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An assessment of Sediment Quality and Benthic Invertebrate Community Structure in Collingwood Harbour

By:
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AN ASSESSMENT OF SEDIMENT QUALITY AND BENTHIC INVERTEBRATE COMMUNITY STRUCTURE IN COLLINGWOOD HARBOUR. AN AREA OF CONCERN: TOWARDS DE-LISTING

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Management Perspective

Environmental managers and regulatory decision-makers have traditionally set water and sediment quality guidelines based on chemical concentrations. The primary advantage of a chemical approach is the apparent ease of simple numerical comparison of concentrations of chemicals found in environmental matrices with levels of these same compounds known to cause a toxic response in biota. However, the chemical approach has been criticized in recent years because it frequently fails to achieve its objectives or because it is so excessively rigorous that it has limited value. It also has proved to be problematic in delisting Areas of Concern for sediment contamination.

This report describes the development of biological objectives for sediments in nearshore habitats in the North American Great Lakes using a modification of the technique developed in the U.K. A large data base has been assembled from reference sites in Lakes Ontario, Erie, Michigan, Superior and Huron and includes information on, (1) the structure of the benthic invertebrate communities, (2) measured environmental variables and (3) the responses of four species of benthic invertebrates (*Hyalella azteca, Chironomus riparius, Hexagenia* spp. and *Tubifex tubifex*) exposed in the laboratory to sediment collected from the same sites. Benthic invertebrates were selected as the most appropriate biological indicators because they are most directly associated with contaminants in sediments through their feeding and behavioral activities.

This study was undertaken to demonstrate the utility of biological guidelines as an alternative to guidelines and criteria for sediments in the Laurentian Great Lakes based solely on comparisons of bulk chemical concentrations of contaminants in sediments. The results of using this method in Collingwood Harbour show the value of this approach in delisting Areas of Concern and providing a more relevant and realistic method for determining environmental impact.

Abstract

This report describes the first results for an alternative approach to the development of sediment quality criteria in the nearshore areas of the Laurentian Great Lakes. It is derived from methods developed in the United Kingdom for establishing predictive relationships between macroinvertebrate fauna and the physico-chemistry of riverine environments. The technique involves a multivariate statistical approach using a) data on the structure of benthic invertebrate communities, b) functional responses (survival, growth and reproduction) in four sediment toxicity tests (bioassays) with benthic invertebrates and c) selected environmental variables at 96 reference ('clean') sites in the nearshore areas of all five Great Lakes (Lakes Superior, Huron, Erie, Ontario and Michigan). Two pattern recognition techniques (using the computer software package PATN) are employed in the analysis: cluster analysis and ordination. The ordination vector scores from the original axes from the pattern analysis are correlated (using CORR in SAS) with environmental variables which are anticipated to be least affected by anthropogenic activities (*e.g.*, alkalinity, depth, silt, sodium, etc.). Multiple discriminant analysis (MDA) is used to relate the site groupings from the pattern analysis to the environmental variables and to generate a model which can be used to predict community assemblages and functional responses at new sites with unknown but potential contamination. The predicted community assemblages and functional responses are then compared with the actual benthic communities and responses at a site and the need for remedial action is determined.

An example of the use of the model for sediment in Collingwood Bay (an area of concern designated by the IJC in Georgian Bay, Lake Huron) is presented and the technique is shown to be more precise in determining the need for remediation than the currently used provincial sediment quality criteria based on Screening Level Concentration (SLC) and laboratory toxicity tests. The ultimate goal of the study is the development of a method to determine the need for, and the success of, remedial action and to predict what benthic communities should look like at a site if it were clean and what responses of organisms in sediment toxicity tests constitute an acceptable endpoint.

Introduction

In 1977, the Ontario Ministry of the Environment reported to the International Joint Commission (IJC) that Collingwood Harbour was an environmental problem due to nuisance algal growth and periodically high bacterial levels. Furthermore, sediment contamination was limiting dredging activity, consumption of fish was restricted and there was a perceived degradation of harbour aesthetics. In 1986, the Ontario Ministry of the Environment and Environment Canada responded to the IJC's request for a Remedial Action Plan (RAP) for all the Areas of Concern (AOC) by establishing a RAP team for Collingwood Harbour. Among the RAP teams responsibilities, are the provision of detailed descriptions of the environmental problems in the Areas of Concern and a description of the degree of impairment.

In 1989, the Government of Canada, announced a $125 million Great Lakes Action Plan (GLAP) to meet its obligations under the 1987 Protocol to the Canada-US Great Lakes Water Quality Agreