began to wane in the late spring and early summer. Phytoplankton was composed mainly of Cryptophyceae followed by Diatomeae during this period. Slightly more than half of the zooplankton community was herbivorous and the phytoplankton community contained mostly edible species including *Rhodomonas minuta*, *Cryptomonas reflexa* and *Stephanodiscus niagarae*.

Summer

Zooplankton reached its first maxima in early summer of 4.1 g m^{-3} before declining to 1.7 g m^{-3} in mid summer and increasing to 3.5 g m^{-3} by late summer (Fig. 7). Zooplankton communities in early and late summer were dominated by Cladocera while a mid summer peak of Cyclopoids was observed. Phytoplankton biomass showed an increasing trend peaking at 3.6 g m^{-3} in mid summer before declining somewhat to 2.8 g m^{-3} in late summer. Phytoplankton was composed of Chlorophyta in the early to mid summer period, while Cryptophyceae and Dinophyceae became more prevalent in the late summer period. The proportion of herbivorous zooplankton peaked in mid July at 80% of the biomass but generally ranged from 50-60% of the total biomass. Phytoplankton contained some edible species including *Cryptomonas reflexa* and *Lagerheimia ciliate* but also inedible species including *Dinobryon divergens* and *Ceratium furcoides*.

Fall

Zooplankton biomass peaked in the early fall at 3.5 g m^{-3} and then declined to 0.4 g m^{-3}. Cladocera dominated the early fall period but then Cyclopoids became more prevalent (Fig. 7). Phytoplankton biomass continued its downward trend falling from 2.1 to 0.6 g m^{-3}. The phytoplankton community was dominated by Dinophyceae in early fall but the late fall contained a mixture of Chlorophyta, Diatomeae and Cryptophyceae. The proportion of herbivorous zooplankton declined from 90% to 70% of the biomass and phytoplankton consisted of inedible *Ceratium furcoides* in early fall but edible algae, particularly *Stephanodiscus niagarae,* was prevalent later in the fall.

Station 908, 2003

Spring

Zooplankton biomass at station 908 was high, though variable during spring, ranging from 2.4 – 3.8 g m^{-3} and was dominated by Calanoids, with a significant amount of Cladocera being observed (Fig. 8). Phytoplankton biomass was initially quite high (≈ 2.0 g m^{-3})but dropped to 0.3 g m^{-3} in late spring. Diatomeae followed by Cryptophyceae were the dominant phytoplankton. The zooplankton community was almost evenly split between herbivores and carnivores, and the phytoplankton community contained inedible (*Fragilaria crotenensis*) and edible (*Rhodomonas minuta*) forms of algae.

Summer

Zooplankton biomass displayed a bimodal pattern with an early summer peak of 5.4 g m^{-3} declining to 1.2 g m^{-3} and rising to its secondary peak of 3.3 g m^{-3} and continued to be variable through to the end of the summer (Fig. 8). Zooplankton was dominated by Cladocera throughout the summer although a significant amount of dreissenid veligers were observed mid summer and Cyclopoids became more prevalent in late summer. Phytoplankton biomass also showed a bimodal distribution declining from a maximum of 2.3 g m^{-3} in early summer to a minimum of 0.9 g m^{-3} in mid summer and increasing to its peak of 2.8 in late summer before declining again into fall. Phytoplankton composition was highly variable, proceeding from Cryptophyceae to Chlorophyta and then to Dinophyceae and Cyanophyta. The proportion of carnivorous zooplankton ranged from 25 – 50% of the zooplankton biomass in the summer. Potential phytoplankton prey for the zooplankton contained edible species including *Rhodomonas minuta*, *Cryptomonas reflexa* and *Coelastrum pseudomicrosporum*, although the inedible *Ceratium furcoides* became prominent in late summer.

Fall

Zooplankton biomass at Stn 908 declined in the fall from 0.7 g m^{-3} to 0.4 g m^{-3} and was composed of a mixture of Cyclopoids and Cladocerans although some Calanoids were also present (Fig. 8). Phytoplankton biomass was very similar to zooplankton biomass in this period and followed the same trend declining from 0.5 – 0.1 g m^{-3}. Phytoplankton composition was quite variable and no single group was dominant. The proportion of predator biomass that was carnivorous ranged from 25 - 40% and the potential prey was composed of edible species of *Cryptomonas* (*C. reflexa*; *C. restriformis*), but also various inedible forms of Cyanophyta as well as *Ceratium furcoides*.

Station 908, 2004

Spring

Zooplankton biomass displayed an increasing trend during the spring of 2004 rising from 0.6 – 2.3 g m^{-3} and Cladocera dominated the zooplankton community (Fig. 9). Phytoplankton biomass was very similar to zooplankton biomass, increasing from 0.5 – 2.1 and displaying the same upward trend. Phytoplankton was comprised of Cryptophyta and Diatomeae. During spring predators were split almost evenly between herbivores and carnivores and the prey contained edible species including *Rhodomonas minuta* and *Stephanodiscus niagarae*.

Summer

Zooplankton biomass displayed a bimodal pattern during summer 2004, soaring to a peak of 5.3 g m^{-3} in early summer, and then declining to 1.2 g m^{-3} by mid summer before rising to a second peak of 5.3 g m^{-3} in late summer (Fig. 9). Cladocerans dominated the zooplankton biomass from early to mid summer, while Cyclopoids became more prevalent during late summer. Phytoplankton biomass reached its lowest level of 0.8 g m^{-3} in early summer as zooplankton biomass peaked, but gradually increased over the mid summer period to a high of 2.9 g m^{-3} as zooplankton levels declined. Phytoplankton biomass declined again in late summer to 0.8 g m^{-3} as zooplankton biomass increased. During the early part of the summer, phytoplankton was comprised of a mixture of Chlorophyta and Diatomeae followed by Chlorophyta in the mid summer and Dinophyceae in the late summer. The ratio of carnivorous to herbivorous zooplankton was highly variable during the summer, with 45% of the zooplankton being carnivorous in the early summer, followed by a sharp decline to 8% in mid summer and ranging from 20 – 30% throughout the rest of the summer. The rapid shifts in zooplankton feeding are associated with rapid shifts in phytoplankton composition. Edible phytoplankton species included *Stephanodiscus niagarae* in the early summer, *Coelastrum pseudmicroporum* and *Pediastrum boryanum* in mid summer, and *Cryptomonas reflexa* in late summer. However, a significant proportion of inedible species including *Dinobryon divergens* and *Ceratium furcoides* were present throughout the summer.

Fall

Zooplankton biomass was still high in the early fall, 4.3 g m^{-3}, but declined rapidly to less than 0.5 g m^{-3} for the remainder of the sampling season (Fig. 9). Cladocera dominated the zooplankton biomass in the early fall, but was later replaced by a mixture of Cyclopoids and calanoids. Phytoplankton biomass increased from 0.8 – 2.0 g m^{-3} at the end of the sampling season and composition shifted from Dinophyceae to Cryptophyceae. Approximately 70 – 80% of the zooplankton was herbivorous during the fall, and the potential prey shifted from largely inedible *Ceratium furcoides* in early fall to edible *Cryptomonas reflexa*.

SUMMARY AND CONCLUSION

Some interesting observations on the lower food web of Hamilton Harbour were made during our study. The first is that phytoplankton biomass was lower than expected, given that the harbour is classified as eutrophic. While other factors, including high total phosphorus concentrations and extended periods of hypoxia in the lower thermal stratum (Burley, *this volume*) were indicative of eutrophic conditions, phytoplankton biomass was indicative of mesotrophic conditions. The reduced biomass suggests that phytoplankton is responding to reductions in phosphorus loadings. However, it also raises the question of whether or not this response is enough to alleviate eutrophication in the harbour.

The second observation was that phytoplankton composition was highly variable. Chlorophyta, Diatomeae, Cryptophyta and Dinophyceae were all dominant groups and different times throughout the sampling season and their respective dominance did not always follow expected seasonal patterns. In part, this may be due to the extreme physical disturbances, including ship traffic and wind and storm events that the harbour is constantly subject to. The constant change observed in species composition would be expected to exert some bottom-up pressure on potential predators including heterotrophic nanoflagellates, ciliates and especially zooplankton.

The third observation is that zooplankton biomass is typically higher than phytoplankton biomass. We would expect the amount of zooplankton predators to be restricted by the availability of phytoplankton prey. Our findings suggest, however, that there may be strong top down control of phytoplankton by zooplankton. We therefore need to consider the feeding ecology of zooplankton which leads to our final observation.

The fourth observation is that the proportion of carnivorous zooplankton to herbivorous zooplankton was relatively high, typically accounting for 25 – 50% of the zooplankton biomass. What these findings suggest is that the standing crop of phytoplankton was sufficient to support a large biomass of herbivorous zooplankton and this in turn was sufficient to support a large biomass of carnivorous zooplankton. Other components of the microbial food web including ciliates and heterotrophic nanoflagellates would also be expected to provide an additional food resource for carnivorous zooplankton. However, it is less than clear how the energy requirements of the zooplankton community is being met and whether or not planktivory is limiting the size of the zooplankton community.

Future research needs to be directed towards understanding energy transfer within the complete microbial and planktonic food web. The high proportion of secondary to primary producers observed in Hamilton Harbour suggests that autochthonous production may not likely be sufficient to sustain the food web. However, other sources of autochthonous energy including benthic algae and macrophytes need to be considered as do allochthonous sources of energy. Furthermore, more integrative research needs to be directed to understanding how the available energy is utilized by the fishery.

Hamilton Harbour continues to be a mesotrophic - eutrophic ecosystem subject to strong physical disturbances. The highly variable nature of the ecosystem was evident in our bi-weekly monitoring of phytoplankton, microbial loop and primary productivity as well as our holistic assessment of the planktonic food web. More intense and continuous sampling strategies need to be deployed to capture short term fluctuations and understand their implications. Continuous monitoring could be achieved with deployment of new technologies including Fluoroprobe, FlowCAM and the Laser Optical Plankton Counter. Energy flow in Hamilton Harbour is driven by primary producers, and research is needed into the transfer of energy from lower to higher trophic levels to understand how fisheries are affected and sustained.

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Table 1. A list of phytoplankton species contributing 5% or more to total biomass of any sample from Stn 258, Hamilton Harbour.

2002	2004
Cyanophyta	**Cyanophyta**
Aphanizomenon flos-aquae	*Lyngbya birgei*
Aphanizomenon sp	*Microcystis aeruginosa*
Chroococcus limneticus	*Pseudanabaena mucioli*
Lyngbya birgei	
Microcystis viridis	
Chlorophyta	**Chlorophyta**
Closteriopsis longissima	*Coelastrum pseudmicroporum Kors*
Closterium dianae	*Coelastrum reticulatum*
Coccomonas orbicularis	*Cosmarium cf margarinatum*
Coelastrum asteroideum	*Lagerheimia ciliata*
Coelastrum reticulatum	*Pandorina morum*
Coenochloris pyrenoidosa	*Staurastrum gracile*
Oocystis lacustris	*Tetraedron minimum*
Oocystis sp	
Scenedesmus braziliensis	
Sphaerocystis schroeteri	
Tetraedron minimum	
Westella botryoides	
Chrysophyceae	**Chrysophyceae**
Ochromonas sp	*Dinobryon divergens*
Diatomeae	**Diatomeae**
Actinocyclus normanii	*Fragilaria capucina*
Fragilaria crotonensis	*Fragilaria crotonensis*
Stephanodiscus binderanus	*Stephanodiscus niagarae*
Stephanodiscus niagarae	
Cryptophyceae	**Cryptophyceae**
Cryptomonas erosa	*Cryptomonas marssonii*
Cryptomonas marssonii	*Cryptomonas reflexa*
Cryptomonas reflexa	*Cryptomonas rostratiformis*
Cryptomonas sp	*Rhodomonas lens*
Rhodomonas lacustris	*Rhodomonas minuta*
Rhodomonas lens	
Rhodomonas minuta	
Dinophyceae	**Dinophyceae**
Gymnodinium helveticum	Ceratium furcoides
Ceratium furcoides	Ceratium hirundenella
	Gymnodinium spp

Table 2. A list of phytoplankton species contributing 5% or more to total biomass of any sample from Stn 908, Hamilton Harbour.

2003	2004
Cyanophyta	**Cyanophyta**
Anabaena crassa	*Microcystis botrys*
Aphanizomenon flos-aquae	*Microcystis wesenbergi*
	Woronichinia naeglianum
Chlorophyta	**Chlorophyta**
Chlamydomonas gracilis	*Chlamydomonas sp*
Chlamydomonas sp	*Coelastrum pseudmicroporum Kors.*
Coelastrum pseudmicroporum Kors.	*Coelastrum reticulum*
Coelastrum reticulatum	*Cosmarium margarinatum*
Monoraphidium contortum	*Pediastrum boryanum*
Pediastrum boryanum	*Staurastrum gracile*
Chrysophyceae	**Chrysophyceae**
Ochromonas sp	*Dinobryon divergens*
Diatomeae	**Diatomeae**
Actinocyclus normanii	*Stephanodiscus niagarae*
Fragilaria capucina	
Fragilaria crotonensis	
Cryptophyceae	**Cryptophyceae**
Cryptomonas erosa	*Cryptomonas erosa*
Cryptomonas marssonii	*Cryptomonas marssonii*
Cryptomonas reflexa	*Cryptomonas reflexa*
Cryptomonas rostratiformis	*Cryptomonas rostratiformis*
Rhodomonas minuta	*Rhodomonas minuta*
Dinophyceae	**Dinophyceae**
Ceratium furcoides	*Ceratium furcoides*
Ceratium hirundenella	*Ceratium hirundenella*
Glenodinium spp	*Gymnodonium spp*
Gymnodinium helveticum	

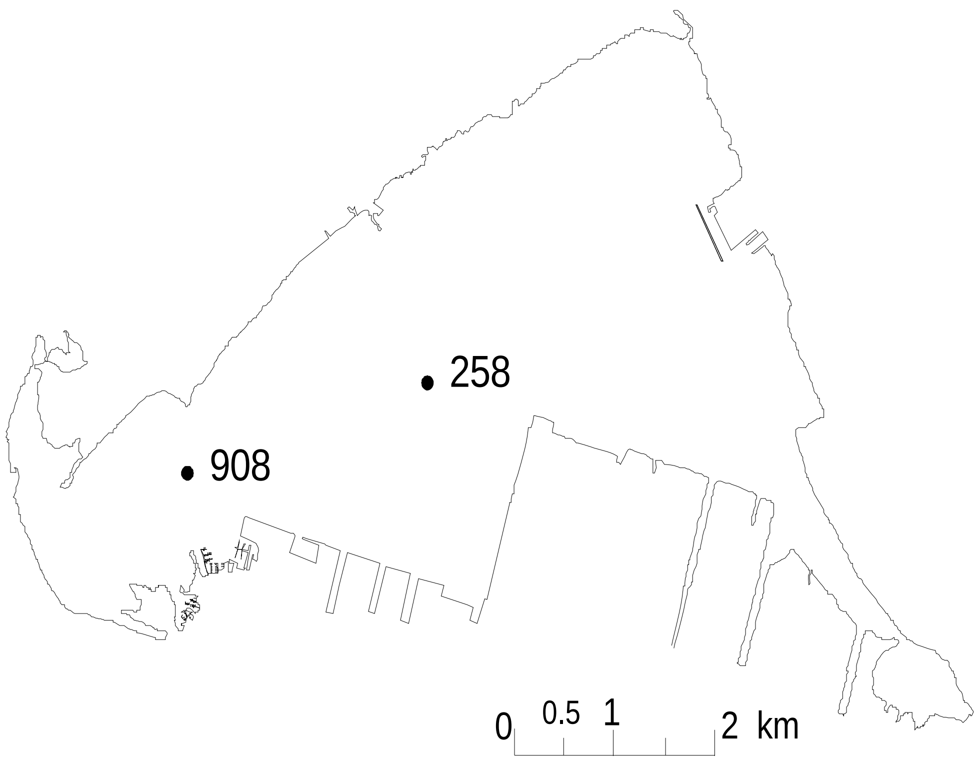

Fig. 1. Map of Hamilton Harbour, Lake Ontario showing planktonic sampling locations

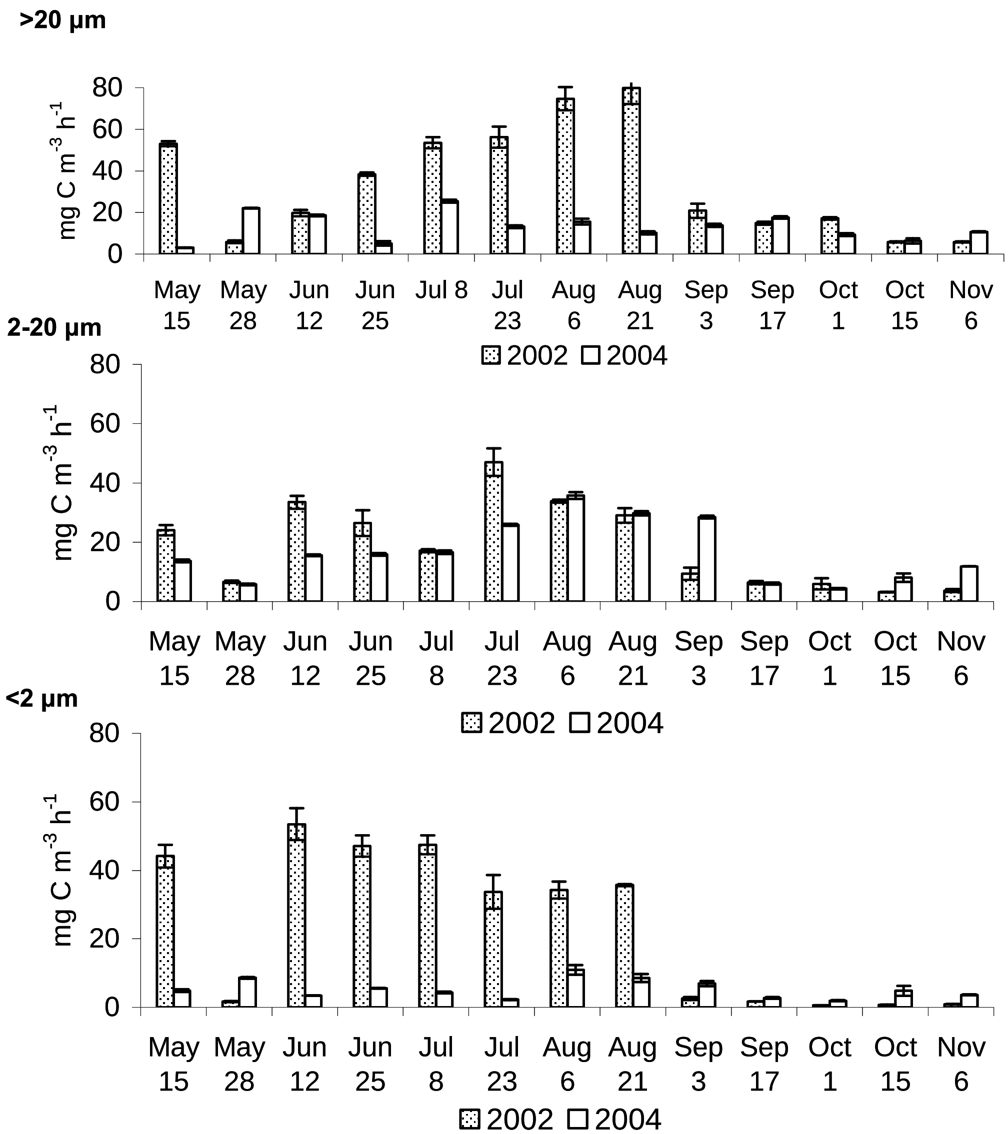

Fig. 2. Size fractionated primary productivity at Station 258 during 2002 and 2004.

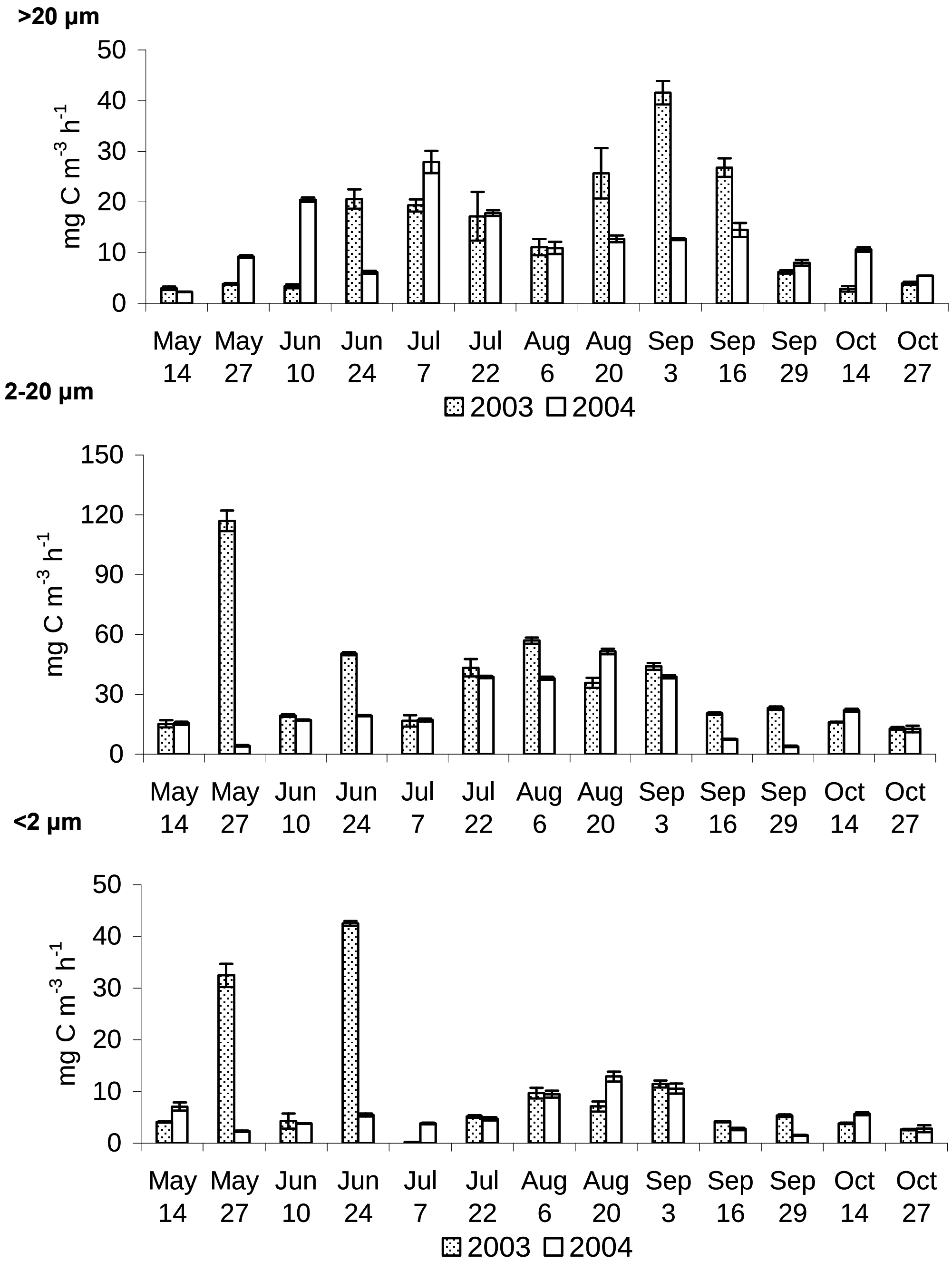

Fig. 3. Size fractionated primary productivity at Station 908 during 2003 and 2004.

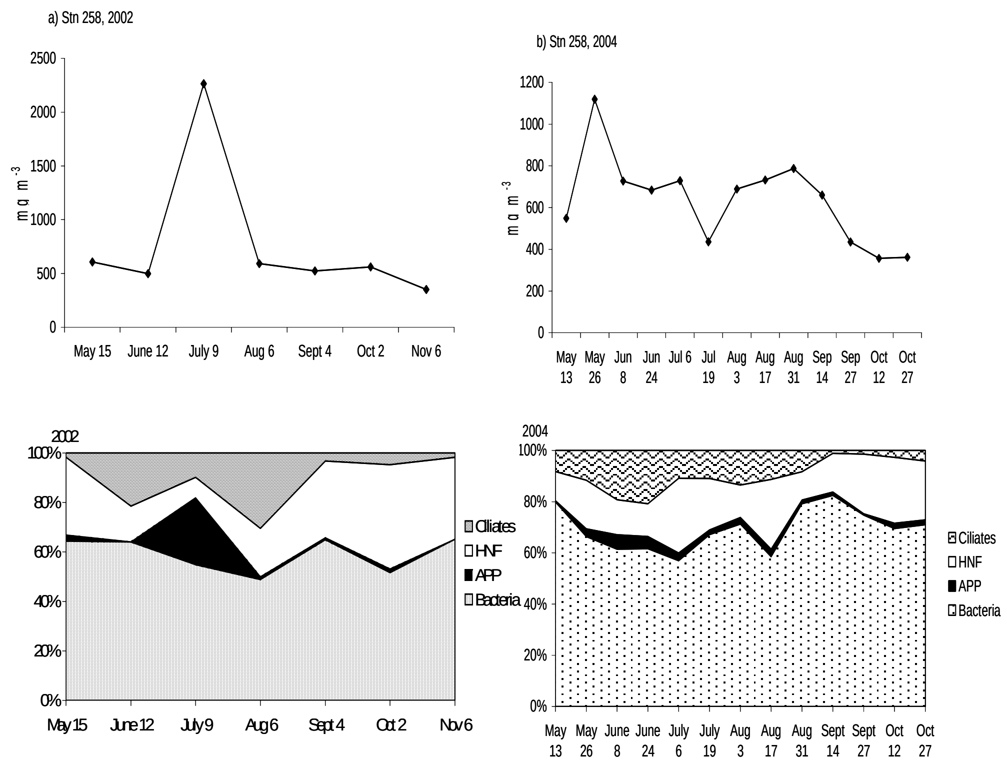

Fig. 4. Biomass and relative composition of microbial loop communities (Bacteria, Autotrophic Picoplankton (APP), Heterotrophic Nanoflagellates (HNF) and Ciliates at station 258 during a) 2002 and b) 2004.

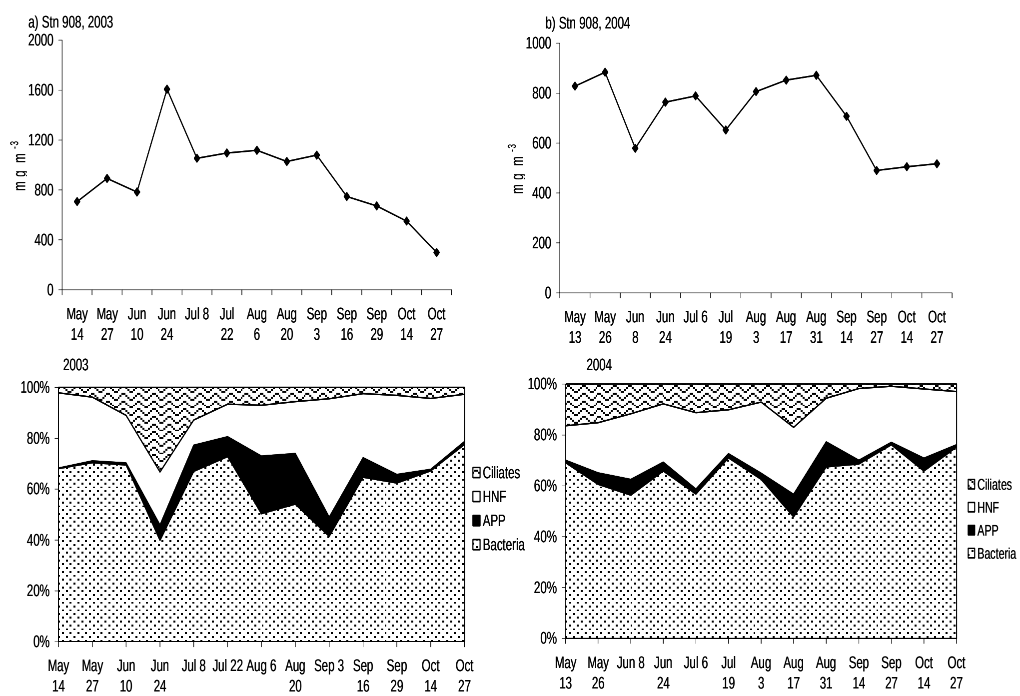

Figure 5. Biomass and relative composition of microbial loop communities (Bacteria, Autotrophic Picoplankton (APP), Heterotrophic Nanoflagellates (HNF) and Ciliates at station 908 during a) 2003 and b) 2004.

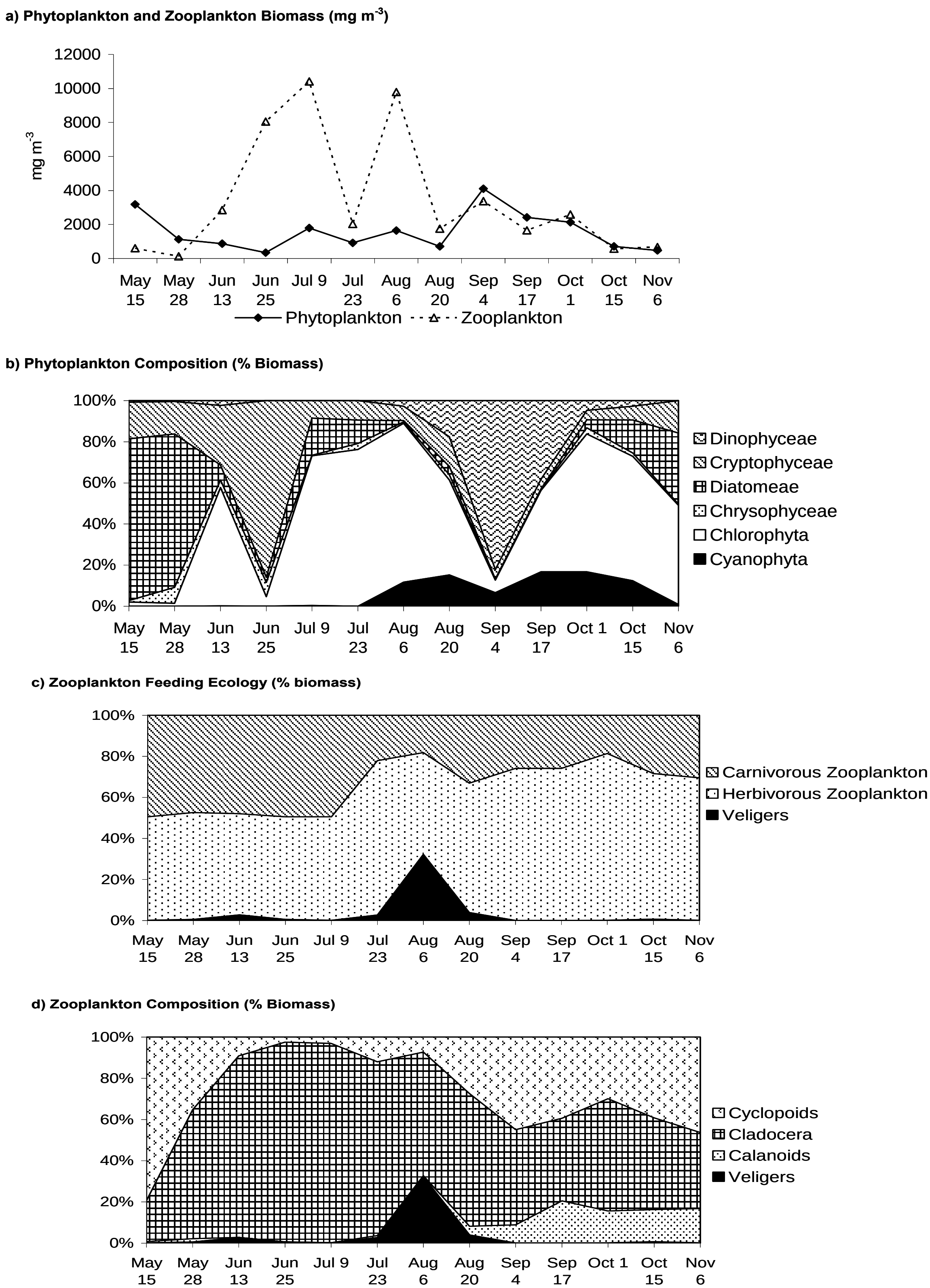

Fig. 6. A representation of the planktonic food web at station 258 during 2002 including a) Phytoplankton and Zooplankton Biomass, b) Phytoplankton composition, c) Zooplankton Feeding Ecology and d) zooplankton composition.

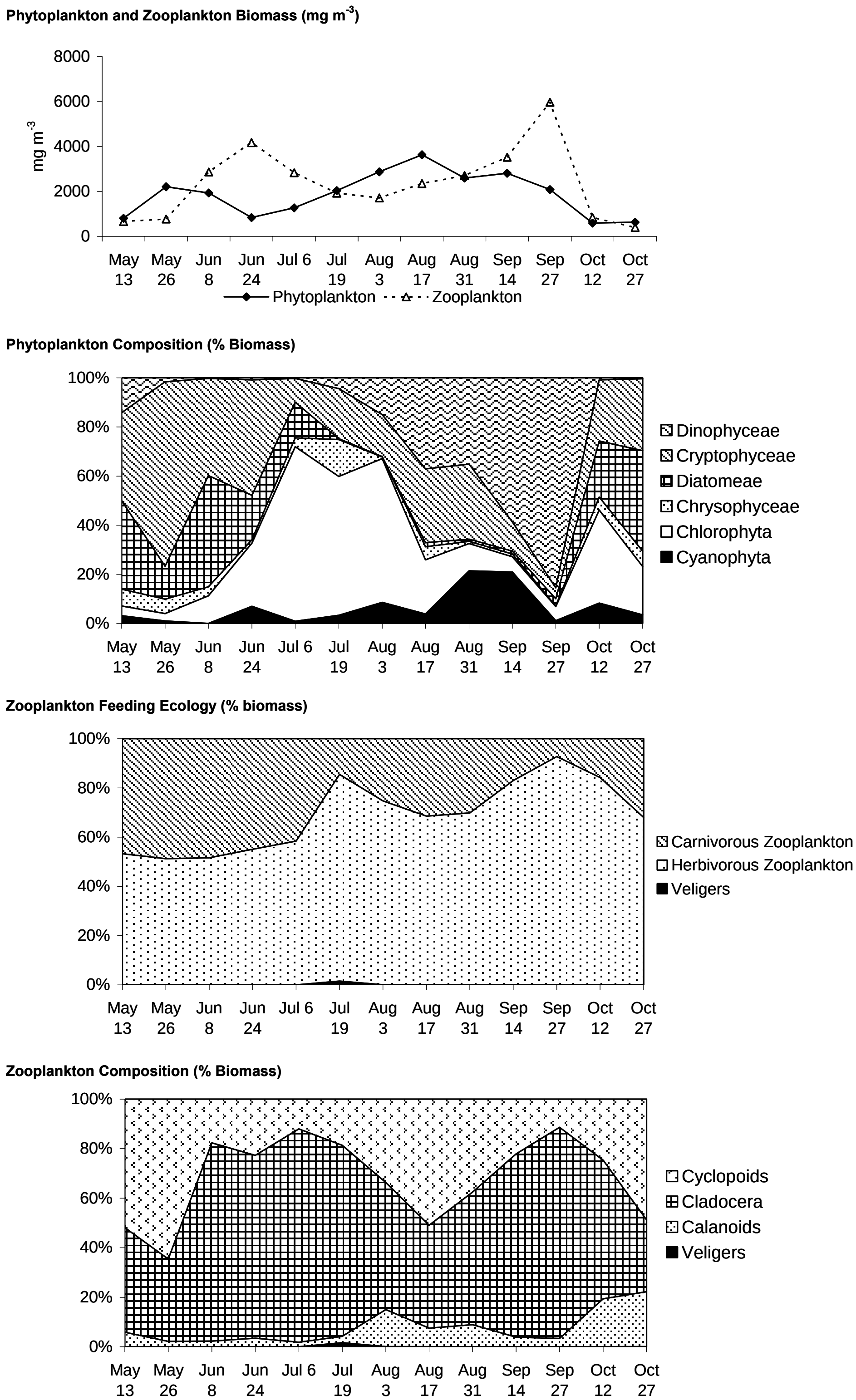

Fig. 7. A representation of the planktonic food web at station 258 during 2004 including a) Phytoplankton and Zooplankton Biomass, b) Phytoplankton composition, c) Zooplankton Feeding Ecology and d) zooplankton composition.

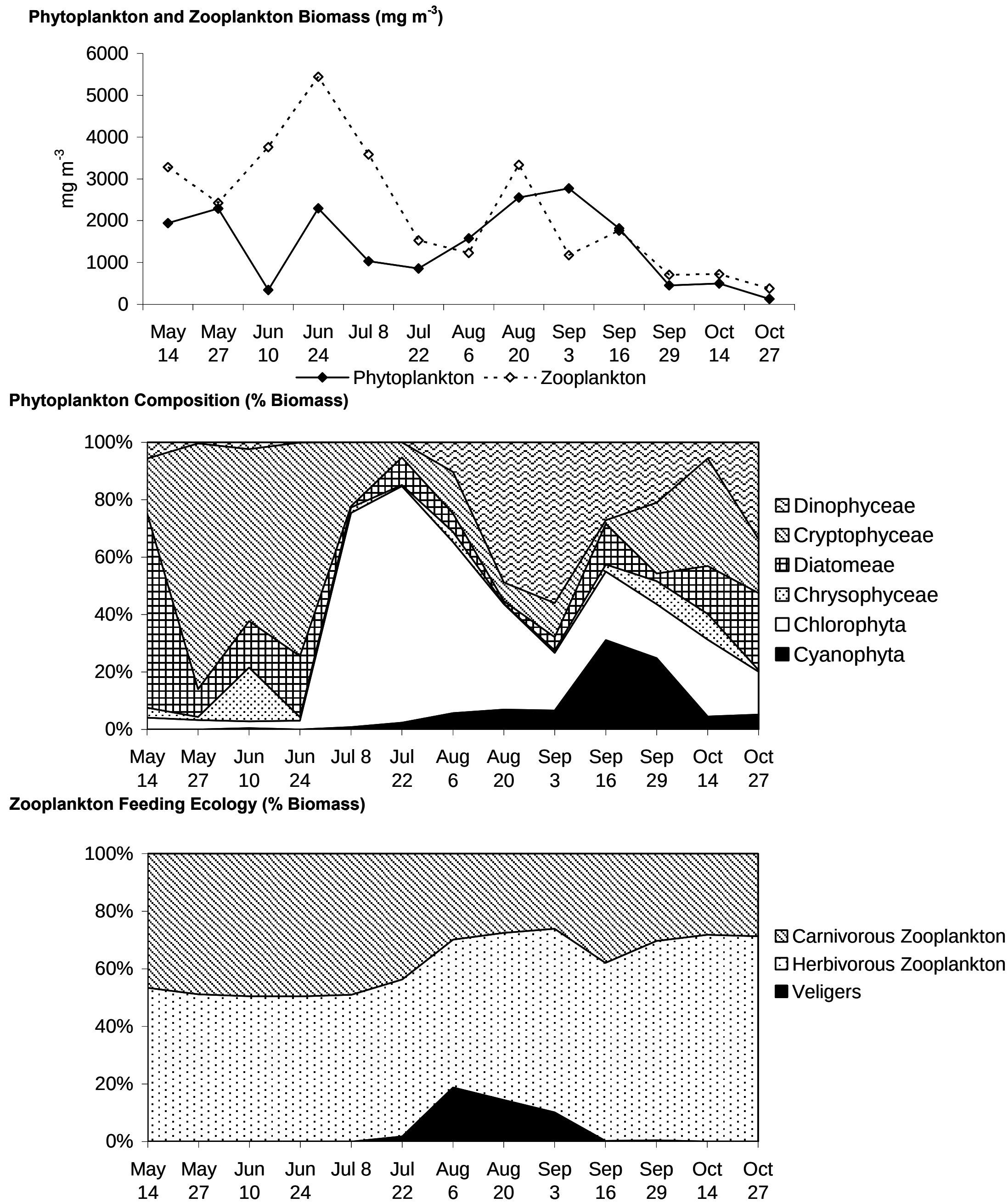

Fig. 8. A representation of the planktonic food web at station 908 during 2003 including a) Phytoplankton and Zooplankton Biomass, b) Phytoplankton composition, c) Zooplankton Feeding Ecology and d) zooplankton composition.

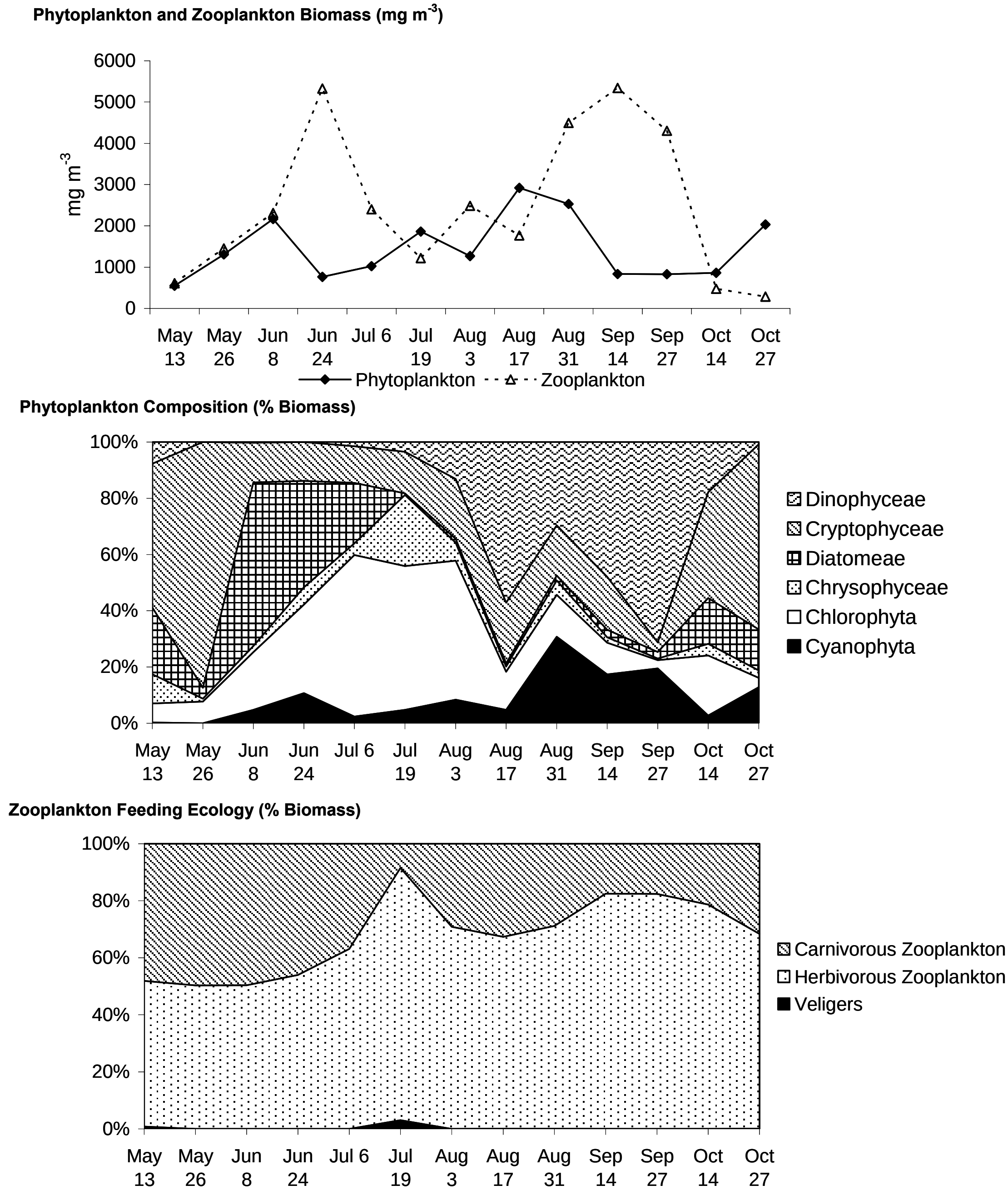

Fig. 9. A representation of the planktonic food web at station 908 during 2004 including a) Phytoplankton and Zooplankton Biomass, b) Phytoplankton composition, c) Zooplankton Feeding Ecology and d) zooplankton composition.

ZOOPLANKTON IN HAMILTON HARBOUR 2002-2004

Jocelyn Gerlofsma, Kelly Bowen and Ora Johannsson

INTRODUCTION

The Hamilton Harbour Remedial Action Plan (RAP) identified the degradation of zooplankton as one of the beneficial use impairments (Hamilton Harbour RAP 1992). It stated zooplankton abundance was high reflecting eutrophication and high productivity. Also, the mean size of zooplankton was small indicating heavy fish predation by a population dominated by planktivores.

During the 1990s, much effort has been spent to reduce phosphorus inputs to the Harbour in order to lower the level of eutrophication. Additionally, DFO and the RAP have improved fish habitat around the edge of the harbour and reduced carp breeding area by installing a fishway into Cootes Paradise for improvement of the native fish community. These efforts are expected to affect the zooplankton community. In order to assess the impact of these improvements on the aquatic ecosystem as a whole and reassess targets for the RAP, DFO has undertaken a long-term study of the Hamilton Harbour aquatic ecosystem starting in 2002.

This is an interim report examining zooplankton species composition, size structure, abundance, biomass and productivity in 2002, 2003 and 2004. The harbour is a large, hydrodynamically complex system with a range of habitats for zooplankton. The sampling stations were chosen to allow examination of the dominant spatial gradients specifically from inshore/shallow to offshore/deep and from regions near the major inflows in the west to the more mixed central basin.

METHODOLOGY

Zooplankton

Biweekly zooplankton sampling was carried out from the beginning of May to the end of October. From 2002 to 2004, five stations were examined with a different set of stations sampled each year (Table 1, Figure 1). All stations, except station 17, were sampled using a 41 litre Schindler-Patalas trap fitted with 64µm mesh. Samples were collected at discrete depths (Table 1) and preserved individually using a 4% sugar buffered formalin solution. For analysis, a single composite sample was constructed for each station-date by combining 50% of the sample from each depth.

The shallowest site, Station 17, was a 100m transect parallel to shore following the 1.5m depth contour, along which the ends and mid-point were sampled. At each of these three points, a Guzzler® diaphragm hand pump and 25mm diameter hose were used to collect 10L of water from 0.5m and 1.0m depths for a total of 60L. The samples were pooled and concentrated using a 64µm mesh net, and preserved as above.

A minimum of 400 individual zooplankters and all loose eggs within a subsample aliquot were enumerated from each sample. At least 100 individuals of each major group were included. *Cercopagis pengoi*, a predatory cladoceran which invaded Lake Ontario in 1998 (MacIsaac et al. 1999), cannot be accurately enumerated from subsamples because their hooked caudal spines become entangled, and form clumps. Therefore the larger organisms from each sample were captured on a 400-µm mesh and all the *C. Pengoi* were removed and counted.

Seasonally-weighted mean (SWM) abundance, biomass and production of zooplankton were calculated over the May 1 to October 31 sampling season. Lengths of cladocerans were measured from the top of the helmet to the base of the tail spine, copepods from the anterior end of the cephalothorax to the end of the caudal rami, and veligers across the widest section of the shell. Body mass (mg dry weight) was estimated from length-weight regressions from the literature, summarized in Johannsson et al. (2000). Regression equations for *Cercopagis* are given in Grigorovich et al. (2000). Production was estimated by the egg-ratio method of Paloheimo (1974) as described in Cooley et al. (1986) where cyclopoid or calanoid nauplii and copepods were assigned to species according to the relative abundance of the adults. When a species' seasonally-weighted mean biomass was <50 mg m^{-2}, production was estimated using P/B relationships as described in Johannsson et al. (2000). However in several cases (e.g. cyclopoid copepods), the egg-ratio production estimate was zero or close to zero, so the P/B production estimate was used despite a SWM biomass >50 mg m^{-2}. Based on the Bay of Quinte data, it appears that P/B values estimated from the literature relationships overestimate production. Therefore, the P/B production estimates were amended using the correction equations for Belleville in the Bay of Quinte (Bowen and Johannsson 2005) as follows:

Cladocerans:

$$\ln(\text{egg-ratio prod.}) = -0.301 + 1.026 \ln(\text{P/B prod.})$$

$$N = 19;\ r^2 = 0.89$$

Cyclopoids:

$$\ln(\text{egg-ratio prod.}) = -0.696 + 1.010 \ln(\text{P/B prod.})$$

$$N = 10;\ r^2 = 0.93$$

Rotifers

Samples were collected using a Guzzler® diaphragm hand pump and 16mm diameter hose. The hose was lowered to the bottom sample depth and then raised through the water column at a set rate (1 pump stroke per 0.5 m (stations 17, 908 and 258) or 1 stroke per 0.25 m (stations 6, WC) (Table 2). The residual volume of water held by the hose was accounted for during the sampling.

For preservation, 6 L of sample water was sieved through 20µm mesh, rotifers were narcotized with soda water and preserved with 4% sugared buffered formalin. For analysis a May to October seasonal composite for each station was made by pooling 50% of the sample from each date.

To enumerate and measure the rotifers a subsample was removed and placed in a Sedgewick-Rafter chamber at 100 x magnification. A minimum of 200 organisms were counted in a sample dominated by one to three species. For more diverse samples, a minimum of 400 individuals were counted. At least 20 individuals of each species were measured. For numerically dominant species, 50 or more individuals were measured. A maximum of 20% or 25% of the sample, by volume, was entirely analysed. Biomass, except for the *Polyarthra* species, was estimated according to the formula of Ruttner-Kolisko (in McCauley 1984). The formulae determined for Lake Erie *Polyarthra* were also applied in Hamilton Harbour (Johannsson et al. 2000). For *P. dolichoptera,* $v = 0.205a^3$ was used where v = volume in μ^3 (or wet body mass in µg x 10^{-6} assuming a density of one) and a = longest dimension in microns, while for *P. vulgaris*, *P. major* and *P. remata,* $v = 0.158a^3$.

RESULTS

Zooplankton Density

In 2002, the seasonally weighted mean (SWM) areal density ranged from 0.69 x10^6 indviduals·m^{-2} at the 1.5 m deep station to 7.66 x10^6 indviduals·m^{-2} at the deepest site, 24 m. There was an offshore to nearshore gradient with the average (volumetric) density increasing towards shallower depths (Table 3).

Species composition at the shallowest station (17) varied relative to the offshore stations (6 and 258) (Table 3). There were a total of 23 taxa at station 17 compared to 18 at station 6 and 19 at station 258. Herbivorous cladocerans were more diverse in the nearshore with more macrophyte-associated species (e.g. *Sida crystallina*, *Camptocercus rectirostris*) and benthic-associated species (e.g. *Alona sp*., *Eurycercus sp*.) and harpacticoid copepods (Balcer et al. 1984) (Table 3). The offshore stations, especially station 258, supported more *Eubosmina coregoni*, *Daphnia galeata mendotae*, *D. retrocurva* and *dreissenid veligers*. At all three stations *Bosmina longirostris* had the highest SWM areal density – 4.19 x10^6 indviduals·m^{-2}.

In 2003, both areal density and density increased towards the offshore, opposite to the pattern in 2002 where density decreased towards the offshore (Table 3). The densities in 2003 were also lower than even the density at the deepest site in 2002. A similar distribution of species occurred at both stations WC and 908 in 2003; 17 taxa were identified at each. *Bosmina* was the most abundant taxon, followed by cyclopoid naupli and copepodids, and veligers (Table 3). Most of the cladocerans, including *Bosmina, Eubosmina, D. retrocurva, D. galeata mendotae*, and *Ceriodaphnia lacustris* had lower densities than in 2002. Cyclopoid densities in 2003 were similar to those in 2002, but calanoid densities were lower. Over the three years studied, *Cercopagis pengoi* was most abundant in 2003.

Sampling in 2004 revisited the offshore stations 258 (24 m in depth) and 908 (14 m in depth). Both stations had lower densities compared to the prior years. Zooplankton at station 258 had a higher areal density but lower density (4.20 $x10^6$ indviduals m^{-2}, 0.193 $x10^6$ indviduals m^{-3}) than zooplankton at station 908 (2.71 $x10^6$ indviduals m^{-2}, 0.175 $x10^6$ indviduals m^{-3}) (Table 3). Station 908 had 18 taxa, and 258 had only 15. Although community composition at the two stations was similar in 2004, there were some differences compared to the previous years. *Bosmina* was still the most common taxon, but numbers were low. Veliger densities also fell drastically in 2004. Densities of other cladocerans varied year to year, depending on the station. The cyclopoids *Diacyclops thomasi* and *Mesocyclops edax* were higher in 2004, but densities of juvenile cyclopoids were intermediate.

Zooplankton Biomass

SWM volumetric biomass was higher in 2002 than 2004, as determined by comparing biomasses at station 258 between years, and was similar or slightly higher in 2004 than 2003, as determined by comparing biomasses at station 908 between years (Table 4).
In 2002, areal SWM biomass had a strong nearshore to offshore gradient, with an offshore biomass at station 258 10 times greater than the nearshore station (7682 mg m^{-2} vs. 748 mg m^{-2}; Table 4). More than 75% of the biomass was comprised of herbivorous cladocerans (bosminids, *Daphnia* and others; Table 4). *Bosmina* was the most dominant taxon at the offshore stations 258 and 6 (Figure 2). *Daphnia* was also important at stations 258 and 6, but not at 17 where other herbivorous cladocerans dominated. Copepod contribution was highest at 258 and lowest at 17 (Table 4). Cyclopoids composed 6.6% - 14.2% of the biomass and calanoids 1.2 % -3.0 % with the greatest percentage of copepods at 258 and the lowest at 17 (Table 4). Harpacticoids were noteworthy only at 17, contributing 2.5% of the biomass. Veliger biomass contribution was highest at station 258 (7.8%) compared to station 6 (2.4%) and station 17 (1.5%).

In 2003, SWM areal biomass again had an offshore to nearshore gradient with offshore station 908 substantially higher than nearshore station WC (3252 mg m^{-2} vs. 500 mg m^{-2}; Table 4). However, the community composition at the two stations was similar. Cladocerans made up 69% of the biomass; the main contributors being bosminids (54.8%) and *Daphnia* sp. (10.6%) (Table 4). Approximately 25% of the biomass came from cyclopiods.

In 2004, SWM areal biomass at station 908 (3452 mg m^{-2}) was similar to the previous year (Table 4). At station 258, the areal biomass was lower in 2004 (biomass = 5433 mg m^{-2}) than in 2002. Cladocerans were still dominant, but *Daphnia*, not bosminids, were the main contributors (Figure 2). Cylcopoid and calanoid biomass values were 25.5% and 5% respectively. Veliger biomass was very low in 2004 (0.1%) relative to the previous two years.

Zooplankton SWM areal biomass values were compared to stations from the Bay of Quinte over the same time period (Figure 2). Belleville (B) and Napanee (N) are both 5 m deep, unstratified stations located in the eutrophic upper bay. The 2002-2004 mean biomass values at B (527 mg m^{-2}) and N (653 mg m^{-2}) were substantially lower than the nearshore Hamilton Harbour station 6 (2357 mg m^{-2}) (Bowen and Johannsson 2006). Hay Bay (HB), located in the mesotrophic middle bay, is 12 m deep and somewhat comparable to the offshore harbour stations 908 and 258. Mean biomass was also lower at HB (2197 mg m^{-2}) than 908 or 258. Conway (32 m deep) is positioned at the mouth of the bay and represents meso-oligotrophic conditions. No shallow nearshore stations were sampled in the Bay of Quinte from 2002 to 2004 that were comparable to 17 and WC. However, mean biomass in a weedy embayment in the upper bay was 358 mg m^{-2} in 2001, a value substantially lower than station 17 (748 mg m^{-2}) in 2002 (Bowen et al. 2003).

Seasonal Biomass Trends

Seasonal biomass patterns varied at the different stations over the three years (Figures 3 and 4). However, *Bosmina* consistently peaked early in the season around the end of June, crashed and was then replaced by a small population of *E. coregoni* until the end of September or early October. *Daphnia* generally started appearing around mid-July when *Bosmina* began their decline. *D. retrocurva* peaked in August at stations 258, 908 and WC in 2002 and 2003. At the shallow stations in 2002 it did not develop large summer populations. In 2004, *D. retrocurva* started increasing earlier in July and remained relatively steady until it peaked in September. *D. retrocurva* biomass then dropped quickly as *D. galeata mendotae* peaked at the end of September. Predatory cladocerans comprised a very small portion of the community, but *Leptodora kindtii* had a strong presence at 908 in August 2003. *C. pengoi* generally had a low and short-lived presence just prior to *L. kindti*.

Cyclopoid biomass varied from year to year. In 2002, cyclopoids were most common from late summer to mid-autumn. In 2003, a spring cyclopoid bloom ended in late June. Cyclopoids remained at low levels until the end of October with only one small peak mid-September at 908. In 2004, cyclopoids had a strong presence in the zooplankton population from early spring until October. Generally, spring populations were composed of *D. thomasi* and later populations were mainly *M. edax.* Never dominant, calanoids generally appeared in late summer or early fall. Harpacticoids were only seen at the shallow station 17, where they formed a small peak in early October 2002.

Veligers also varied from year to year. A short bloom of veligers occurred in early August 2002. In 2003, they peaked sharply in late August at 908, and at WC from late August to mid-September. Veligers in 2004 were very low in abundance and made no significant contribution to biomass at any point in the season.

Zooplankton Production

Seasonal (May-October) areal production increased from inshore to offshore with station depth (Figure 5), and during our study, was highest at station 258 in 2004 (107.1 g m^{-2}) and lowest at station WC in 2003 (7.7 g m^{-2}). In 2003, seasonal production at the offshore station 908 (45.5 g m^{-2}) was lower than the offshore station 258 in 2002, but higher than the nearshore stations (6 and 17). In 2004, the production at station 908 (51.5 g m^{-2}) was similar to the previous year at the same station (Table 5, Figure 5). Production at station 258 (61.7 g m^{-2}) was lower than that found in 2002.

The relative contributions of the major zooplankton groups to total seasonal production were similar across stations within a year, but differed between years. In 2002, the majority of the production was from herbivorous cladocerans, ranging from 96% (32.9% bosminids, 1.3% *Daphnia* and 61.8% others) at station 17 to 81.5% at station 258 (Figure 5). At the stations 6 and 258, *Bosmina* and *Daphnia* contributed more than any other species to zooplankton production (Figure 6). Cyclopoid production ranged from 3.8 % at station 17 to 7.4 % at station 258, whereas calanoids represented less than 1% of the total (Table 5). Veliger production was high at 258 (10.4%) compared to stations 6 (3.1%) and 17 (1.4%).

In 2003, *Bosmina* was again dominant, representing 67% of production at WC and 57% at 908. Predatory cladocerans, mainly *L. kindtii,* were noteworthy at Station 908 (3.4%). Cyclopoids contributed proportionately more to the production (13.3-16.9 %) than in 2002, but the total production (average=3.7 g m^{-2}) was similar to 2002 (Table 5). Calanoids production was very low (average = 0.06 g m^{-2}), adding less than 1 % of the total. Veliger production contributed more to the nearshore (WC, 6.4%) than to offshore (908, 3.8%).

In 2004, the percent contribution of the different zooplankton groups was similar between the two stations, 258 and 908 (Table 5, Figure 6), with the areal production higher at station 258. In contrast to previous years, cladoceran production at both stations was dominated by *Daphnia* (56%), not *Bosmina* (24%). Also, veligers crashed in 2004 with only 76 mg m^{-2} production at station 908 and 127 mg m^{-2} at 258.

Cladoceran Length

Small bosminids dominated the system in 2002 and 2003, resulting in a consistently low mean cladoceran length ranging from 313µm to 349µm (Figure 7). The higher relative abundance of *Daphnia* increased the cladoceran mean size the following year (423µm to 430µm). Compared to the Bay of Quinte, harbour cladocerans tended to be smaller in 2002 and 2003, but larger or similar in size in 2004.

Oxygen and Zooplankton

At station 258 in 2002, zooplankton samples taken from various depths in the water column were analyzed discretely on four different dates. These depths were chosen to represent a gradient in dissolved oxygen, and to show how zooplankton biomass changed when the oxygen dropped to hypoxic (1-4 mg l^{-1}) and near-anoxic (<1 mg l^{-1}) levels. On July 9th and 23rd, the oxygen levels gradually decreased from > 8 mg l^{-1} at the surface to < 1 mg l^{-1} at the bottom of the metalimnion (7-9 m deep) (Figure 8). In the epilimnion and at the top of the metalimnion, zooplankton biomass was high (July 9: 420-479 mg l^{-1} and July 23: 230-360 mg l^{-1}), but dropped (July 9: 162 mg l^{-1} and July 23: 160 mg l^{-1}) near the bottom of the metalimnion where the oxygen levels fell below 1 mg l^{-1}. Zooplankton biomass remained low in the hypolimnetic samples (July 9: 130 mg l^{-1} and July 23: 191 mg l^{-1}). The hypolimnetic oxygen increased to 2-3 mg L^{-1} between 11-14 m and then decreased back down to about 1 mg l^{-1}. In September, the oxygen profile indicated that oxygen remained high in the epilimnion (Sept. 4: 10.5 mg l^{-1} and Sept. 17: 7 mg l^{-1}), dropped rapidly over the 4-5 m of the metalimnion to <0.5 mg l^{-1} (11-13 m deep) and remained there through the hypolimnion. Zooplankton biomass was the highest (Sept. 4: 529 mg l^{-1} and Sept. 17: 159 mg l^{-1}) in the metalimnion at oxygen concentrations of 6 mg l^{-1} to 0.5 mg l^{-1}. Zooplankton biomass in the hypolimnon was higher than in the epilimnion, even with low oxygen levels, on both dates (Figure 8). On all four dates the percent composition of the major zooplankton groups did not vary with depth, except for *C. pengoi.* It was only in one sample - July 23 at 5 m.

Rotifers

Each year, a single seasonal composite sample was analyzed for rotifers at each station. A total of 27 rotifer taxa were identified in Hamilton Harbour between 2002 and 2004, most to the species level. Ten families were represented. Between 15 and 18 taxa were found at each station in 2002 and 2004, and between 12 and 14 taxa in 2003 (Table 6). Seven taxa were found in all seven samples, whereas 5 taxa were found in only one sample.

In 2002, rotifer areal and volumetric density increased from the nearshore to the offshore (Table 6). In 2003 and 2004, areal density again increased, but volumetric density decreased along this nearshore offshore gradient. In 2003, rotifer areal density at nearshore WC ($2.54 \times 10^5 m^{-2}$) was higher than the nearshore stations 17 and 6 in 2002. In 2004, when stations 908 and 258 were re-sampled, densities were lower than the previous years.

Numerically, *Keratella cochlearis* was the most dominant taxon at all stations, where it comprised between 52 and 77% of the total rotifer community. Another dominant species was *Polyarthra dolychoptera* (4-17% of the total). These are both fairly small rotifers, averaging 114 and 87 µm, respectively. Several taxa were more abundant at the shallow macrophtye station 17, including *Filinia terminalis*, *Lecane spp*. and *Trichocerca porcellus*, whereas *Kellicottia bostoniensis* and *Pompholyx sulcata* tended to be associated with the offshore (Table 6).

Although seasonal mean rotifer and zooplankton density values were in the same range, mean rotifer biomass was generally only 1-3% that of zooplankton biomass because of their small size. One exception occurred at the WC station in 2003, where rotifer biomass was 9% of the SWM zooplankton biomass. During the three years, the areal biomass was highest at the deep stations and lowest at the shallow stations, ranging between 8.1 mg m^{-2} at Station 17 and 136.5 mg m^{-2} at Station 258 in 2002 (Table 6; Figure 8). Values between 2003 and 2004 at station 908 were similar (82.2 mg m^{-2} in 2003; 78.6 mg m^{-2} in 2004), but there was a drop in the rotifer biomass from 2002 to 2004 at station 258 (136.6 mg m^{-2} in 2002; 85.4 mg m^{-2} in 2004). Rotifer biomass in Hamilton Harbour was generally two to three times greater than in the Bay of Quinte over this same time period (Figure 8).

Due to its relatively large size (390 µm), *Asplanchna priodonta* was generally the most dominant rotifer by biomass in the harbour. This taxon usually comprised between 43.9 and 61.8% of total biomass at most stations (Figure 8). One exception was 908 in 2004, which supported a more diverse rotifer assemblage. Other dominant rotifer genera by biomass were *Keratella* (10.2%-30.4%), *Polyarthra* (2.8%-35.6%) and *Trichocera* (0.6%-19.2%). In the Bay of Quinte, *Polyarthra*, followed by *Asplanchna* and *Trichocera* were the most dominant rotifers.

DISCUSSION

Hamilton Harbour remains a polluted, eutrophic body of water, although much improved through the remediation actions undertaken over the past thirty years. Total phosphorus has declined from 54 µg l^{-1} to 33 µg l^{-1}, chlorophyll *a* from 33 µg l^{-1} to 11.4 µg l^{-1} and Secchi depth has increased from 1.5 m to 2.4 m (MOE 1981, Painter et al. 1990, Burley *this volume*). Near-anoxic conditions (O_2 < 1 mg l^{-1}) in the hypoliminion remain a problem, but are less persistent than in the 1970s (MOE 1981, Burley *this volume*). Has the zooplankton community responded to these improvements?

Little historical information is available on the zooplankton of Hamilton Harbour. The first studies were conducted by Harris (1976), Piccinin (1977) and Piccinin and Harris (1980) during the 1975-1979 period at the height of eutrophication, before remediation commenced. A brief study was also conducted in 1990 when Koenig (1992) examined copper and cadmium levels in Hamilton Harbour plankton. These studies provide some background with which to compare the current status of the zooplankton community. The zooplankton community in Hamilton Harbour in 2002-2004 can also be compared to the upper Bay of Quinte (2002-2004: TP = 31-46 µg l^{-1}), another eutrophic embayment in Lake Ontario (Nicholls and Millard 2006). The present study also provides an opportunity to look at inter-annual variation and inshore-offshore gradients in zooplankton community dynamics.

The zooplankton community in the late 1970s was primarily dominated by large rotifers and the cladoceran *Bosmina longirostris*. There were very few other cladoceran species and only two copepod species (Harris 1976, Piccinin 1977, Piccinin and Harris 1980). The dominant rotifers were *Keratella quadrata*, *Brachionus angularis*, *Filinia terminalis* and *Trichocera cylindrical*. These species are all indicative of eutrophic conditions in the Great

Lakes (Gannon and Stemberger 1978). Unfortunately the studies in 1975-1979 do not provide abundance or biomass data comparable to the 2002 -2004 data. In 1990, the zooplankton community was still dominated by cladocerans and rotifers. Cladocerans included *Bosmina* (86% biomass), *Daphnia* sp. (12%) and *Leptodora kindtii* (2%) (Koenig 1992). The dominant rotifers were *Polyarthra sp., Synchaeta, Keratella and Pompholyx.* Although it was one of the most abundant species in the 1970s, only a few *B. angularis* were present in August 1990. Shifts in composition have continued into the early 2000s. Numerically, the dominant species was still a rotifer, *K. cochlearis*, followed by a cladoceran *B. longirostris*, but the presence of *Daphnia sp*. and copepods continued to grow (Table 3 & 6). The rotifers *B. angularis* and *T. cylindrical* were not present at any of the stations in the 2000s, and *F. terminalis* was found in very low densities at station 258 in 2004 (Table 6). The present zooplankton community still indicates that Hamilton Harbour is eutrophic. Higher abundances of cladocerans and cyclopoids compared to calanoids is a good indicator of eutrophic waters (Gannon and Stemberger 1978). Also, Patalas (1972) found that zooplankton communities dominated by *B. longirostris, E. coregoni, D. retrocurva, D. galeata mendotae, Mesocyclops edax, Diacyclops thomasi* indicated more eutrophic conditions in the Laurentian Great Lakes, and these are all present in high numbers in Hamilton Harbour. The decline in rotifers may be associated partly with the decline in total phosphorous; however increased predation by the growing populations of copepods, and competition and interference mortality associated with the increasing presence of *Daphnia* spp. are also likely involved (Gilbert 1988, MacIsaac and Gilbert 1991). The change to a zooplankton community less dominated by rotifers and with higher biodiversity reflects an improvement in Hamilton Harbour waters, its foodweb structure, and energy flow to higher trophic levels.

In the 1970s anoxic and near anoxic ($O_2 < 1$ mg l^{-1}) conditions in Hamilton Harbour's hypolimnion limited most of the zooplankton biomass to the epilimnion from June to September. There was even one occurrence in 1979 where a stable period in July caused anoxic conditions to rise within 4 m of the surface (Piccinin and Harris 1980). *Bosmina*, the most abundant zooplankton, disappeared from the hypolimnion when near-anoxia to anoxia occurred, but rotifers were still present (Harris 1976, Piccinin and Harris 1980). Bottom aeration experiments during the late 1970s increased the zooplankton biomass of *Bosmina* and *Filinia* sp. and their presence was observed in the hypolimnetic waters (Harris 1976, Piccinin 1977). Hypolimnetic hypoxia was still a problem during this present study (mean hypolimnetic for 2002-2004 = 1.3 mg l^{-1} to 1.6 mg l^{-1}; Burley *this volume*). In 2002, over the stratification period the mean hypolimnetic oxygen was 1.3 mg l^{-1}. There were four occurrences of mean hypolimnetic oxygen levels below 1.0 mg l^{-1} from August to early October, whereas in the 1970s the hypolimnion was persistently anoxic or near-anoxic (O_2 = 0-1 mg l^{-1}) from June to September (Harris 1976, MOE 1981, Burley *this volume*). The discrete depth zooplankton samples from this study showed that zooplankton biomass (200 mg l^{-1} – 500 mg l^{-1}) was quite strong in the metalimnion even where oxygen dropped to <1 mg l^{-1} (Figure 8). Zooplankton also occupied the hypolimnion ranging in biomass from 94-291 mg l^{-1} on the 4 dates measured. In July, elevated oxygen in the hyplomnion indicated the water came in from Lake Ontario, possibly bringing the zooplankton with it (Hamblin and He 2003). Yet, in September the oxygen was < 1 mg l^{-1} over the whole hypolimnion (Figure 8). Zooplankton may migrate into the lower oxygen areas to avoid fish predation. In Irondequoit Harbour, a refuge from planktivorous fish was created in the metalimnion by bring the oxygen levels to no higher than 2 mg l^{-1} using hypolimnetic oxygen injection (Klumb et al. 2004). Some zooplankton such as *D. galeata mendotae*, are tolerant of low oxygen levels and can migrate to layers of low oxygen to avoid fish (Heberger and Reynolds 1977).

Strong predation of zooplankton by fish is a concern in the harbour as indicated by the low mean cladoceran length (Figure 7). The mean length of cladocerans is interpreted as an indicator of the level of planktivory as fish preferentially consume larger zooplankton individuals (Cooley et al. 1986). Mills et al. (1987) have proposed that mean zooplankton community length can serve as an indicator of balance between piscivores and planktivores within the fish community. The same should be true of cladocerans and a single group may provide a more consistent measure as it is not affected by the relative abundance of copepods to cladocerans which are very different in shape. However, as of yet, no optimum mean size of cladocerans has been determined to best reflect the status of the fish community. The high chlorophyll *a*/total phosphorus (CHLa/TP) ratios in the harbour (0.41 to 0.62 during the present study) confirm that the pelagia in the harbour is an 'odd-linked' system dominated by planktivores (Mazumder 1994, Dahl et al. 1995, MacDougall et al. 2001, Nicholls and Millard 2004). Cooley et al. (1986) indicated that high planktivory might mask improvements in eutrophic waters, since predation keeps larger zooplankton taxa from becoming dominant. Larger cladocerans can strongly depress algal abundance, which is an indicator of trophic condition and contributes to the CHLa/TP ratio. In many systems, dreissenids usurp a significant proportion of pelagic productivity and route it through the benthos (e.g. Johannsson et al. 2000, Johannsson and Nicholls 2003). Hamilton Harbour is one of the few shallow Great Lakes systems where dreissenids are not abundant (Dermott and Bonnell *this volume*), and the CHLa/TP ratio is still an indicator of the relationship between plankton and fish. In the Harbour, cladoceran mean length ranged from 320-425 µm over this three year study (Figure 6). These measures tended to be lower than those in the Bay of Quinte over the same time period, with the exception of 2004 when *Daphnia* were abundant (Figure 4). Overall, planktivory can still be considered high in the harbour. The harbour is a nursery area for many young fish and high levels of planktivory might be expected to be the norm, not an impairment. Food web models can help to define optimum zooplankton composition and cladoceran size in the harbour.

In 2002 and 2003, the zooplankton community was examined for inshore to offshore trends. In both years, based on areal measurements, there was an inshore to offshore gradient with the greatest density, biomass and production of zooplankton occurring in the offshore. There was also a distinctive zooplankton community at the shallowest station, 17 (1.5 m deep). In the spring, the zooplankton community at this shallow station was similar to that at the offshore sites (6 and 258) with a high abundance of *Bosmina* and very few other species (Figure 2). As the submergent macrophytes began growing in early summer, the community started to include many benthic-associated species (e.g. *Alona* sp., *Eurycercus* sp. and harpacticoids (Balcer et al. 1984)) and macrophyte-associated species (e.g. *Sida crystallina* (Fairchild 1981)), rarely seen in the offshore.

The two offshore stations 908 and 258 were each visited in two years, and both sampled in 2004. This allowed for a comparison between the deeper, mid-harbour region which is strongly influenced by incursions of water from Lake Ontario (station 258), and the shallower flats in the western end of the harbour which are closer to the major natural inputs (station 908). Zooplankton areal density, biomass and production were about 1.5 times higher at station 258 than 908. Thus the deeper hypolimnion is supporting a larger zooplankton community, although the seasonal areal phytoplankton production at the two sites is similar (258 = 190 g C m^{-2} and 908 = 215 g C m^{-2}). There are at least two possible reasons for this observation and both may be operating. First, it may suggest that the shorter hypolimnion at station 908 does not provide as good protection from predation, both vertebrate and invertebrate, as does the deeper hypolimnion at station 258. Second, it may suggest that the micro- and macro-zooplankton at 258 are more effectively recycling the settling organic material as it falls through the hypolimnion because it remains in the water column for a longer period of time, and therefore, the zooplankton in the hypolimnion can live and metabolize at these low oxygen concentrations. The zooplankton data provide less support for the former than the latter. Cladoceran mean length was very similar between the two stations and zooplankton community composition was also very similar. These two observations indicate that predation did not favour one station over the other. If that is true, then zooplankton are effectively using hypolimnetic food sources despite the lower oxygen levels.

The two deep water stations also give us the opportunity to start assessing the degree of inter-annual variability in zooplankton community structure, biomass and productivity. The main differences occurred between 2004 and 2002 at station 258. Although *Daphnia* were more abundant in 2004, which can be associated with an increase in zooplankton biomass and productivity (Johannsson et al. 2000, Johannsson and Nicholls 2003), the relative proportion of cladocerans in the population decreased while the relative proportion of cyclopoids increased. This resulted in biomass and production levels 45% and 42% lower in 2004 than in 2002, respectively. The Harbour is known for its high spatial and temporal variability in water movement, and temperature and oxygen patterns. Harris (1976) found significantly different seasonal trends in zooplankton and phytoplankton between the west and central regions of the Harbour in the mid 1970s. Therefore, we were surprised to find conformity in community structure across the harbour within all three years. This suggests that some of the source promoting the biological gradient is either annually variable or has changed – a subject that needs further consideration.

In summary, the zooplankton of Hamilton Harbour still reflects a highly productive and eutrophic system that is being dominated by planktivores. Compared to the mid-1970s the zooplankton community has changed to one less dominated by rotifers and with higher biodiversity which indicate substantial improvement. Overall, the Hamilton Harbour zooplankton community is quite dynamic, varying in composition from the very nearshore to the inshore, and in biomass and productivity from the inshore to the offshore. Thankfully, from a monitoring perspective, greater variability was observed between years than within years. There are many influences on the zooplankton community including a variety of water inflows -creeks, sewage treatment plants and Lake Ontario, the presence of reed and macrophyte beds, and an abundant and relatively diverse fish community to list a few. Monitoring of the zooplankton along with the other lower trophic levels (microbial loop, phytoplankton and benthos) and fish should continue for a better understanding of this dynamic system and allow for modelling efforts to better define optimal conditions.

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Table 1. Hamilton Harbour zooplankton stations and sample depths for 2002-2004.

Station	Year(s) Sampled	Sample Depths (m)	Station Depth (m)
17	2002	0.5, 1	1.5
6	2002	1, 3, 5	6.0
WC	2003	0.5, 1.5, 2.5	3.5
908	2003, 2004	1, 3, 5, 7, 9, 11, 13	14
258	2002, 2004	1, 3, 5, 7, 9, 11, 15, 19, 22	24

Table 2. Hamilton Harbour rotifer sampling depth and pump interval for 2002-2004.

Station	Sample Depth (m)	Sample interval depth (m)	Hose length (m)
17	0-1	0.50	3
6	0-5	0.25	6
WC	0-2.5	0.25	6
908	0-13	0.50	24
258	0-22	0.50	24

Table 3: Seasonal mean densities (individuals·m^{-2}) of zooplankton taxa in Hamilton Harbour from 2002-2004.

		2002			**2003**		**2004**	
Species	**Group**	**HH17**	**HH6**	**HH258**	**HHWC**	**HH908**	**HH908**	**HH258**
Bosmina longirostris	Herb. Clad	363 841	1 682 367	4 191 419	340 169	1 946 628	1 091 014	1 658 812
Eubosmina coregoni	Herb. Clad	1 655	92 232	568 871	10 528	130 580	162 692	158 073
Daphnia retrocurva	Herb. Clad	1 648	83 905	336 719	21 712	93 062	203 274	319 271
Daphnis galeata mendotae	Herb. Clad	880	30 593	61 340	3 696	22 187	72 359	129 275
Other *Daphnia sp.*	Herb. Clad	0	366	0	0	0	9 761	1 524
Ceriodaphnia lacustris	Herb. Clad	122 283	2 387	10 509	219	337	0	0
Ceriodaphnia sp.	Herb. Clad	245	0	0	0	0	147	0
Chydorus sphaericus	Herb. Clad	2 834	21 007	134 047	15 769	73 651	28 029	34 833
Diaphanosoma birgei	Herb. Clad	15 126	5 970	14 136	338	3 365	18 903	25 522
Eurycercus sp.	Herb. Clad	10 695	0	0	0	0	0	0
Sida crystallina	Herb. Clad	1 418	0	0	0	0	0	0
Camptocercus rectirostris	Herb. Clad	9 122	0	11	0	0	0	0
Ilyocryptus spinifer	Herb. Clad	0	0	0	23	0	0	0
Alona sp.	Herb. Clad	24 075	124	2 632	73	616	155	0
Chydorus piger	Herb. Clad	0	120	0	0	0	0	0
Leptodora kindtii	Pred. Clad	12	1 169	1 638	56	3 187	1 106	1 696
Polyphemus pediculus	Pred. Clad	44	0	0	0	0	0	0
Cercopagis pengoi	Pred. Clad	4	300	498	204	2 371	318	305
Diacyclops thomasi	Cyclopoid	44	2 482	14 995	2 945	23 239	44 235	34 417
Cyclops vernalis	Cyclopoid	1 058	879	7 225	824	1 903	614	2 914
Tropocyclops extensus	Cyclopoid	0	0	0	0	0	102	338
Mesocyclops edax	Cyclopoid	44	3 825	44 318	886	9 264	51 754	80 143
Eucyclops agilis	Cyclopoid	4 688	0	89	0	0	517	906
Cyclopoida nauplii	Cyclopoid	31 808	105 333	473 815	155 911	735 364	480 354	824 356
Cyclopoida copepodids	Cyclopoid	25 142	86 810	408 194	58 054	387 872	423 185	716 204
Skistodiaptomus oregonensis	Calanoid	0	56	671	0	32	78	102
Leptodiaptomus siciloides	Calanoid	0	3 138	21 983	165	3 016	17 861	37 280
Eurytemora affinis	Calanoid	321	0	0	0	0	0	0
Calanoida nauplii	Calanoid	2 483	34 582	154 304	9 330	42 779	57 499	106 647
Calanoida copepodid	Calanoid	3 676	9 796	30 283	1 909	8 813	25 457	42 363
Harpacticoida nauplii	Harp.	3 706	0	0	37	299	0	0
Harpacticoida adults	Harp.	35 934	56	18	89	713	149	0
Dreissenia veligers	Dreissenid	20 996	122 210	1 183 954	69 801	234 334	15 505	28 445
Total (individuals·m^{-2})		**683 779**	**2 289 708**	**7 661 668**	**692 740**	**3 723 610**	**2 705 067**	**4 203 428**
Total (individuals·m^{-3})		**455 853**	**381 618**	**319 236**	**197 926**	**265 972**	**193 219**	**175 143**

Table 4. Percent distribution of zooplankton seasonal biomass (May 1- October 31, 2002-2004) amongst the taxonomic groups in Hamilton Harbour.

	2002			**2003**		**2004**	
	17	6	258	WC	908	908	258
Cladocerans	88.2	87.7	75.0	69.0	69.0	69.3	69.5
Herbivorous	88.2	87.2	74.6	68.3	66.4	68.3	68.5
Carnivorous	0.0	0.5	0.3	0.7	2.6	1.0	0.9
Cyclopoids	6.6	7.8	14.2	25.6	26.8	26.3	25.1
Calanoids	1.2	2.1	3.0	0.0	1.3	4.2	5.3
Harpacticoids	2.5	0.0	0.0	0.0	0.0	0.0	0.0
Veligers	1.5	2.4	7.8	5.4	2.9	0.1	0.1
Total (mg m^{-3})	**498.8**	**351.8**	**331.1**	**142.8**	**232.3**	**247.0**	**234.2**
Total (mg m^{-2})	**748**	**2 357**	**7 682**	**500**	**3 252**	**3 458**	**5 433**

Table 5. Percent distribution of zooplankton seasonal biomass (May 1- October 31, 2002-2004) amongst the taxonomic groups in Hamilton Harbour.

	2002			**2003**		**2004**	
	17	6	258	WC	908	908	258
Cladocerans	96.0	90.0	81.5	76.7	82.7	84.6	82.5
Herbivorous	95.9	89.3	81.0	75.4	79.3	83.3	81.3
Carnivorous	0.0	0.6	0.5	1.2	3.4	1.4	1.2
Cyclopoids	3.8	6.3	7.4	16.9	13.3	14.4	16.1
Calanoids	0.2	0.6	0.5	0.1	0.2	0.8	1.2
Veligers	1.4	3.1	10.6	6.4	3.8	0.1	0.2
Total (mg m^{-3})	**10 334**	**4 934**	**4 462**	**2 207**	**3 250**	**3 676**	**2 569**
Total (mg m^{-2})	**15 501**	**29 602**	**107 100**	**7 724**	**45 493**	**51 462**	**61 658**

Table 6. Rotifer total biomass (areal and volumetric), total number of taxa and density (indivuduals·m^{-2}) of common rotifers in Hamilton Harbour

	2002			2003		2004	
Taxa Name	**HH17**	**HH6**	**HH258**	**HHWC**	**HH908**	**HH908**	**HH258**
Areal Biomass (mg·m^{-2})	8.06	64.59	136.56	44.62	82.23	78.59	85.36
Volumetric Biomass (mg·m^{-3})	5.37	10.76	5.69	12.75	5.87	5.61	3.56
Total No. Taxa	18	16	15	12	14	16	17
Density of Common Taxa (ind·m^{-2})							
Asplanchna priodonta	6 154	61 538	164 103	35 897	47 863	11 966	81 239
Conochilus unicornis	3 077	30 769	41 026	35 897	79 772	35 897	54 159
Filinia terminalis	10 769	0	0	21 538	0	0	27 080
Kellicottia bostoniensis	0	7 692	287 179	0	0	0	108 319
Kellicottia longispina	0		82 051	0	0	107 692	54 159
Keratella cochlearis	166 154	1230 769	6 358 974	1 550 769	4 371 510	2 811 966	4 292 122
Keratella cochlearis tecta	3 077	23 077	164 103	57 436	159 544	119 658	54 159
Keratella quadrata	4 615	61 538	451 282	57 436	207 407	215 385	67 699
Lecane spp	15 385	0	0	0	0	0	0
Ploesoma hudsonii	3 077	7 692	0	0	0	0	0
Ploesoma truncatum	4 615	38 462	123 077	0	47 863	0	0
Polyarthra dolychoptera	43 077	292 308	328 205	603 077	526 496	921 368	230 177
Polyarthra major	0	7 692	0	0		179 487	27 080
Polyarthra vulgaris	9 231	0	0	0	127 635	442 735	230 177
Pompholyx sulcata	0	0	0	0	590 313	179 487	148 938
Synchaeta kitina	6 154	84 615	205 128	35 897	191 453	47 863	0
Synchaeta sp.	7 692	30 769	246 154	50 256	15 954	47 863	40 619
Synchaeta stylata	0	23 077		0	0	0	0
Trichocerca multicrinis	1 538	53 846	41 026	57 436	0	191 453	121 858
Trichocerca porcellus	23 077	15 385	41 026	28 718	31 909	0	0
other taxa	6 154	15 385	123 077	28 718	79 772	95 726	54 159
Total Areal Density (ind·m^{-2})	313 846	1 984 615	8 656 410	2 563 077	6 477 493	5 408 547	5 591 944
Total Volumetric Density (ind·m^{-3})	209 231	330 769	360 684	732 308	462 678	386 325	232 998

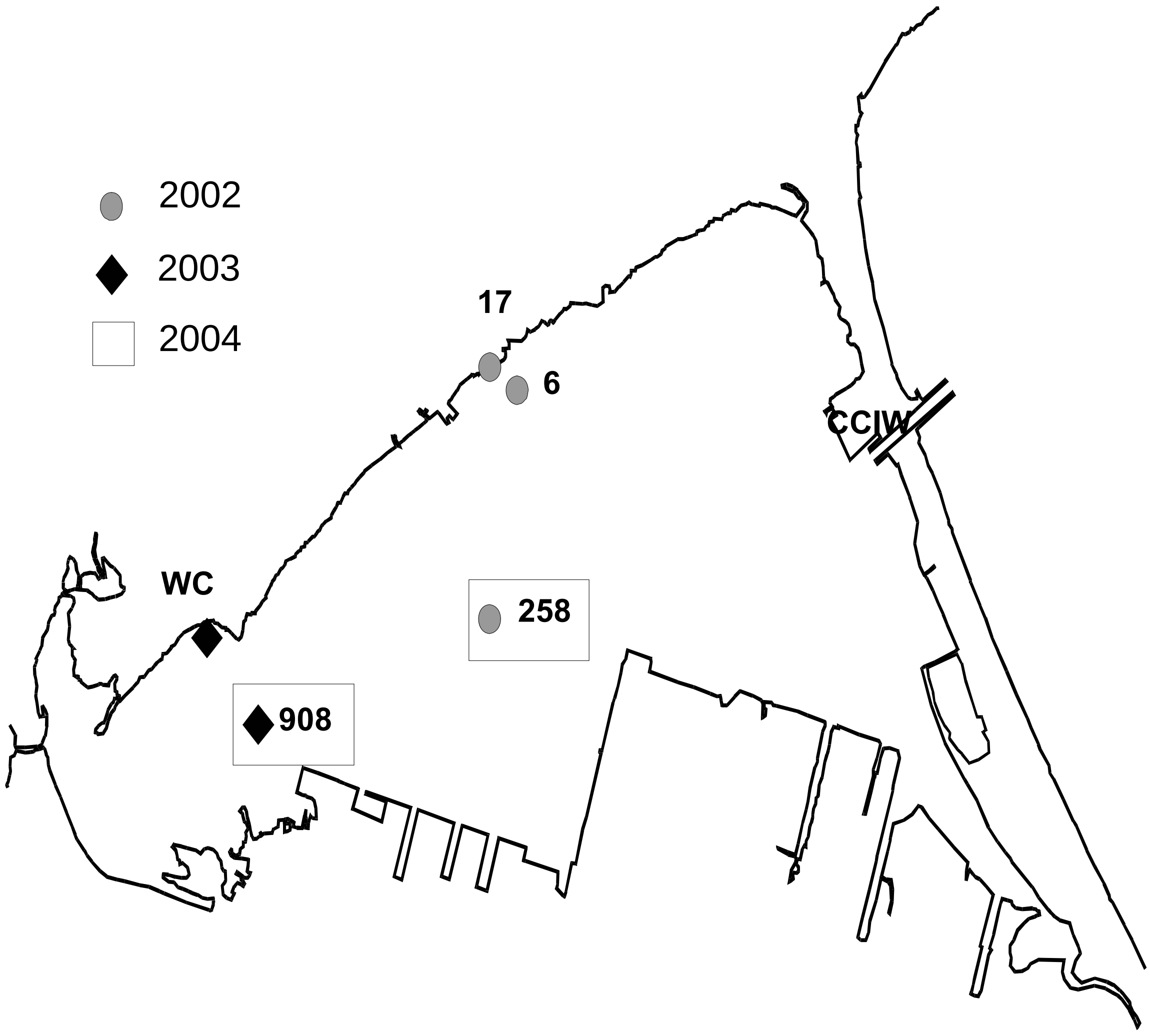

Fig. 1. Zooplankton sampling Stations in Hamilton Harbour for 2002, 2003, and 2004.

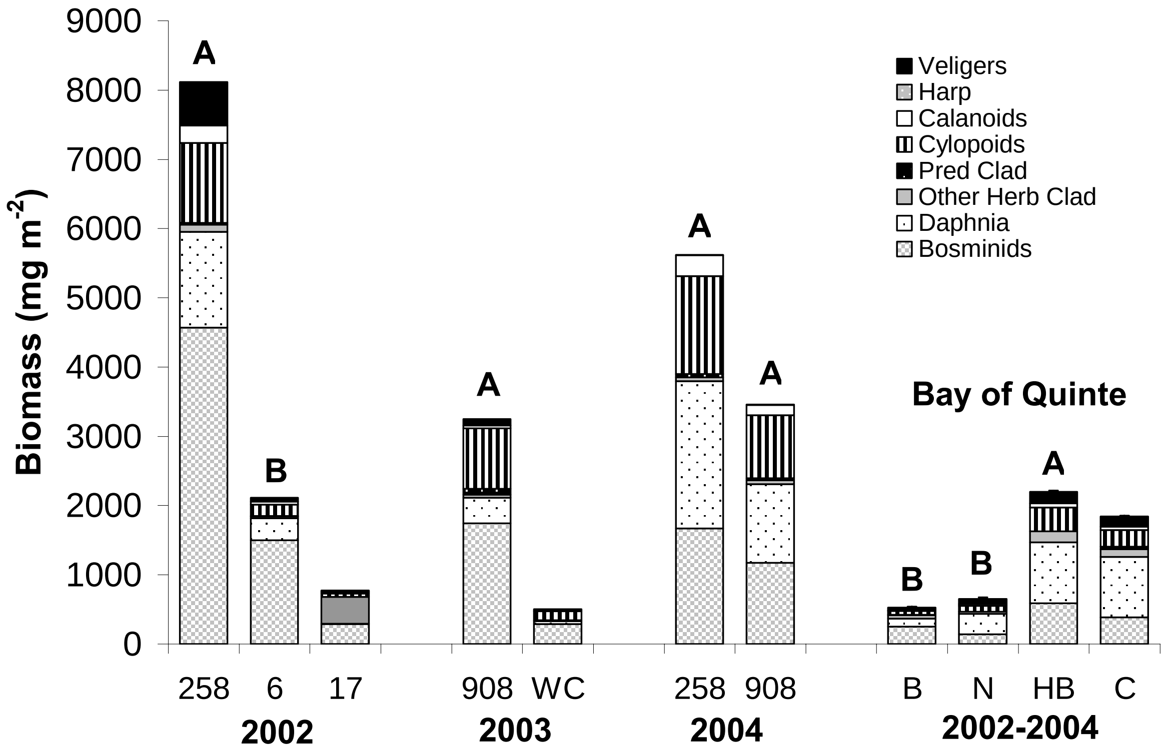

Fig. 2. The seasonal weighted mean areal biomass in Hamilton Harbour for the dominant zooplankton groups in 2002, 2003 and 2004. Mean biomass and standard errors are also shown for the Bay of Quinte stations from 2002 to 2004 (B = Belleville, N = Napanee, HB = Hay Bay and C = Conway). 'A' and 'B' above the bars denote comparable stations between Hamilton Harbour and the Bay of Quinte.

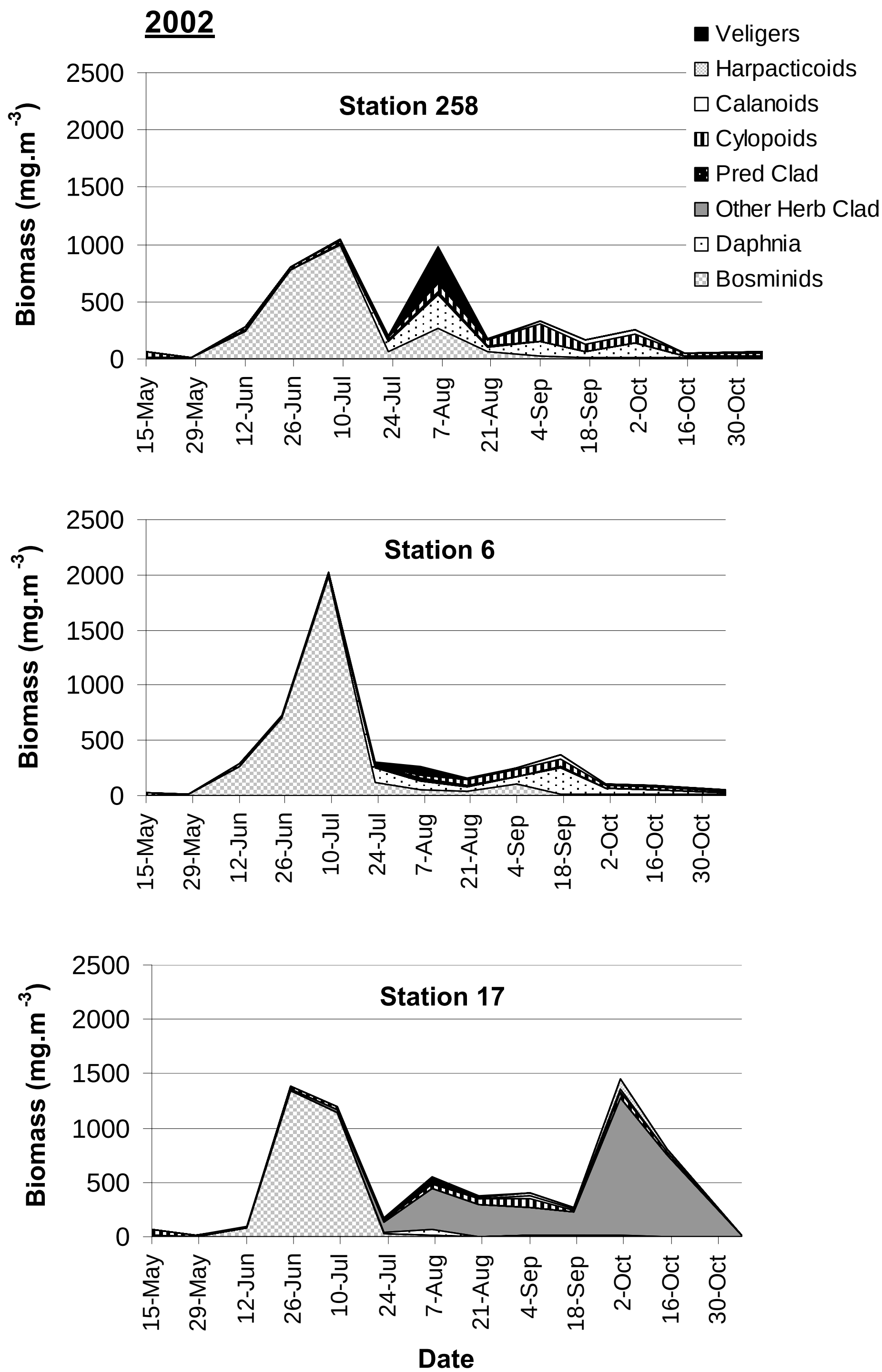

Fig. 3. May to October 2002 seasonal trends in volumetric dry biomass at stations 258, 6, and 17 in Hamilton Harbour. Bosminids include *Bosmina* and *Eubosmina*, *Daphnia* includes *D. galeata mendotae* and *D. retrocurva*. "Other Herb Clad" represent the remaining herbivorous cladocerans. "Pred Clad" are the predatory cladocerans.

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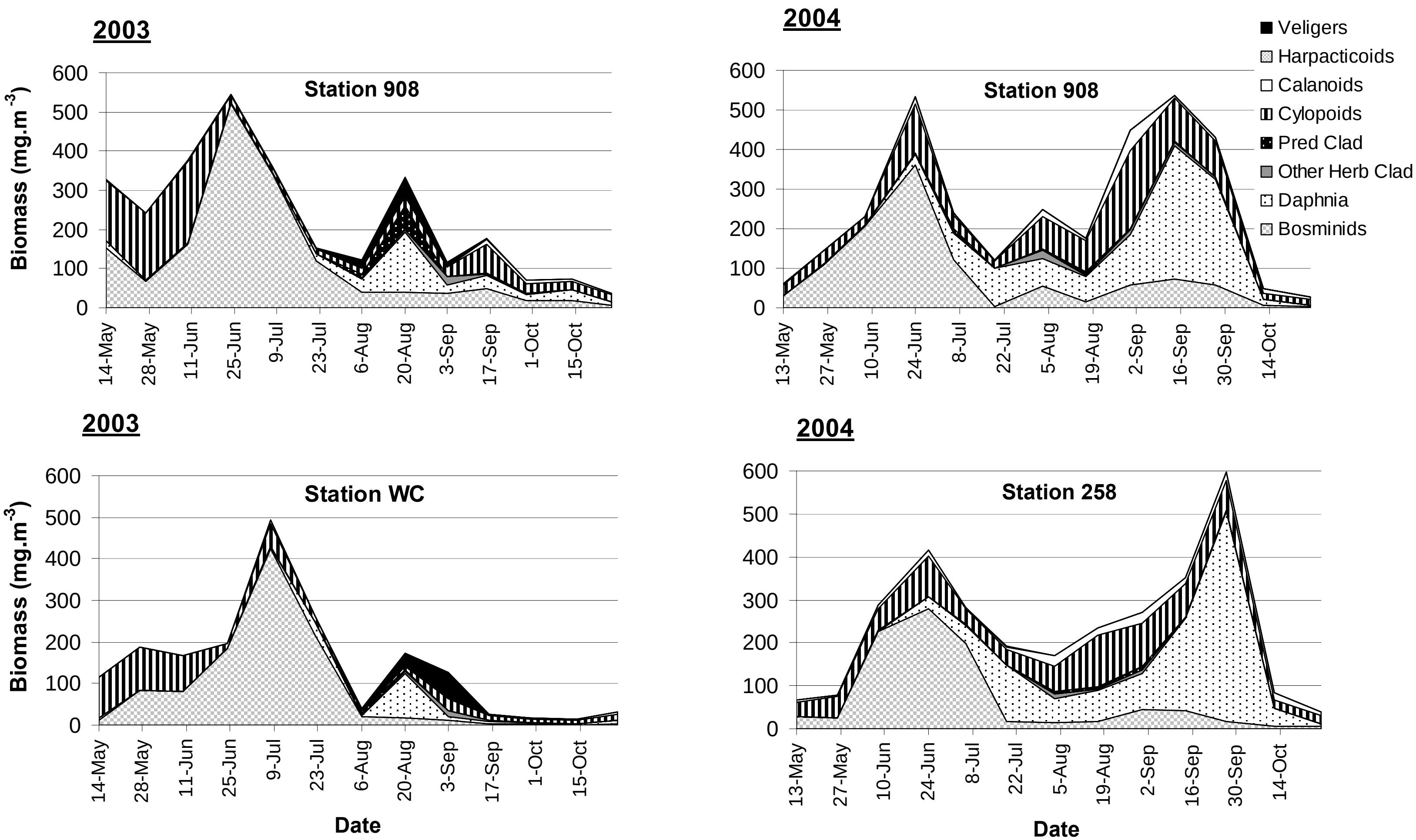

Fig. 4. May to October 2003 and 2004 seasonal trends in volumetric dry biomass at stations 908, WC and 258 in Hamilton Harbour. Bosminids include *Bosmina* and *Eubosmina*, Daphnia includes *D. galeata mendotae* and *D. retrocurva*. "Other Herb Clad" represents the remaining herbivorous cladocerans. "Pred Clad" are the predatory cladocerans.

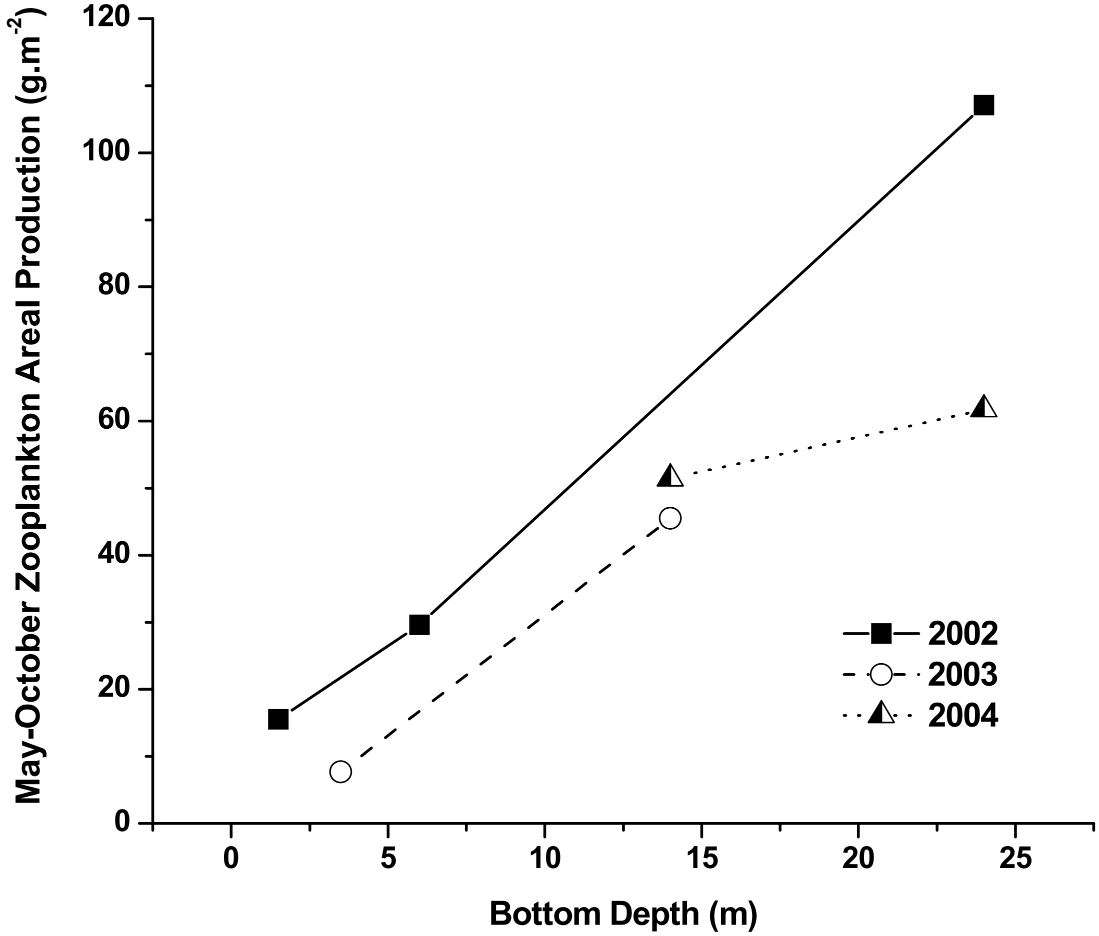

Fig. 5. Areal zooplankton production at different depths in Hamilton Harbour in 2002, 2003, and 2004.

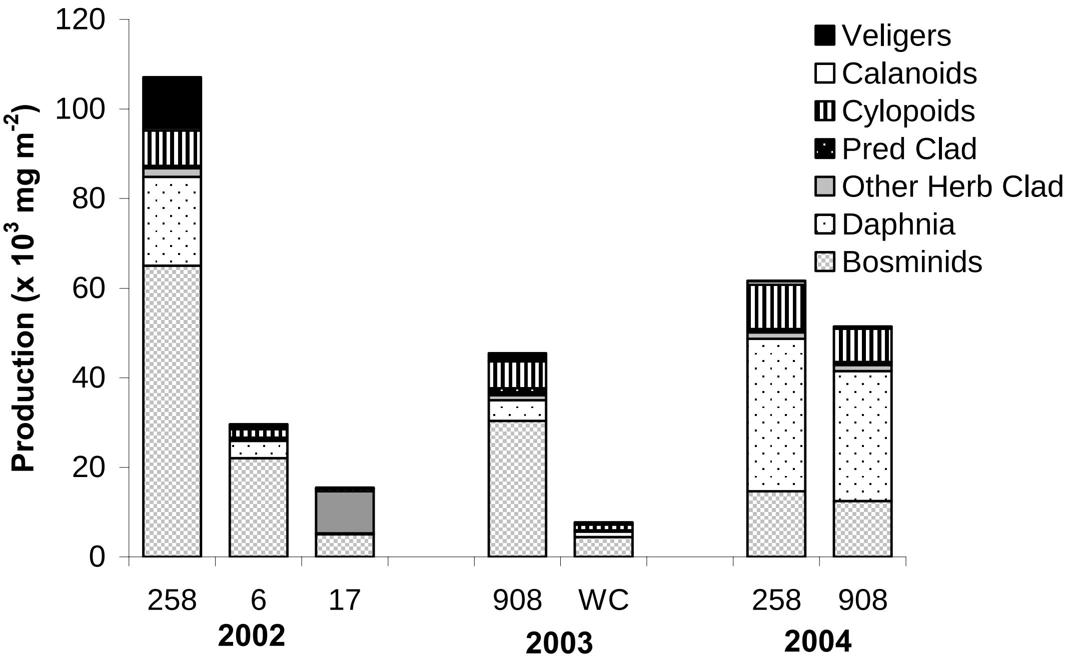

Fig. 6. Seasonal (May-October) areal production for the dominant zooplankton groups in Hamilton Harbour at the stations sampled in 2002, 2003 and 2004.

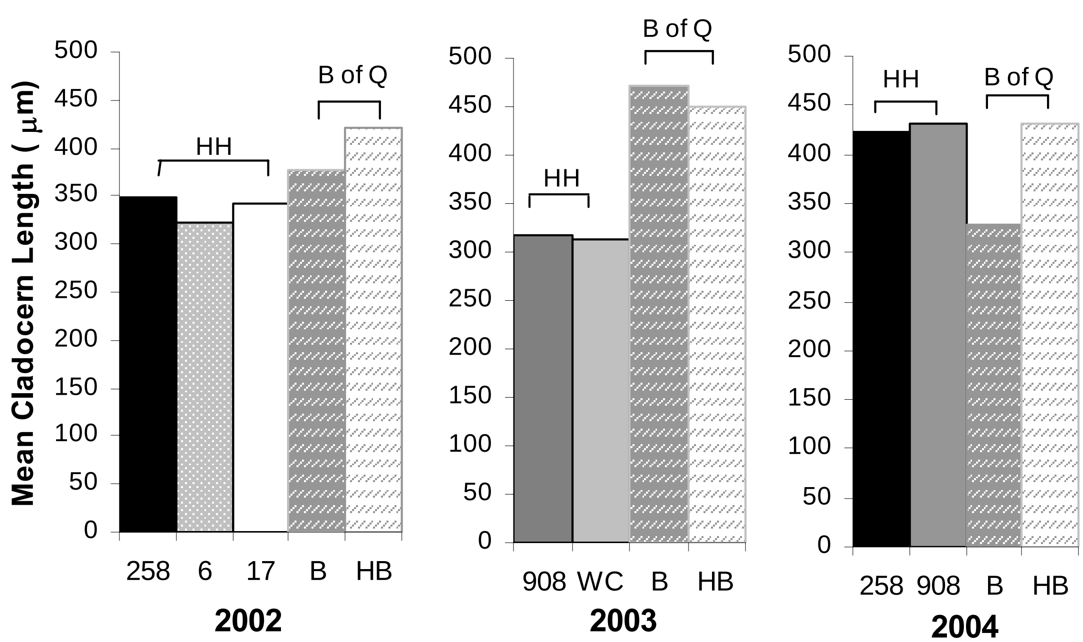

Fig. 7. June 1- October 6 time-weighted mean length of cladocerans at the Hamilton Harbour (HH) stations (17, 6, 258, WC and 908) and upper Bay of Quinte (B of Q) (Belleville = B and Hay Bay = HB).

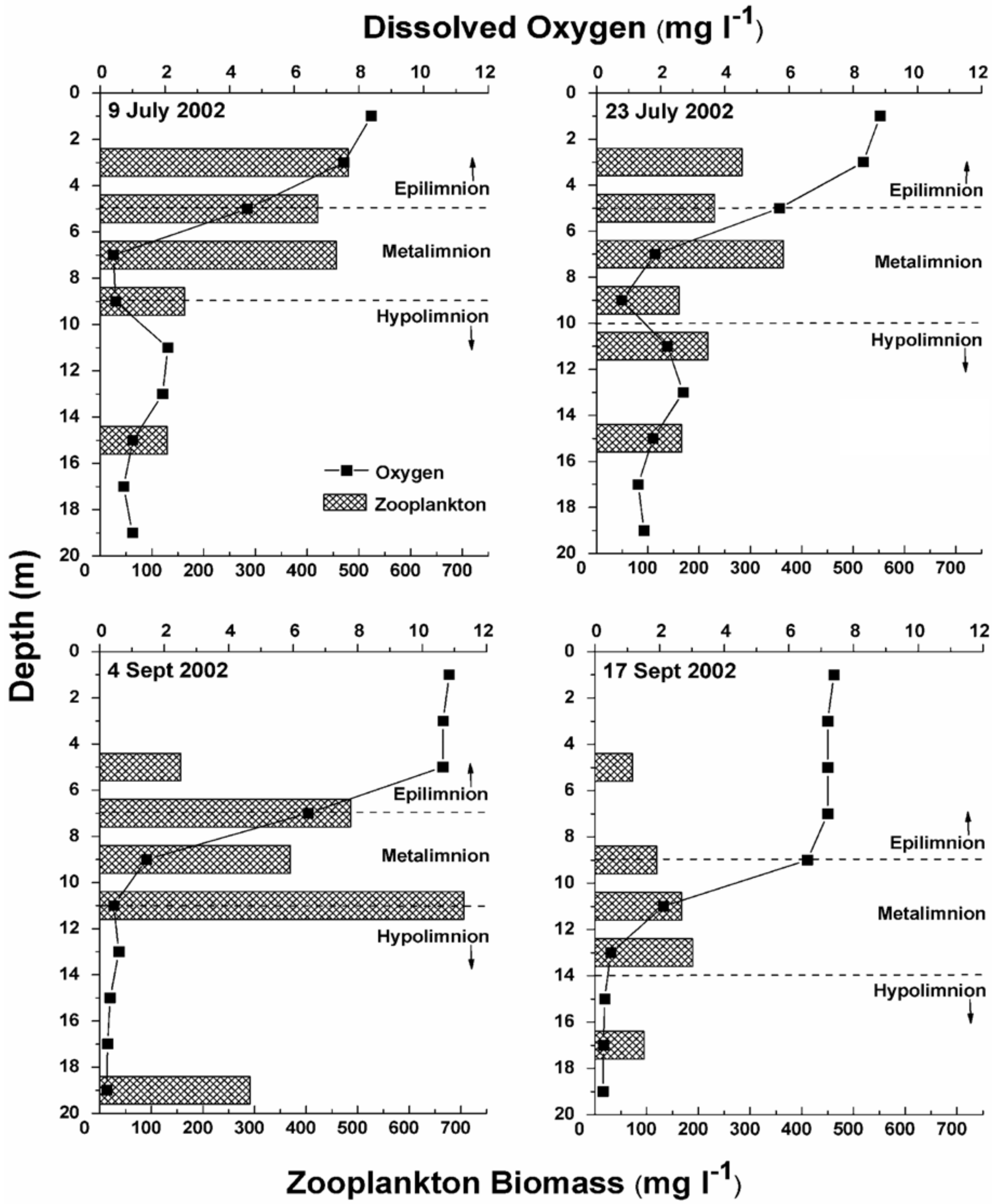

Fig. 8. Vertical distribution of zooplankton biomass at station 258 plotted with oxygen levels on July 9 and 23, September 4 and 17 2002 in Hamilton Harbour. Zooplankton samples not taken at all depths - bars indicate depths analyzed.

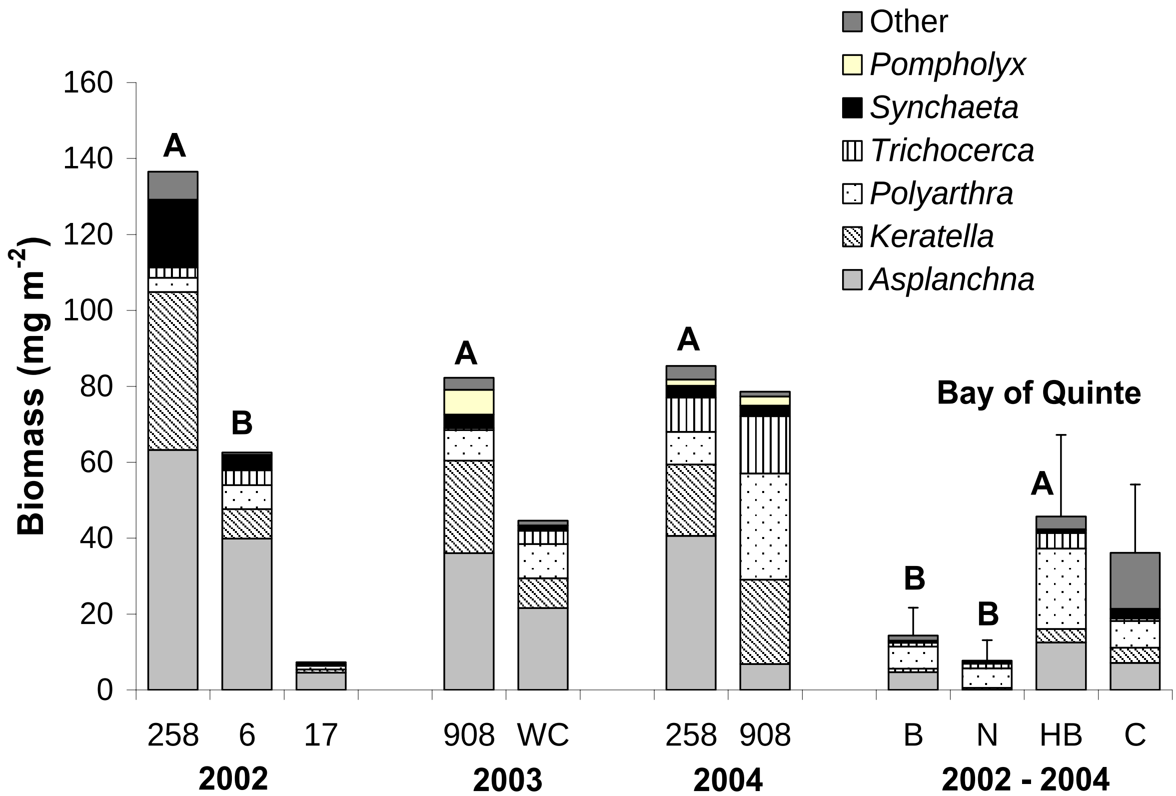

Fig. 9. Seasonal mean biomass (mg·m^{-2}) of dominant rotifer genera in Hamilton Harbour from 2002 to 2004. Mean biomass and standard errors are also shown for the Bay of Quinte stations from 2002 to 2004 (B = Belleville, N = Napanee, HB = Hay Bay and C = Conway). 'A' and 'B' above the bars denote comparable stations between Hamilton Harbour and the Bay of Quinte.

BENTHIC FAUNA IN HAMILTON HARBOUR: 2002 - 2003

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INTRODUCTION

Hamilton Harbour has had a history of poor water quality from eutrophication, low dissolved oxygen levels, and contaminated sediments due industrial and municipal wastewater. In 1981 the harbour was named an Area Of Concern having significant environmental degradation and severe impairment of beneficial uses. Prior to 1964, untreated sewage was discharged into the harbour. Secondary sewage treatment was begun in 1973, expanded in 1979 and improved in 1996 in order to removal much of the nitrate and phosphorus. Industrial loadings were also reduced between 1967 and 1983, with phosphorus (P) and ammonia loadings from the major steel plants declining by 93 % and 94 % respectively (Ontario Ministry of Environment (OME) 1985). A Remedial Action Plan (RAP) to restore the harbour has been developed by a consultative process between government agencies, municipalities and local industries. One of the goals is to restore the benthic community to one that does not diverge from unimpacted sites of comparable physical characteristics, and also to reduce sediment-associated contaminants so that toxicity is not higher than in control sediments (BARC 2006)

Since 1913, benthic fauna have been used as indicators of water quality and oxygen concentrations. These relatively stationary organisms, over a time period of months to years, integrate the chemical changes in the sediments and water column above them (Brundin 1958, Weiderholm 1980). In 1964, Johnson and Matheson (1968) surveyed the macro-invertebrate community and sediments of Hamilton Harbour, when they found no macro-invertebrates present in approximately 2 km^2 nearest the steel mills. Elsewhere, the benthic community was dominated by the pollution tolerate oligochaete worm *Limnodrilus hoffmeisteri*, (Milbrink 1983) and the sewage worm *Tubifex tubifex* was abundant at depths beyond 12 m. In 1984, the same sites used by Johnson and Matheson were resampled by the Department of Fisheries & Oceans (DFO) to examine what changes had occurred following the initial environmental improvements to the harbour (Portt et al. 1988 unpublished report). At that time, no area of the harbour was devoid of benthic macro-invertebrates, but oligochaetes remained the only invertebrates in the deepest part of the harbour. Hanna (1993) found that in spite of the improvement in abundance and species composition, the benthic community in 1989 continued to reflect eutrophic conditions, with only oligochaetes and a few Crustacea present at sites deeper than 12 m during summer.

A re-survey of the benthic community in the harbour was undertaken in 2002 and 2003 to examine what changes have occurred in response to the improved water clarity and reduced nutrient concentrations following the implemented remedial actions. Also, invading zebra mussels (*Dreissena* spp.) have colonized the docks and breakwalls in the harbour since 1990, increasing water clarity and altering the community. In addition to resampling the soft bottomed community at sites sampled previously, the current work also examined the benthic community in shallow sandy habitats at depths < 3 m in conjunction with research on the shallow nearshore fish habitat.

METHODS

A benthic survey of Hamilton Harbour was conducted in conjunction with plankton sampling in 2002 and 2003, no benthic samples were collected during the plankton surveys in 2004. In 2002, the sites chosen had an offshore gradient from the shallow habitat, along the relatively undeveloped north shore at La Salle Park, to the deep central part of the harbour (naturalized beach - industrial harbour). The three plankton sites were at the DFO electrofishing transect site 17 in 1.5 m depth, site 6 at 6.0 m depth near La Salle Park, and at mid harbour Ontario Ministry of Environment (OME) site 258 at a depth of 24 m (OME 1981, Poulton 1987), which is the same as Johnson & Matheson's site 19. Benthic sampling was done at only two of these three sites (site 17 and 258) in May, July and late September. A nearby site 10 at 7.4 m depth was substituted for the plankton site 6 because historical benthic data existed at site 10 from 1964 and 1984 (Johnson and Matheson 1968, Portt et al. 1988, DFO unpublished data). The bottom fauna was also sampled at an additional 6 nearshore sites during May, July and September 2002 to examine the community at depths < 3 m (Fig. 1). These nearshore sites were between La Salle Park and the Burlington sewage treatment plant (STP), of these, sites 10, 32 and 33A had been sampled by Johnson and Matheson (1968). In September 2002, a spatial survey sampled another 5 deeper sites > 9 m with one replicate analyzed from these sites. These 5 sites had been sampled by Johnson and Matheson (1968), and were between Willow Cove and along the harbour's south shore including the heavily industrialized steel mills at the east end of the harbour. Thus in September 2002, a total of 13 sites were sampled to give a spatial view of distribution and biomass of the benthic community in the harbour (Table 1).

In 2003, only two sites were sampled for plankton and benthos. Both sites were located at the west end of the harbour; the shallow nearshore site <4m depth was at Willow Cove (WC), and the deeper Environment Canada site 908 at 14 m was halfway between Willow Cove and the piers on the south shore. Benthic samples were collected at site 908 in May 2003, but they were contaminated with oil. Site 908 is in a no anchor area (submerged oil pipeline), so it was decided to shift the bottom sampling further west to the location of spatial site 3 that was used in 2002. The new site was satisfactory and resampled again in July and October 2003.

The coordinates (based on GPS) for the benthic sites sampled in 2002-2003 are given in Table 1. Wherever possible, site locations and sampling methods were matched to those used in 1964 (Johnson and Matheson 1968) to allow comparisons with historical densities. A large 9 inch Ekman (area 0.05 m^{-2}) was used for sites with a soft bottom; a smaller 6 inch mini-Ponar (area 0.023 m^2) was used at sites with depths <5 m having sandy substrates. Between 1 to 3 replicates were collected at each site. During 2002, only one replicate was analyzed from each site during each season. In 2003, all three replicate samples collected were analyzed each season (Table 1). The collected sediment was screened through a 580 µm screening bucket (US standard #30 mesh), as used by Johnson and Matheson (1968). Screened residues were transferred into Mason jars and preserved using neutralized formalin (37% formaldehyde) with added $CaCO_3$. Based on the volume in the jars, the required volume of formaldehyde (100 % neutral formalin) was added to result in a final concentration of 8 to 10 % formalin. Formalin was used to fix and harden the oligochaete and flatworms in order to reduce their fragmentation especially in the sandy sediments.

Within 2 weeks after collection, the preservative was changed to alcohol for long term storage. The formalin was decanted through a 180 µm brass sieve (US standard #80 mesh), and the sample rinsed with cold tap water. This finer mesh ensured no small organisms retained on the 580 micron screen in the field would be lost during re-screening in the lab. The retained residue, including organisms, was returned to the original field jar and re-preserved with 50% isopropyl alcohol to which a small amount of Rose Bengal was added to stain organisms in order to assist sorting.

Later when analyzed, the alcohol was poured through an 180 µm mesh screen and the sample rinsed with tap water. This insured that alcohol-water convection currents would not interfere with the sorting process. To separate the benthos into its constituents, 5-10 ml aliquots were examined under a stereo dissecting microscope at 6X power. Any organisms present were removed, identified to taxa (usually Family), and their numbers tallied using a 9 unit laboratory counter.

The biomass (wet weight including shells) for each taxa was measured directly as blotted wet weight after rinsing in distilled water. (Dermott and Paterson 1974, Dermott 1979) The enumerated organisms, including fragments, were transferring into a drop of distilled water on a clean Petri dish. A separate drop was used for each taxa that could be identified under the dissecting microscope. The organisms were counted when being sorted. If the organisms could not be identified under the dissecting scope they were placed in alcohol in covered dish until examined in more detail. After counting, the organisms in the water drops were weighed. They were removed from the water drop with forceps, blotted on filter paper to remove excess water, transferred onto a tared weighing pan and weighed on an analytical balance to the nearest 0.05 mg. Surface tension in the water drop made picking up the clump of small organisms, such as ostracods and nematodes, easier than if they were submerged in a vial or dish of water after being counted. Distilled water was added to the drops to keep them from dehydrating until the sample was weighted. Weights of small specimens whose combined weight was less than 0.05 mg, were calculated from average weights in samples which had sufficient numbers so that their pooled weight was measurable. Density and total wet weight (including shells) of each identified taxa were recorded. Biomass is reported as wet weight including shells except for the areal weighted calculations which used shell-free biomass.

Organisms requiring further identification under a compound microscope, such as oligochaetes and chironomids, were weighted to the level of family or subfamily prior to mounting on microscope slides. Wet mounts in water were usually sufficient to identify the chironomids to genus. *Dreissena polymorpha* and *Dreissena bugensis* were separated into species based on shell shape. Newly settled *Dreissena* mussels <1 mm were identified only to genus. After weighing, the organisms were then placed into labelled vials in 70 % ethanol that contained about 1 % glycerine to prevent complete desiccation.

For each site, average density per sample, and wet biomass were calculated for the organisms. Average annual density at the sites and standard errors were calculated where multiple replicates or seasons were sampled. The average benthic community diversity was calculated for each site based on the Margalef (1958) diversity index D calculated as: number of species -1 / natural log (total number of individuals). It was assumed that a minimum of two oligochaete taxa (*Limnodrilus hoffmeister* and immature with hairs) were present at all sites where oligochaetes were found but not identified.

Area weighted biomass was calculated based on bottom area (10^3 m^2) of the depth contour zones of the harbour listed in chapter 1, and the annual average wet biomass during 2002 -2003 for the major taxa, oligochaetes, chironomids, Sphaeriidae, and *Dreissena*. The wet shell-on biomass of the molluscs was converted to shell-free wet tissue by multiplying by the average percent shell-free ratio for each family, 30 % for Sphaeriidae and 56 % for *Dreissena*. Total Biomass in grams was then summed for all the contour zones in the harbour. Area weighted biomass (wet shell-free g m^{-2}) was then calculated by dividing Total Biomass by total area of the harbour. Standard error and average were used to calculate the low error estimate (Ave. - S.E.).

RESULTS

The four nearshore transect sites 13A, 15A, 17 and 18 were located on sandy sediments at depths ranging between 1.4 and 2.3 m (Fig. 1, Table 1). Sites 17 and 18, located near La Salle Park, were well protected by the constructed habitat islands, and the north shore. Thus substrates at these two sites contained more plant debris and macrophytes than did the substrate at exposed sites 13A and 15. Substrates at the deeper sites were finer silts with higher organic content. A large deposit of dead zebra mussel shells occurred at depths between 6 and 7.4 m off site 10. Visible oil and tar residues occurred in the sediment grabs at the deepest site 19 in the middle of the harbour and at sites 14 and 34 along the south industrial shore (Fig. 1). Due to the oil, benthic sampling at site 908 was relocated to nearby site 3.

The highest density of benthic macroinvertebrates occurred at nearshore site 17 at 1.5 m depth, located near La Salle Park. Lowest density in 2002 occurred at site 3 in the west end, and density was also low at site 14 near the south shore. Total density ranged from 2,440m^{-2} to 38,835 m^{-2}. The bottom fauna of Hamilton Harbour remained dominated by oligochaete worms which were the only invertebrates found at several of the deeper sites. Oligochaete densities ranged from 2,440 m^{-2} at site 3 to 29,233 m^{-2} at site 33 to (Table 2). Average density at site 908 / 3 increased to 27,182 m^{-2} in 2003 compared to only 2400 m^{-2} in 2002 but again oligochaetes formed over 97 % of the invertebrates present. In 2003, both *Pisidium* sp. and chironomids were present at low density at site 908 / 3, while in September 2002 they were absent at site 3 (Table 2).

Total wet biomass of the invertebrates ranged from 0.38 g m^{-2} at site 14 near the south shore, to 120.9 g m^{-2} at site 10 (Table 3, Fig. 2). In spite of the large biomass of the shelled zebra mussels, the non-Dreissena biomass in the harbour remained greater than 55 % of the total benthic biomass present at all sites except 17, 18 and especially at site 10 where the biomass of non-Dreissena invertebrates represented only 5 % of the total wet shell-on biomass. (Fig. 2). Site 908 / 3 had the next highest benthic biomass of 21.8 g m^{-2}, which was due to the high numbers of oligochaetes.

After oligochaetes, chironomids were the next most common invertebrates collected from the harbour. Chironomid density was greatest at site 17, while biomass was greatest at site WC (4.7 g m^{-2} wet). This biomass exceeded that of the oligochaetes at this site (Table 3). Although seasonally chironomid density was greatest at WC in October (Tables 4 to 7 and 5), their biomass was very constant over the summer, ranging from 4.78 g m^{-2} wet in May, 2003 to 4.79 g m^{-2} wet in October. As with zebra mussels, chironomid density was greatest in water less than 7 m, the midges were rare or absent at sites deeper than 9 m. Seasonal biomass of the chironomids was highest at 7 m in the spring before the larger midges (*Chironomus plumosus* and *C. attenuatus*) emerged (Fig. 3). The chironomid biomass in the shallows was more consistent over the summer as the smaller species present in the shallows often have two or more generations a year, compared to one synchronous emergence in deeper water.

In 2003, total density and wet biomass were much greater at site 908 / 3 in July than at nearshore site WC (Tables 4 to 7). As with other sites > 9 m in the harbour, tubificid oligochaetes dominated the benthic fauna at site 908. Chironomids dominated the benthic fauna at site WC, but oligochaetes were as important in the biomass of the October 2003 samples at site WC (Tables 4 and 5). A few snails and amphipods were present at this 3.5 m deep nearshore site. No insects other than chironomids were collected at site WC. Seasonally, biomass was greatest in the fall at site WC (11.48 g m^{-2}), but greatest in spring (26.10 g m^{-2}) at the deeper site 908 / 3 (Tables 5 and 7). The higher biomass in May 2003 at site 908 was due to large maturing tubificid oligochaetes, and a few Chironomids, especially *Procladius* sp. which mature and emerge in July.

Zebra mussels (*Dreissena polymorpha*) dominated the *Dreissena* in the harbour during 2002. Quagga mussels (*D. bugensis*) represented less than 0.6 % of the mussels collected and were present only at sites 10 and WC. The abundance of zebra mussels was inversely related to depth with few sites beyond 8 m supporting live mussels (Fig. 4). However, mussel biomass was greatest at site 10 in 7.4 m of water. At this site, average seasonal shell-on wet mussel biomass exceeded 115.2 g m^{-2}, in spite of the low density (306 m^{-2}), because very large mussels were attached to the dead shells present at that site (Tables 2 and 3). The estimated wet tissue of the mussels at site 10 was 64.5 g m^{-2} (wet shell-free), assuming the ratio of wet shell-free tissue to total wet weight including shells was 56 % as in eastern Lake Erie (Dermott et al. 1993). The shell-free dry biomass equivalent of this wet tissue would be 7.80 g m^{-2} or 12.1% of the wet shell-free biomass (Dermott et al. 1993). Wet shell-on biomass of the *Dreissena* at shallow sites 13A, 15, 17, and 18 averaged 3.23 g m^{-2} during 2002. This would represent an average of 1.81 g m^{-2} of wet tissue (shell-free).

The numerous small mussels at these four nearshore sites (< 2 m depth) were those that had settled during the summer. The exposed nature of these shallow sites, unstable sandy sediments, and risk of heavy ice scour at depths less than 2 m during winter would make these sites unfavourable mussel habitat except during summer. The number of small mussels increased rapidly over the summer at the shallow inshore sites (# 13A to 18), however weight gain was limited (Fig. 5). At more protected site WC (3.2 m), the majority of the mussels settled between July and October 2003. However, there was little increase in *Dreissena* biomass following the settlement (Table 4 and 5). At the same time, a few *Dreissena* settled at site 908 / 3 (Tables 6 and 7). The 7 m depth zone (sites 10 and 32) had the greatest average wet weight, but the high *Dreissena* biomass at site 10 in September 2002 was due to one sample containing a clump of very large mussels (Fig. 5). Average *Dreissena* biomass in the harbour was much less than in Lake Erie or similar habitats in the Bay of Quinte (Table 8).

Mussel biomass was greatest along the north shore and almost absent on the soft silty bottom of the harbour (Fig. 6). The few *Dreissena* at site 19 in 2002 (Table 2) likely had fallen from the mooring float at that site, rather than settled and grown on the soft anoxic sediments during the summer. Although not sampled in 2002, populations of mussels occur on the rocky fill along the west and east sides of the harbour, and along the docks and breakwalls of the harbour. Unpublished data from midsummer 2002 indicated that zebra mussels were very common along the dock at Canada Centre for Inland Waters (CCIW). During October 2002, density of mussels which had settled in fish cages along the dock at CCIW over the summer was 7525 m^{-2}, with a wet biomass including shells of about 1050 g m^{-2}, representing settlement and growth between June 1 and October 3, 2002. In mid summer 2003, mussels were very rare along the rocky shoreline at the entrance to CCIW (43° 18.12'N: 079° 48.05'W) due to extensive mortality during the previous cold winter. In November 2003, density of mussels (all less than 10 mm length) was about 100 m^{-2} on the rocks at the entrance to CCIW. Variability of the settlement was high between sites and between years. In October 1991, a year after the arrival of mussels in the harbour, density at the entrance to CCIW ranged from 71 to 3033 m^{-2}, with a wet biomass ranging from 117.6 to 434 g m^{-2} (including shells). In 1991, mussel density on the rocky causeway over the outlet from the Burlington Sewage Treatment Plant (STP) was 29,890 with a wet biomass of 6847 g m^{-2}. In December 1992, this rocky causeway supported a *Dreissena* population of 265,081 m^{-2} having a total wet biomass of 8694 g m^{-2}. Of these, the density of adult size mussels with shell length greater than 10 mm was only 1079 m^{-2}, with a wet biomass (including shells) of 2346 g m^{-2}.

The amphipod *Gammarus fasciatus*, was collected only at shallow sites of less than 3.5 m depth (Table 2). No *Echinogammarus ischnus* were collected in sediment samples from 2002 or 2003, but they were abundant on rocky substrates and among the zebra mussels on the rocks near CCIW (DFO unpublished). Specimens of the Lepidoptera *Acentria* were found only at site 13A in July 2002. These were the only non-chironomid insects collected in the benthic grabs during 2002 and 2003.

In addition to the worms, only a few small fingernail clams (Sphaeriidae) and encysted Harpacticoida were present in the sediments from the hypolimnion of the harbour. The isopod *Caecidotea* sp. and the Platyhelminthes were collected only at site 10, possibly associated with the zebra mussel shells, although isopods and flatworms *Dugesia* sp., and *Hydrolimax grisea* are common on the rocks near shore. Gastropods (*Valvata sincera*, and *Pleurocera acuta*) were present at only 2 sites, #32 and WC, both < 8 m deep. Although gastropods are present on the rocks along the shore (*Physella* sp., and *Bithynia tentaculata*), they were absent in the samples on the sandy substrates from all sites < 3 m deep. *Hydra* were found only at sites less than 7 m depth in 2002 but not at site WC in 2003 (Table 2 and 4). Water mites (Hydracarina) were present to 9 m depth, but their small size added little to the macroinvertebrate biomass. One specimen of the Hirudinea *Mooreobdella fervida* was collected at site 33A in May 2002. Like the flatworms, the leeches *Dina* sp. and *Glossiphonia* sp. are present on the rocks along the shore (DFO unpublished).

The calculated area weighted shell-free biomass as g m^{-2} is displayed in Table 9. As 62 % of the harbour area is below 10 m, the average biomass present in the hypolimnion has a large effect on the total biomass in the harbour. Not surprising, oligochaetes had the greatest area weighted biomass in the harbour at 11.2 g m^{-2} (Table 9). However, *Dreissena* were the next most important invertebrate with an area weighted wet biomass of 5.1 g m^{-2} shell-free. This area weighted shell-free biomass would be about 9.1 g m^{-2} wet with shells, slightly less than the estimated numeric average from all the samples of 10.1 g m^{-2} wet with shells (Table 8). Chironomids, with a high density were minor components to the overall harbour biomass (0.6 g m^{-2}; Table 2 and 9). However, variability of *Dreissena* biomass was very high (range 0.33 to 9.90 g m^{-2}), so their low estimate was less than the low estimate of the chironomid biomass (0.35 g m^{-2}).

In 2002 - 2003, the average Margalef (1958) diversity index was greatest at site WC (2.55). Diversity at site 908 / 3 (0.99) was higher than at most of the other sites deeper than 9 m in the harbour (Table 2). This consistent low diversity at the deeper sites reflects the limitations put on the community by the low hypolimnetic oxygen levels, perhaps in combination with high metal and polyaromatic hydrocarbons (PAH) levels in the sediments.

Historical Comparisons

Data from the same sites during September 1964 and 1984 (Johnson and Matheson 1968, Portt et al. 1988) indicated that total macroinvertebrate density (pre-*Dreissena*) was lower in September 2002 than in 1984 at sites: 3, 6, 10, 14, and 19 (Table 10). This was due to greatly reduced density of oligochaetes at all these sites, as well as site 34. Most sites had an increased density of invertebrates between 1964 and 1984, followed by a general reduction by 2002. In 1964, oligochaetes were the only invertebrates collected at several sites, while no invertebrates were present at site 34 in 1964. Composition of the oligochaete species was not examined in 2002, preventing comparisons of changes in species abundance since 1984. No worms of the family Naididae were identified in 1964, but they were present in 1984 (Table 10). The Naididae are more sensitive to low oxygen than are the pollution tolerant Tubificid worms. In 1984, almost all the invertebrates collected were oligochaetes or chironomids, which remained the most abundant benthic taxa in 2002. A comparison of the oligochaete abundance indicated a reduction in density in the deeper parts of the harbour between 1984 and 2002 (Fig. 7). The number of chironomid taxa increased between 1984 and 2002 at sites: 6, 10, 14, 19 and 32. In 1964, only 4 chironomid taxa were identified in the harbour, 13 taxa were present in 2002. For sites sampled in 1964, 1984 and 2002, the largest number of chironomids were present at site 10 (173 per Ekman).

Spatial Survey

Zebra mussels (*Dreissena*; with their shells on), represented more than 50 % of the total invertebrate biomass at 3 sites along the north shore (10, 17, and 18). Chironomids were important near the north and northeast shore of the harbour (sites 13, 15, 17, 18 and 32), and at Willow Cove in the west. They were rare in the samples from the middle and southwest end of the harbour. Oligochaetes formed over 90 % of the biomass at all sites beyond 8 m depth except site 6 at 9.6 m deep located east of Willow Cove. The benthic community at all sites in the middle of the harbour and along the industrial south shore was almost exclusively oligochaetes, with a biomass above 18 g m^{-2} near the steel mills (Fig. 8).

Species diversity measured as the Margalef (1958) index D was below 2 at most sites in the harbour. In 2002 - 2003, the highest diversity index was at site WC (2.55) in the west, diversity was also high at sites near La Salle Park and the northeast corner of the harbour. Diversity at site 908 / 3 (0.99) was higher than at most of the other sites deeper than 9 m in the harbour (Table 2, Fig. 9). Most sites deeper than 9 m had diversity values less than 1 (Tables 2 and 10), indicating a restricted benthic community. This consistent low diversity at the deeper sites reflects the limitations put on the community by the low hypolimnetic oxygen, perhaps in combination with high metal and PAH levels in the sediments.

DISCUSSION

The community present at depths beyond 9 m suggests the benthic species are still restricted by severe oxygen limitation during part of the year in spite of improved environmental conditions in the harbour. Oligochaete density in 1964 was considerably lower than in 1984 or 2002. In 1964, Johnson and Matheson (1968) found no organisms present at 6 sites. That year was a time of high eutrophication and high metal levels, up to 25 % Fe_2O_3 in sediments of the harbour's south-east part, which prevented survival of even the sewage worms *Tubifex tubifex* and *Limnodrilus hoffmeisteri* (Johnson and Matheson 1968). Following sewage treatment after 1973, oligochaete populations increased in all parts of the harbour resulting in the highest populations in the time series examined. Further improvements to the sewage treatment since 1984 and filtering by the zebra mussels have reduced the amount of organic matter settling into the deeper parts of the harbour. By 2002, this has reduced the density of oligochaete worms, which feed on the bacteria and organic matter in the deeper sediments.

The increased number of chironomid taxa between 1984 and 2002 at 5 sites suggests some improvements in water quality. The tribe Tanytarsini and many of the small Chironomini species such as *Cladopelma* and *Polypedilum* appeared at several sites in 2002. These small genera have higher oxygen requirements than most of the larger Chironomini genera, such as *Chironomus* or *Cryptochironomus* (Brundin 1958). A total of 4 chironomid taxa were found in 1964, 8 in 1984 and 15 were present in 2002. No amphipods or gastropods were collected at any of these common sites in any of the survey years, but all these 9 sites were deeper than 6 m.

During 1984, no samples other than in Windermere Basin were collected from depths shallower than 4 m, so no comparison can be made for the benthic community along the north-east shore of the harbour where oxygen is rarely limited. In 1989, Hanna (1993) collected up to 15 invertebrate species from the area near La Salle at depths less than 5 m. This number was similar to the number of taxa present in the summer of 2002 on the firmer nearshore substrates in the same area. Hanna (1993) found that oligochaetes were the only class of invertebrates present at depths beyond 12 m from June to August 1989. Ten oligochaete species were identified in 1964, 9 were found in 1984 and Hanna (1993) found only 7 with *Limnodrilus claparedeanus, L. udekemianus, Spirosperma ferox* and *Potomothrix moldaviensis* being absent in 1989. No worms of the family Naididae were identified in 1964, 1989 nor 2002, but they were identified in 1984. Worms in the family Naididae are less tolerant of eutrophication than are most of the genera in the family Tubificidae.

In 1989, Hanna (1993) found that the Margalef species diversity averaged 1.4 at shallow sites (< 8 m deep), and 0.5 at deeper sites in Hamilton Harbour. In comparison, the diversity of the benthic community inhabiting the 5 to 7 m depth in the Big Bay portion of the eutrophic Bay of Quinte averaged 1.11 (SE= 0.05) during 1966, 1.14 (SE=0.18) in 1985, and increased to 3.19 (SE= 0.06) in 2001. Benthic diversity in the Bay of Quinte at 20 m depth (Glenora site) was 2.99 (SE= 0.12) in 2001 (DFO unpublished data). Insects, gastropods, and amphipods were very rare in the Hamilton samples including those from littoral sites < 2.5 m depth. Hanna (1993) also found only one non-chironomid insect species in sites near the north shore, and that amphipods and gastropods represented only 0.02 % and 0.01 % of the invertebrates respectively. The periodic low oxygen episodes that occur in the nearshore as shallow as 3.5 m (Burley, *this volume*) would restrict the distribution of the Ephemeroptera and Trichoptera in the nearshore compared to the more typical community of the sublittoral zone as occurs in the Bay of Quinte. Added to the limitations by poor water quality, intensive fish predation in the limited suitable habitat may be reducing the benthic community in the shallows. Gobies (*Neogobius melanostomus*) are the most common fish in the harbour and they have been accused of reducing benthic populations in Lake Erie (Barton et al. 2005).

The composition of the benthic fauna in Hamilton Harbour remains constrained at depths beyond 8 m, mainly due to the summer anoxia of the hypolimnion (Burley, *this volume*). Oxygen concentrations below 1 mg l^{-1} restrict not only the fish but also the benthic invertebrates that can survive in the middle of the harbour (Warren et al. 1973). In addition, levels of Cr, Cu, Hg, Pb and especially Zn are elevated in the mid harbour and deeper west-harbour sediments (OME 1981, Krantzberg 1994, Jackson et al. 1995). These mid harbour sediments were shown to reduce survival of the amphipod *Hyalella* in sediment assays (Munawar et al. 1999), and the PAH contaminated sediments in the harbour induce strong genotoxic responses (Marvin et al. 2000). With time, contaminated sediments from 20 years ago should become buried under less contaminated particles following the improvements to the waste water and storm drain management. However continuing anoxic conditions at the sediment surface and re-suspension of contaminants from shipping and storm disturbance (Rukavina and Versteeg 1996) may continue to make the harbour sediments an unfavourable environment for the re-establishment of a normal benthic community.

Mussel density in the harbour was less than densities in Lake Erie and the Bay of Quinte. The large area of unsuitable, soft bottomed habitat beyond 8 m in the harbour limited the average biomass in the harbour to about 1/10 the wet biomass that existed on comparable substrates and depths in Lake Erie (Jarvis et al. 2000), in spite of the lower algal biomass in eastern Lake Erie.

The benthic community living above 9 m is also very limited in species composition. Amphipods, gastropods and Turbellaria were very rare in samples from this part of the harbour. For the littoral and sublittoral zones between of 1.4 and 5 m, the community is also severely restricted. The only insect collected was the Lepidoptera *Acentria*, which was also collected by Hanna (1993). Other insects, such as Trichoptera, Ephemeroptera, Odonata, and Coleoptera were absent in any of the benthic samples collected in 2002 and 2003. As a result, average diversity in the 1 to 8 m zone of the harbour was 1.98 (S.E = 0.2). Comparative Margalef diversity values of over 2.5 exist in eastern Lake Erie (Dermott 1994), and up to 5.1 at 1 m depth in relatively pristine Batchawana Bay (Dermott 1984). Even at 6 m depth in the Bay of Quinte, another Area of Concern with eutrophication problems, the Margalef (1958) diversity index had increased from less than 1 in 1982 to above 3.0 by 2000 (Dermott unpublished, Bay of Quinte Annual

Report 2002). In comparison, the total benthic biomass in the Bay of Quinte at Big Bay was less than 5 g m^{-2} wet with shells, yet above 10 g m^{-2} in Hamilton (Fig. 2).

SUMMARY

A seasonal survey of the benthic fauna was conducted at 7 sites along the north side of the harbour in 2002 (May, July, September), with additional samples collected along the south side in September Oligochaetes dominated the fauna with densities from 23,880 at 1.7 m depth to 19,690 at 23 m in the middle of the harbour. Chironomids ranged from 410 to 6800 m^{-2} above 9 m depth, but were rare below 9 m. Zebra mussels on sand and silts above 9 m depth ranged from 7 to 6000 m^{-2}. Other invertebrate types were rare in the anoxic sediments below 8 m depth. Benthic Diversity ranged from 0.2 along the south side to a maximum of only 2.5 off the north shore at 3 m depth. The benthic community present beyond 9 m still suggests severe oxygen limitation during mid summer.

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Table 1. Site location, number of seasons sampled, and sampling devices used to examine benthic fauna in Hamilton Harbour, 2003-2003. Spring, summer and fall seasons were May, July and late September, with several sites sampled only in September 2002. Site 3 was substituted for site 908 in July and September 2003. A 9 inch Ekman (0.05 m^2) or 6 inch mini-Ponar (0.023 m^2) were used depending on substrate.

		Sampling Year - 2002			
Site	Depth (m)	Device	Seasons	Latitude	Longitude
13A	-1.4	mini-Ponar	3	43° 18.666'	079° 48.481'
15A	-1.9	mini-Ponar	3	43° 18.756	079° 48.626
17	-1.5	mini-Ponar	3	43° 18.201	079° 50.354
18	-2.3	mini-Ponar	3	43° 18.133	079° 50.405
10	-7.4	Ekman	3	43° 17.725	079° 51.203
33A	-5.6	Ekman	3	43° 18.540	079° 48.460
32	-7.8	Ekman	3	43° 18.221	079° 48.612
4	-9.2	Ekman	1	43° 16.559	079° 52.046
6	-9.6	Ekman	1	43° 17.181	079° 51.883
3	-12.2	Ekman	1	43° 16.840	079° 52.530
34	-13.5	Ekman	1	43° 17.104	079° 48.624
14	-14.0	Ekman	1	43° 16.665	079° 50.944
19	-23.5	Ekman	3	43° 17.172'	079° 50.224'
		Sampling Year - 2003			
Site	Depth (m)	Device	Seasons	Latitude	Longitude
WC	-3.2	mini-Ponar	3	43° 17.183'	079° 52.268'
908	-14.6	Ekman	1	43° 16.867	079° 51.883
03	-14.2	Ekman	2	43° 16.768'	079° 52.443'

Table 2. Average density (no. m^{-2}) and S.E. of benthic invertebrates in Hamilton Harbour during 2002 (May, July, September) and 2003 for sites WC and 908 (May, July, October). Samples without S.E. were only sampled in late September, depths are in meters.

		TOTALS		Non-*Dreissena*		Nematoda		*Hirudinea*	Oligochaeta		*Dreissena* spp.		*D.*	*D.*	Sphaeriidae	
Site	Depth	**Density**	**S.E.**	Density	S.E.	Density	S.E.	Density	Density	S.E.	Density	S.E.	*bugensis*	*polymorpha*	Density	S.E.
13A	1.4	16035	6244	14880	5683	165	123	0	11085	3997	1155	919	0	1155	0	0
17	1.5	38835	14786	32880	12176	570	60	0	23880	10813	5955	2984	0	5955	0	0
15A	1.9	13470	1277	13410	1279	270	135	0	6405	2162	60	40	0	60	45	26
18	2.3	18480	7865	15885	5786	120	30	0	10695	3631	2595	2116	0	2595	0	0
33A	5.6	31047	5891	31040	5890	33	18	7	29233	5220	7	7	0	7	140	76
10	7.4	19933	5131	19627	5164	0	0	0	12537	2757	307	33	0	226	73	55
32	7.8	20680	3982	20647	3969	13	13	0	16687	1177	33	18	80	33	227	87
4	9.2	12400	na	12400	na	0	na	0	11960	na	0	-	0	0	0	-
6	9.6	15928	na	12868	na	0	na	0	11200	na	3060	na	0	3060	60	na
3	12.2	2440	na	2440	na	0	na	0	2440	na	0	-	0	0	0	-
34	13.5	31140	na	31120	na	20	na	0	15740	na	20	na	0	20	500	na
14	14.0	4740	na	4740	na	0	na	0	4720	na	0	-	0	0	0	-
19	23.5	20013	6854	20007	6855	0	0	0	19687	6799	7	7	0	7	287	66
WC	3.2	15566	4027	13117	2641	0	0	0	8414	1775	2449	1533	2376	73	733	242
908	14.2	27182	2534	27180	2534	0	0	0	26524	2466	2	2	0	2	462	218

		Gastropoda	Gammarus		Isopod	Harpacticoida		Chironomidae		Acarina	Lepidoptera	*Hydra*		Platyhelminthes	No.	Ave.
Site	Depth	Density	Density	S.E.	Density	Density	S.E.	Density	S.E.	Density	Density	Density	S.E.	Density	Species	Diversity
13A	1.4	0	15	15	0	90	90	3075	1548	210	210	30	15	0	18	1.92
17	1.5	0	75	54	0	0	0	6795	3259	840	0	720	357	0	16	1.90
15A	1.9	0	0	0	0	30	30	6405	1724	210	0	45	26	0	12	1.69
18	2.3	0	0	0	0	45	45	4065	2038	375	0	270	135	0	17	2.03
33A	5.6	0	0	0	0	627	323	933	483	27	0	40	31	0	11	1.33
10	7.4	0	0	0	13	5407	3326	1460	1001	60	0	80	53	7	11	1.37
32	7.8	7	0	0	0	3260	2609	413	66	40	0	0	0	0	12	1.31
4	9.2	0	0	0	0	420	na	0	na	20	0	0	na	0	4	0.47
6	9.6	0	0	0	0	148	na	600	na	860	0	0	na	0	13	1.80
3	12.2	0	0	0	0	0	na	0	na	0	0	0	na	0	2	0.21
34	13.5	0	0	0	0	14860	na	0	na	0	0	0	na	0	6	0.68
14	14.0	0	0	0	0	0	na	20	na	0	0	0	na	0	3	0.37
19	23.5	0	0	0	0	7	6.7	20	20	7	0	0	0	0	7	0.49
WC	3.2	6	5	5	0	44	26	3652	726	5	0	0	0	0	19	2.55
908	14.2	0	0	0	0	0	0	67	15	2	0	0	0	0	9	0.99

Table 3. Average wet biomass (mg m^{-2} with shell) and S.E. of benthic invertebrates in Hamilton Harbour during 2002 (May, July, September) and 2003 for sites WC and 908 (May, July, October). Samples without S.E. were only sampled in late September, depths are in meters.

		TOTALS		Non-*Dreissena*		Nematoda		*Hirudinea*	Oligochaeta		*Dreissena* spp.		*D.*	*D.*	Sphaeriidae	
Site	Depth	**Biomass**	**S.E.**	Biomass	S.E.	Biomass	S.E.	Biomass	Biomass	S.E.	Biomass	S.E.	*bugensis*	*polymorpha*	Biomass	S.E.
13A	1.4	5690	3058	3245	1173	3	2	0	2111	692	2445	2087	0	2445	0	0
17	1.5	12851	4551	6137	1129	6	2	0	3492	294	6714	3585	0	6714	0	0
15A	1.9	3712	1914	3463	1678	3	2	0	1686	607	249	242	0	249	78	76
18	2.3	7732	4120	4223	1128	5	0	0	2349	738	3509	3300	0	3509	0	0
33A	5.6	12217	2505	11809	2128	1	1	382	10263	2565	408	408	0	408	147	73
10	7.4	120911	104021	5670	683	0	0	0	4661	1080	115241	104531	329	114912	102	98
32	7.8	9178	1031	9133	1035	1	1	0	6857	1847	45	41	0	45	545	186
4	9.2	7334	na	7334	na	0	na	0	7326	na	0	-	0	0	0	-
6	9.6	6924	na	4278	na	0	na	0	3622	na	2646	na	0	2646	138	na
3	12.2	2544	na	2544	na	0	na	0	2544	na	0	-	0	0	0	-
34	13.5	18668	na	18636	na	2	na	0	17286	na	32	na	0	32	1160	na
14	14.0	384	na	384	na	0	na	0	360	na	0	-	0	0	0	-
19	23.5	18898	6203	18896	6204	0	0	0	18598	6175	2	2	0	2	286	56
WC	3.2	9728	1298	8346	980	0	0	0	3132	450	1382	443	570	812	444	136
908	14.2	21805	2429	21804	2430	0	0	0	21188	2475	1	1	0	1	449	133

		Gastropoda	Gammarus		Isopod	Harpacticoida		Chironomidae		Acarina	Lepidoptera	*Hydra*		Platyhelminthes	No.	Biomass
Site	Depth	Biomass	Biomass	S.E.	Biomass	Biomass y	S.E.	Biomass	S.E.	Biomass	Biomass	mg m^{-2}	S.E.	Biomass	Species	Diversity
13A	1.4	0	2	2	0	2	2	896	442	84	146	3	2	0	18	2.39
17	1.5	0	29	22	0	0	0	1571	616	333	0	60	29	0	16	2.28
15A	1.9	0	0	0	0	2	2	1655	1271	36	0	3	2	0	12	2.50
18	2.3	0	0	0	0	2	2	1643	613	192	0	23	14	0	17	2.48
33A	5.6	0	0	0	0	5	3	1005	434	4	0	1	1	0	11	1.52
10	7.4	0	0	0	4	46	26	840	430	13	0	3	2	1	11	1.26
32	7.8	33	0	0	0	29	27	1664	1094	4	0	0	0	0	12	1.48
4	9.2	0	0	0	0	4	na	0	na	0	0	0	na	0	4	0.51
6	9.6	0	0	0	0	8	na	142	na	0	0	0	na	0	13	2.05
3	12.2	0	0	0	0	0	na	0	na	0	0	0	na	0	2	0.21
34	13.5	0	0	0	0	188	na	0	na	0	0	0	na	0	6	0.73
14	14.0	0	0	0	0	0	na	24	na	0	0	0	na	0	3	0.68
19	23.5	0	0	0	0	1	1	11	11	1	0	0	0	0	7	0.50
WC	3.2	68	1	1	0	1	1	4700	726	3	0	0	0	0	19	-
908	14.2	0	0	0	0	0	0	155	46	1	0	0	0	0	9	-

Table 4. Average seasonal and annual density (m^{-2}) of benthic fauna at site WC in western Hamilton Harbour, 2003.

Site	**WC**		**WC**		**WC**		**Average**	
Depth (m)	3.2		3.7		3.9		Density	
Date	May 14		July 22		Oct 09		2003	
	Ave.	S.E.	Ave.	S.E.	Ave.	S.E.	Ave.	S.E.
TOTALS	**5485**	**2769.6**	**12158**	**381.3**	**28336**	**6344.6**	**15326**	**3938.4**
non Dreissenids (total)	**5441**	**2744.2**	**12129**	**388.9**	**21061**	**3031.7**	**12877**	**2554.2**
OLIGOCHAETA (total)	**2875**	**1457.0**	**8199**	**102.7**	**14168**	**1937.5**	**8414**	**1775.0**
MOLLUSCA:								
DREISSENIDAE:	44	25.4	29	14.7	7275	3389.8	2449	1553.4
Dreissena bugensis	0	0.0	0	0.0	7128	3359.4	2376	1533.6
Dreissena polymorpha	44	25.4	29	14.7	147	38.8	73	23.2
SPHAERIIDAE:								
Pisidium sp.	29	29.3	1320	415.9	851	322.7	733	242.3
Musculium securis	0	0.0	0	0.0	0	0.0	0	0.0
GASTROPODA:								
Amnicola limosa	15	14.7	0	0.0	0	0.0	5	5.5
CRUSTACEA:								
AMPHIPODA:								
Gammarus fasciatus	15	14.7	0	0.0	0	0.0	5	4.9
CLADOCERA:								
Ilyocryptus	0	0.0	59	38.8	0	0.0	19.6	14.9
HARPACTICOIDA:	0	0.0	0	0.0	132	50.8	44.0	26.4
CHIRONOMIDAE (total)	**2508**	**1298.6**	**2552**	**254.0**	**5896**	**889.8**	**3652**	**725.7**
CHIRONOMINI:								
Chironomus sp.	44	25.4	59	38.8	1980	268.8	694	331.0
Chironomus anthracinus	0	0.0	704	250.2	1071	588.9	592	242.5
Chironomus atritibia	0	0.0	0	0.0	233	63.9	235	63.9
Chironomus plumosus	132	50.8	689	172.9	0	0.0	234	117.7
Cladopelma	0	0.0	0	0.0	147	77.6	49	33.2
Cryptochironomus	44	25.4	29	14.7	176	50.8	83	28.8
Dicrotendipes	0	0.0	0	0.0	15	14.7	5	4.9
Endochironomus subtendens								
Glyptotendipes	0	0.0	15	14.7	0	0.0	5	4.9
Microchironomus	59	38.8	191	102.7			83	42.4
Polypedilum halterales	997	651.8	117	29.3	616	225.8	577	236.5
Tribelos jucundus	0	0.0	15	14.7	59	58.7	24	19.6
TANYTARSINI:								
Paratanytarsus	15	14.7	29	29.3	103	63.9	49	24.8
Tanytarsus	249	139.9	44	25.4	103	63.9	132	54.4
TANYPODINAE:								
Procladius	968	462.2	660	88.0	1393	271.6	1007	189.5
MISCELLANEOUS TAXA								
HYDACARINA:	0	0.0	0	0.0	15	14.7	4.9	4.9
Number of species	11.7	1.3	15.0	0.6	18.0	0.6	14.9	1.0
Diversity Index	2.51	0.18	2.49	0.10	2.64	0.01	2.54	0.06

Table 5. Average seasonal and annual wet biomass (g m^{-2} wet+shells) of benthic fauna at site WC in western Hamilton Harbour, 2003.

Site	**WC**		**WC**		**WC**		**Average**	
Depth (m)	3.2		3.7		3.9		Biomass	
Date	May 14		July 22		Oct 09		2003	
	Ave.wt	S.E.	Ave.wt.	S.E.	Ave.wt.	S.E.	Ave.wt.	S.E.
TOTALS	**7.487**	**3.259**	**10.202**	**1.763**	**11.484**	**1.527**	**9.724**	**1.298**
non Dreissenids (total)	**6.907**	**2.841**	**8.457**	**0.823**	**9.664**	**0.930**	**8.342**	**0.980**
OLIGOCHAETA (total)	**1.924**	**0.889**	**3.238**	**0.526**	**4.234**	**0.149**	**3.132**	**0.450**
MOLLUSCA:								
DREISSENIDAE:	0.581	0.481	1.745	0.941	1.820	0.872	1.382	0.443
Dreissena bugensis	0.000	0.000	0.000	0.000	1.710	0.835	0.570	0.373
Dreissena polymorpha	0.581	0.481	1.745	0.941	0.110	0.038	0.812	0.390
SPHAERIIDAE:								
Pisidium sp.	0.019	0.019	0.694	0.229	0.619	0.178	0.444	0.136
Musculium securis	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
GASTROPODA:								
Amnicola limosa	0.180	0.180	0.0	0.0	0.0	0.0	0.068	0.068
CRUSTACEA:								
AMPHIPOD								
Gammarus fasciatus	0.002	0.002	0.0	0.0	0.0	0.0	0.001	0.001
CLADOCERA:								
Ilyocryptus	0.0	0.0	0.003	0.001	0.0	0.0	0.001	0.001
HARPACTICOIDA:	0.0	0.0	0.0	0.0	0.004	0.001	0.001	0.001
CHIRONOMIDAE (total)	**4.78**	**2.22**	**4.522**	**0.920**	**4.796**	**0.743**	**4.700**	**0.726**
CHIRONOMINI:								
Chironomus sp.	0.378	0.218	0.116	0.066	1.355	0.327	0.616	0.221
C. anthracinus	0.0	0.0	1.753	0.642	0.974	0.279	0.909	0.324
C. atritibia	0.0	0.0	0.0	0.0	1.253	0.266	1.253	0.266
C. plumosus	2.687	1.283	1.858	0.527	0.0	0.0	1.515	0.564
Cladopelma	0.0	0.0	0.0	0.0	0.132	0.070	0.044	0.030
Cryptochironomus	0.109	0.059	0.067	0.035	0.324	0.095	0.167	0.052
Dicrotendipes	0.0	0.0	0.0	0.0	0.021	0.021	0.007	0.007
Endochironomus								
Glyptotendipes	0.0	0.0	0.026	0.026	0.000	0.000	0.009	0.009
Microchironomus	0.028	0.020	0.065	0.044			0.031	0.014
Polypedilum halterales	0.239	0.158	0.028	0.006	0.145	0.054	0.137	0.057
Tribelos jucundus	0.0	0.0	0.029	0.044	0.119	0.119	0.049	0.039
TANYTARSINI:								
Paratanytarsus	0.003	0.003	0.015	0.015	0.038	0.024	0.019	0.010
Tanytarsus	0.150	0.082	0.015	0.007	0.035	0.027	0.066	0.032
TANYPODINAE:								
Procladius	1.173	0.606	0.550	0.105	0.400	0.142	0.708	0.217
MISCELLANEOUS TAXA								
HYDACARINA:	0.0	0.0	0.0	0.0	0.010	0.010	0.003	0.003

Table 6. Seasonal average and annual density (m^{-2}) of benthic fauna at site 908 (03) in western Hamilton Harbour, 2003.

Site	HH-908		HH-03		HH-03		HH-03/908	
Depth (m)	14.6		14.2		14.2		Average Density	
Date	May 14		July 22		Oct 09		2003	
	Ave.	S.E.	Ave.	S.E.	Ave.	S.E.	Ave.	S.E.
TOTALS	**23060**	**1518.8**	**32487**	**2219.0**	**25753**	**6822.2**	**27100**	**2538.9**
non Dreissenids (total)	**23060**	**1518.8**	**32487**	**2219.0**	**25747**	**6819.9**	**27098**	**2538.5**
OLIGOCHAETA (total)	**22807**	**1464.5**	**32233**	**2245.0**	**24533**	**6373.9**	**26524**	**2466.4**
MOLLUSCA:								
DREISSENIDAE:								
Dreissena bugensis	0	0.0	0	0.0	0	0.0	0	0.0
Dreissena polymorpha	0	0.0	0	0.0	7	6.7	2	2.2
SPHAERIIDAE:								
Pisidium sp.	120	41.6	93	24.0	1173	433.5	462	217.9
Musculium securis	0	0.0	0	0.0	0	0.0	0	0.0
GASTROPODA:								
Amnicola limosa	0	0.0	0	0.0	0	0.0	0	0.0
CRUSTACEA:								
AMPHIPODA:								
Gammarus fasciatus	0	0.0	0	0.0	0	0.0	0	0.0
CLADOCERA:								
Ilyocrytus sp.	20	11.5	100	30.6	7	6.7	42	17.5
HARPACTICOIDA:	0	0.0	0	0.0	0	0.0	0	0.0
CHIRONOMIDAE (total)	**113**	**24.0**	**60**	**11.5**	**27**	**17.6**	**67**	**15.6**
CHIRONOMINI:								
Chironomus sp.								
C. anthracinus	13	6.7	40	20.0	0	0.0	18	8.5
C. atritibia								
C. plumosus	0	0.0	0	0.0	20	11.5	7	4.7
Cladopelma								
Cryptochironomus	0	0.0	0	0.0	0	0.0	0	0.0
Dicrotendipes								
Endochironomus	0	0.0	0	0.0	7	6.6	2	2.2
Glyptotendipes								
Microchironomus								
Polypedilum	0	0.0	0	0.0	0	0.0	0	0.0
Tribelos jucundus								
TANYTARSINI:								
Paratanytarsus	13	6.7	0	0.0	0	0.0	4	2.9
Tanytarsus	0	0.0	0	0.0	0	0.0	0	0.0
TANYPODINAE:								
Procladius	87	26.7	20	11.5	0	0.0	36	15.6
MISCELLANEOUS TAXA								
HYDRACARINA:	0	0.0	0	0.0	7	6.7	2	2.2
Number of species	9	0.0	8	0.6	7	1.2	8	0.5
Diversity Index	1.134	0.011	0.947	0.075	0.883	0.134	0.988	0.058

Table 7. Seasonal average and annual wet biomass (g m^{-2} wet+shells) of benthic fauna at site 908 (03) in western Hamilton Harbour, 2003.

Site	**HH-908**		**HH-03**		**HH-03**		**HH-03/908**	
Depth (m)	14.6		14.2		14.2		Average Biomass	
Date	May 14		July 22		Oct 09		2003	
	Ave.wt	S.E.	Ave.wt	S.E.	Ave.wt	S.E.	Ave.wt	S.E.
TOTALS	**26.105**	**0.796**	**25.710**	**2.518**	**13.574**	**3.601**	**21.796**	**2.427**
non Dreissenids (total)	**26.105**	**0.796**	**25.710**	**2.518**	**13.570**	**3.599**	**21.795**	**2.427**
OLIGOCHAETA (total)	**25.706**	**0.829**	**25.294**	**2.470**	**12.565**	**3.308**	**21.188**	**2.476**
MOLLUSCA / BIVALVIA:								
DREISSENIDAE:	0.000	0.000	0.000	0.000	0.004	0.004	0.001	0.001
Dreissena bugensis	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
Dreissena polymorpha	0.000	0.000	0.000	0.000	0.004	0.004	0.001	0.001
SPHAERIIDAE: (total)								
Pisidium sp.	0.193	0.041	0.273	0.069	0.881	0.254	0.449	0.133
Musculium securis	0.000	0.000	0.000	0.000	0.000	0.000	0.00	0.000
GASTROPODA:								
Amnicola limosa	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
CRUSTACEA:								
AMPHIPODA:								
Gammarus fasciatus	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
CLADOCERA:								
Ilyocryptus	0.001	0.001	0.002	0.000	0.001	0.001	0.001	0.000
HARPACTICOIDA:	0.000	0.000	0.000	0.000	0.000	0.000	0.000	0.000
CHIRONOMIDAE (total)	**0.205**	**0.086**	**0.141**	**0.046**	**0.119**	**0.119**	**0.155**	**0.046**
CHIRONOMINI:								
Chironomus sp.								
C. anthracinus	0.092	0.050	0.115	0.054	0.000	0.000	0.069	0.028
C. atritibia								
C. plumosus	0.000	0.000	0.000	0.000	0.101	0.101	0.034	0.034
Cladopelma								
Cryptochironomus								
Dicrotendipes								
Endochironomus	0.000	0.000	0.000	0.000	0.019	0.019	0.006	0.006
Glyptotendipes								
Microchironomus								
Polypedilum halterales								
Tribelos jucundus								
TANYTARSINI:								
Paratanytarsus	0.009	0.006	0.000	0.000	0.000	0.000	0.003	0.002
Tanytarsus								
TANYPODINAE:								
Procladius	0.104	0.046	0.027	0.014	0.000	0.000	0.044	0.021
MISCELLANEOUS TAXA								
HYDRACARINA:	0.000	0.000	0.000	0.000	0.004	0.004	0.001	0.001

Table 8. Comparative average density (no. m^{-2}) of *Dreissena* spp. and wet biomass (g m^{-2} with shells).

Location	Year	No. of sites	Density Ave.	Density S.E.	Biomass Ave.	Biomass S.E.
Hamilton Harbour	2002	13	1015.2	503.8	10.1	8.8
Lake Erie	1998	36	5731.0	1398.0	958	292
Upper Bay of Quinte *	1998	64	4712.5	n.a	981	402.1
	2000	96	38865.0	n.a	2438.3	767.7

* area weighted average for Upper Bay of Quinte between Trenton and Telegraph Narrows.

Table 9. Area of depth zones (km^2); number of samples; annual Average Biomass and Standard Error (g m^{-2}); calculated Total Biomass per area (g) and low error (Ave - S.E.); and Area Weighted Biomass (g m^{-2}) of major benthic groups in Hamilton Harbour 2002-2003. Values are all wet shell-free biomass without the shells of the mollusks.

			Average Biomass		Total Biomass		Weighted Biomass	
Depth	Area	n	gm^{-2}	S.E.	Ave x Area	S.E.-Low	Ave	S.E.-Low
			<u>Oligochaetes</u>					
0 - 2	2.4617	9	2.409	0.386	5.9311	4.9808		
2 - 5	1.2369	6	2.741	0.392	3.3900	2.9056		
5 - 10	4.2664	11	6.546	1.154	27.9279	23.0053		
10 - 15	5.1119	6	10.345	5.215	52.8802	26.2236		
15+	7.8728	3	18.598	6.175	146.4182	97.8059		
Sum	**20.9496**				**236.5473**	**154.9210**	**11.291**	**7.395**
			<u>Chironomids</u>					
Depth	Area	n	gm^{-2}	S.E.	Ave x Area	S.E.-Low	Ave	S.E.-Low
0 - 2	2.4617	9	1.441	0.183	3.5478	3.0977		
2 - 5	1.2369	6	3.171	1.528	3.9229	2.0326		
5 - 10	4.2664	11	0.730	0.303	3.1153	1.8218		
10 - 15	5.1119	6	0.448	0.373	2.2916	0.3870		
15+	7.8728	3	0.011	0.011	0.0840	0.0000		
Sum	**20.9496**				**12.9617**	**7.3390**	**0.619**	**0.350**
			<u>Sphaeriidae</u>					
Depth	Area	n	gm^{-2}	S.E.	Ave x Area	S.E.-Low	Ave	S.E.-Low
0 - 2	2.4617	9	0.006	0.006	0.0144	0.0000		
2 - 5	1.2369	6	0.067	0.067	0.0824	0.0000		
5 - 10	4.2664	11	0.186	0.093	0.7953	0.3972		
10 - 15	5.1119	6	0.121	0.082	0.6167	0.1968		
15+	7.8728	3	0.086	0.017	0.6755	0.5430		
Sum	**20.9496**				**2.1843**	**1.1369**	**0.104**	**0.054**
			<u>*Dreissena*</u>					
Depth	Area	n	gm^{-2}	S.E.	Ave x Area	S.E.-Low	Ave	S.E.-Low
0 - 2	2.4617	9	1.808	0.753	4.4514	2.5969		
2 - 5	1.2369	6	1.376	0.595	1.7022	0.9658		
5 - 10	4.2664	11	23.668	22.898	100.9772	3.2836		
10 - 15	5.1119	6	0.005	0.004	0.0238	0.0012		
15+	7.8728	3	0.001	0.001	0.0088	0.0000		
Sum	**20.9496**				**107.1635**	**6.8474**	**5.115**	**0.327**

Table 10. Benthic density (per Ekman grab, 0.05 m^{-2}) at 9 sites in Hamilton Harbour sampled in late September 1964, 1984 and 2002 (also 2003 at Site 3). Depths are those in 1984. An Ekman and 580 micron screen (#30 mesh) were used each year.

	Site 3				**Site 4**			**Site 6**		
	11.0 m				8.0 m			10.0 m		
	1964	1984	2002	2003	1964	1984	2002	1964	1984	2002
NEMATODA:	na	na	0	0	na	na	0	na	na	0
HIRUDINEA										
OLIGOCHAETA (total)	**580**	**880**	**122**	**1326**	**220**	**560**	**598**	**530**	**1140**	**560**
NAIDIIDAE	na	na	na	na	0	18	na		11	na
TUBIFICIDAE:	na	na	na	na			na			na
Immature with hairs	na	na	na	na	0	48	na	0	388	na
Immatures without hairs	na	na	na	na	90	380	na	240	670	na
Limnodrilus cervix					20	23		0	18	
L. claparedianus						28			4	
Limnodrilus hoffmeisteri					110	28		90	36	
Tubifex tubifex					0	7		200	4	
Quistradrilus multisetosus						28			9	
DREISSENIDAE (total)	**0**	**0**	**0**	**0.1**	**0**	**0**	**0**	**0**	**0**	**153**
Dreissena bugensis										
Dreissena polymorpha	0	0	0	0.1	0	0	0	0	0	153
SPHAERIIDAE (total)	**0**	**0**	**0**	**23.1**	**0**	**0**	**0**	**0**	**3**	**3**
Pisidium sp.	0	0	0	23.1	0	0	0	0	3	3
Musculium partumeium										
GASTROPODA:										
AMPHIPODA:										
HARPACTICOIDA:	0	0	0	0	0	0	21	0	0	7
CHIRONOMIDAE (total)	**0**	**0**	**0**	**3.3**	**0**	**0**	**0**	**3**	**1**	**30**
CHIRONOMINI:										
Chironomus attenuatus	0	0	0	0.3	0	0	0	0	0	3
Chironomus atritibia	0	0	0	0.9	0	0	0	0	0	0
Cladopelma	0	0	0	0	0	0	0	0	0	1
Cryptochironomus	0	0	0	0	0	0	0	0	0	7
Endochironomus subtendens	0	0	0	0.1	0	0	0	0	0	0
Glyptotendipes polytomus	0	0	0	0	0	0	0	3	0	1
Polypedilum halterales	0	0	0	0	0	0	0	0	0	16
TANYTARSINI:										
Cladotanytarsus	0	0	0	0	0	0	0	0	0	1
Paratanytarsus	0	0	0	0.2	0	0	0	0	1	0
Tanytarsus stellatus										
ORTHOCLADIINAE:										
Thienemanniella										
TANYPODINAE:										
Procladius	0	0	0	1.8	0	0	0	0	0	1
HYDRACARINA:	0	0	0	0.1	0	0	1	0	0	43
Number of species	2	2	2	8	4	8	4	5	10	13
Diversity Index	0.157	0.147	0.208	0.988	0.556	1.106	0.467	0.637	1.277	1.796
TOTALS	**580**	**880**	**122**	**1353**	**220**	**560**	**620**	**533**	**1144**	**796**

Continued on next page

Table 10 . Continued. Benthic density per Ekman grab (0.05 m^{-2}) in 1964, 1984 and 2002.

	Site 10 6.0 m			Site 14 13.0 m			Site 19 23.0 m		
	1964	1984	2002	1964	1984	2002	1964	1984	2002
NEMATODA:	na	na	0	na	na	0	na	na	0
HIRUDINEA:									
OLIGOCHAETA (total)	**230**	**970**	**403**	**100**	**1320**	**236**	**76**	**1480**	**861**
NAIDIIDAE:	na	na	na	na	na	na	0	0	na
TUBIFICIDAE:	na	na	na	na	na	na			na
Immature with hairs	na	na	na	na	na	na	0	258	na
Immatures without hairs	na	na	na	na	na	na	4	1063	na
Limnodrilus cervix							5	0	
L. claparedianus							0	4	
Limnodrilus hoffmeisteri							65	76	
Tubifex tubifex							2	22	
Quistradrilus multisetosus							0	57	
DREISSENIDAE (total)	**0**	**0**	**17**	**0**	**0**	**0**	**0**	**0**	**1**
Dreissena bugensis									
Dreissena polymorpha	0	0	17	0	0	0	0	0	1
SPHAERIIDAE (total)	**0**	**0**	**9**	**0**	**0**	**0**	**0**	**2**	**11**
Pisidium sp.	0	0	9	0	0	0	0	0	11
Musculium partumeium	0	0	0	0	0	0	0	2	0
GASTROPODA:									
AMPHIPODA:									
HARPACTICOIDA:	0	0	236	0	0	0	0	0	1
CHIRONOMIDAE (total)	2	2	173	0	0	1	0	0	3
CHIRONOMINI:									
Chironomus attenuatus	2	2	155	0	0	0	0	0	3
Chironomus atritibia									
Cladopelma									
Cryptochironomus	0	0	1	0	0	0	0	0	0
Endochironomus subtendens									
Glyptotendipes polytomus									
Polypedilum halterales	0	0	8	0	0	0	0	0	0
TANYTARSINI:									
Cladotanytarsus									
Paratanytarsus	0	0	2	0	0	0	0	0	0
Tanytarsus stellatus									
ORTHOCLADIINAE:									
Thienemanniella	0	0	1	0	0	0	0	0	0
TANYPODINAE:									
Procladius	0	0	6	0	0	1	0	0	0
HYDRACARINA:	0	0	7	0	0	0	0	0	1
Number of species	3	3	10	2	3	3	5	7	7
Diversity Index	0.367	0.291	1.368	0.217	0.278	0.366	0.924	0.821	0.885
TOTALS	**232**	**972**	**845**	**100**	**1320**	**237**	**76**	**1488**	**878**

Continued on next page

Table 10. Continued. Benthic density per Ekman grab (0.05 m^{-2}) in 1964, 1984 and 2002.

	Site 32			Site 33			Site 34		
	7.8 m			6.0 m			12.0 m		
	1964	1984	2002	1964	1984	2002	1964	1984	2002
NEMATODA:	0	na	2	0	0	3	0	0	1
HIRUDINEA:									
OLIGOCHAETA (total)	**170**	**na**	**952**	**56**	**1640**	**1847**	**0**	**1590**	**787**
NAIDIIDAE	na	na	na	0	206	na	0	na	na
TUBIFICIDAE:	na	na	na			na		na	na
Immature with hairs	na		na	0	258	na	0	na	na
Immatures without hairs	na		na	30	996	na	0	na	na
Limnodrilus cervix	na			13	45		0		
L. claparedianus					13				
Limnodrilus hoffmeisteri	na			7	32		0		
Tubifex tubifex	na			6	32				
Quistradrilus multisetosus					58				
DREISSENIDAE (total)	**0**	**0**	**3**	**0**	**0**	**0**	**0**	**0**	**1**
Dreissena bugensis									
Dreissena polymorpha	0		3	0	0	0	0	0	1
SPHAERIIDAE (total)	**0**	**na**	**20**	**0**	**0**	**1**	**0**	**1**	**25**
Pisidium sp.	0		20	0	0	1	0	0	25
Musculium partumeium	0		0	0	0	0	0	1	0
GASTROPODA:									
AMPHIPODA:									
HARPACTICOIDA:	0		422	0	0	56	0	0	743
CHIRONOMIDAE (total)	**0**	**na**	**27**	**5**	**17**	**95**	**0**	**0**	**0**
CHIRONOMINI:									
Chironomus attenuatus	0		22	5	16	87	0	0	0
Chironomus atritibia			1						
Cladopelma									
Cryptochironomus	0		0	0	0	1	0	0	0
Endochironomus subtendens									
Glyptotendipes polytomus									
Polypedilum halterales									
TANYTARSINI:									
Cladotanytarsus									
Paratanytarsus									
Tanytarsus stellatus	0		0	0	0	1	0	0	0
ORTHOCLADIINAE:									
Thienemanniella	0		0	0	1	0	0	0	0
TANYPODINAE:									
Procladius	0		4	0	0	6	0	0	0
HYDRACARINA:	0	na	5	0	0	4	0	0	0
Number of species	2	0	10	6	10	10	0	3	6
Diversity Index	0.195	0.00	1.239	1.216	1.214	1.184	0.00	0.271	0.680
TOTALS	**170**	**na**	**1431**	**61**	**1657**	**2006**	**0**	**1591**	**1557**

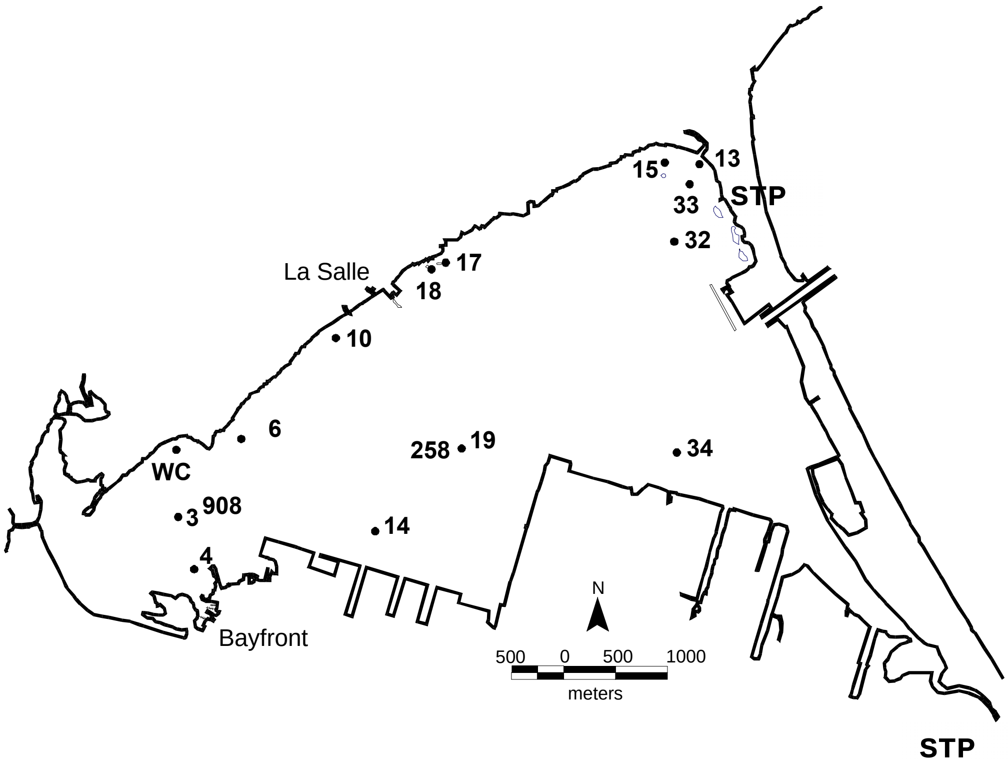

Fig. 1. Site locations of benthic sampling during 2002 – 2003.

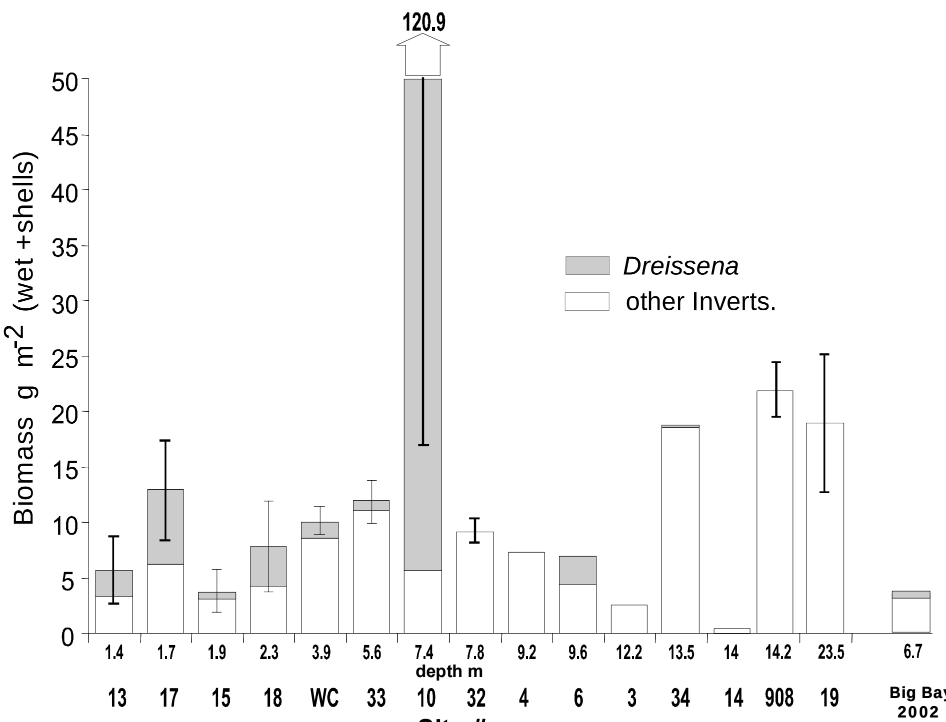

Fig. 2. Average total wet benthic biomass (of all species) and non-*Dreissena* wet biomass (with shells) in Hamilton Harbour between May and September 2002 -2003.

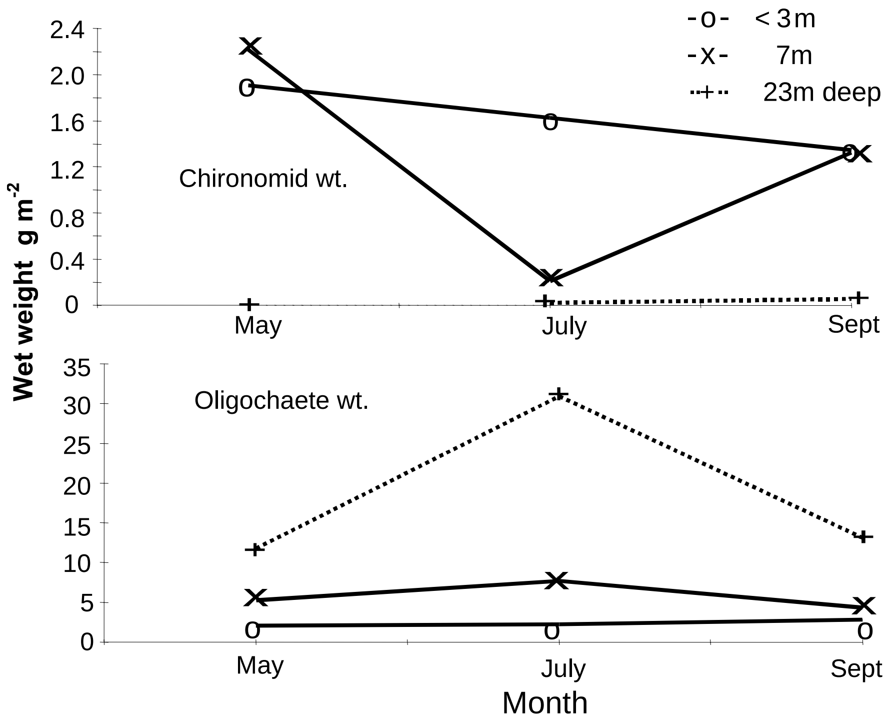

Fig. 3. Chironomid and oligochaete wet biomass (g m^{-2}) at three depth zones in Hamilton Harbour from samples collected in May, July and September 2002.

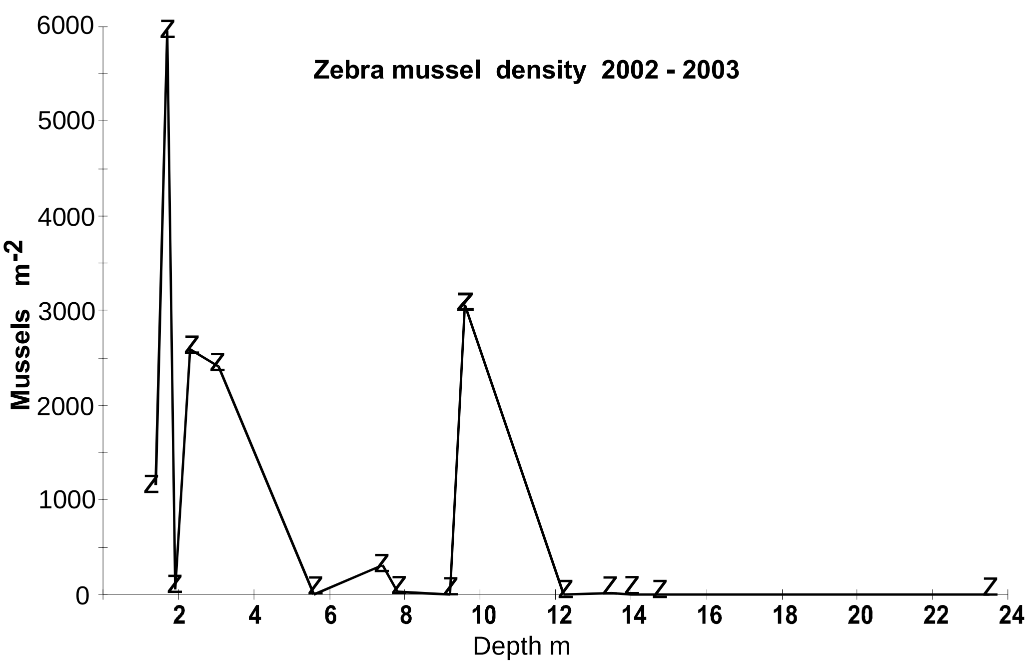

Fig. 4. Depth distribution of *Dreissena* mussels (Z) in Hamilton Harbour during 2002 - 2003.

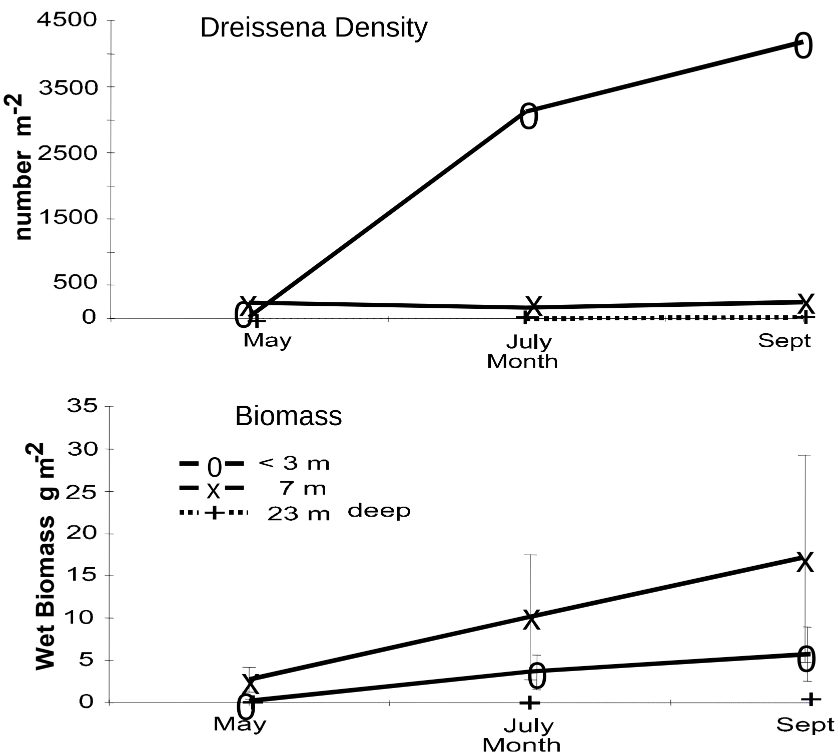

Fig. 5. Seasonal density and wet biomass (g m^{-2} with shells) of *Dreissena* spp. in three depth zones of Hamilton Harbour from samples collected in May, July and September 2002.

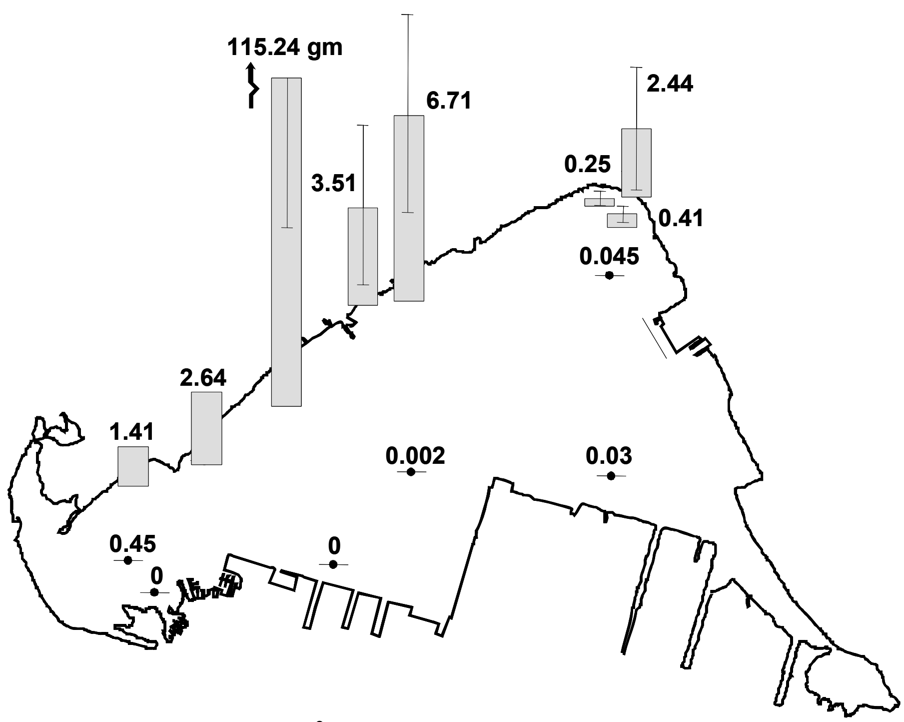

Fig. 6. Average wet biomass (g m^{-2} with shells) of *Dreissena* spp. at the benthic sites between May and September 2002.

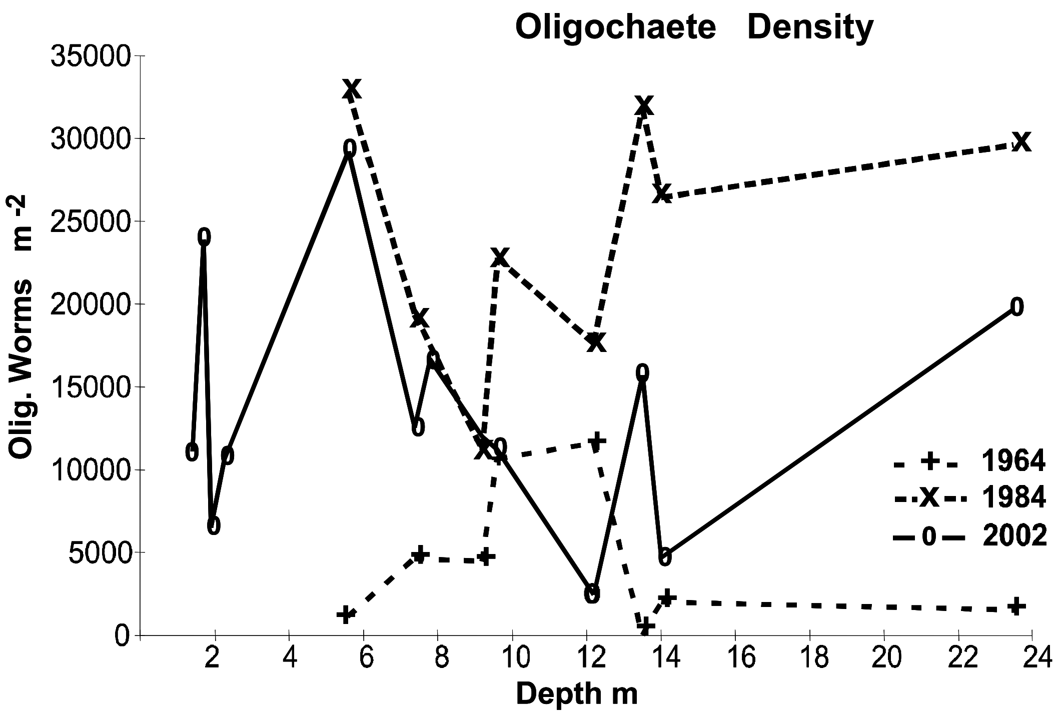

Fig. 7. Average density of oligochaetes (May, July, September) in Hamilton Harbour during 2002, and their density at the same sites in 1964 and 1984.

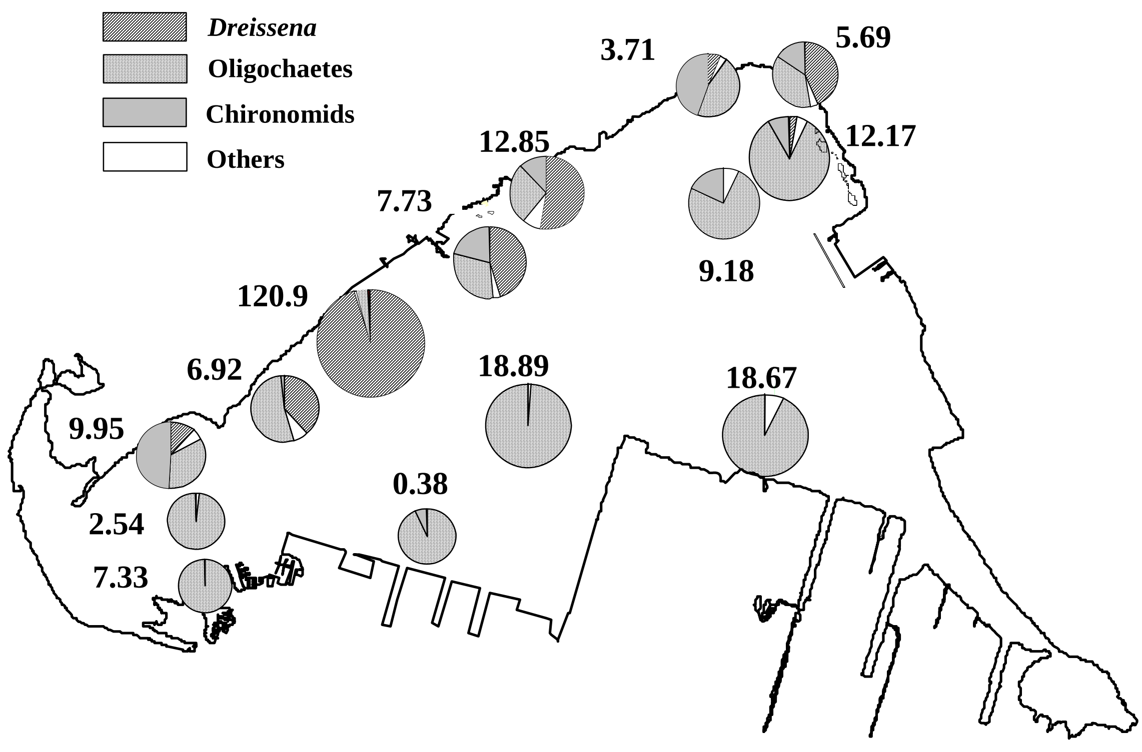

Fig. 8 Benthic biomass (g m^{-2} wet with shells) and composition of the major taxa groups in Hamilton Harbour September 2002.

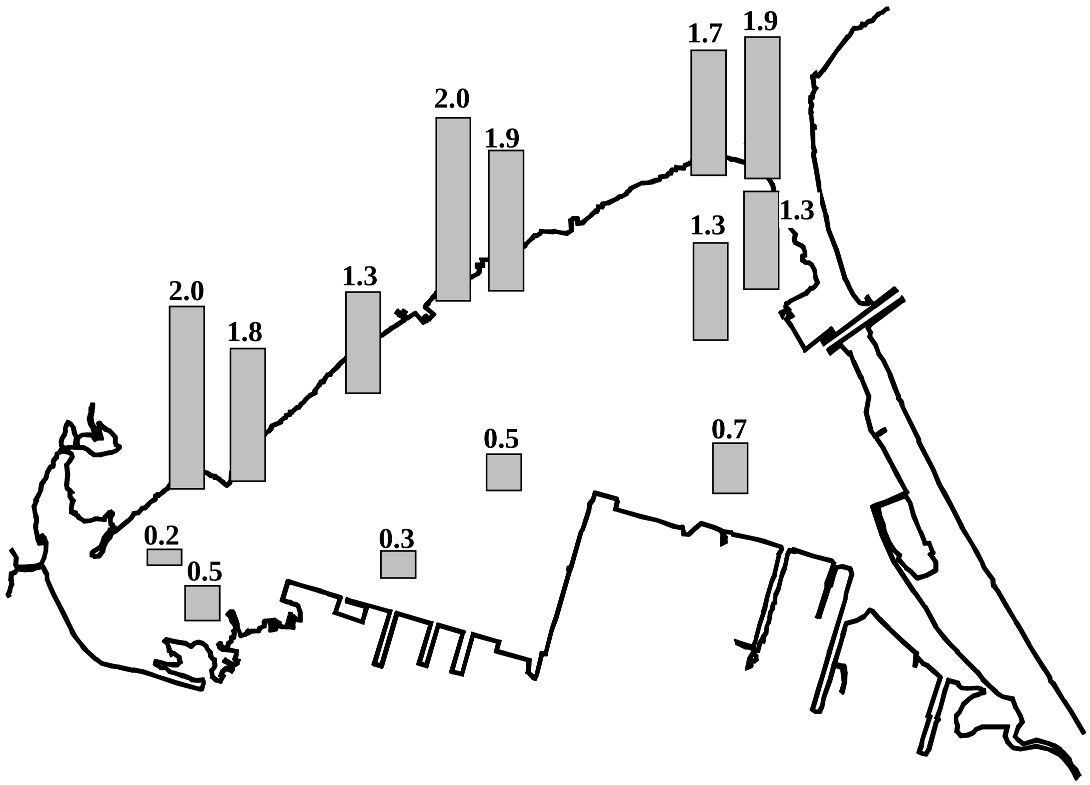

Fig. 9 . Average species diversity of the benthic fauna at the benthic sites in Hamilton Harbour sampled between May and October 2002.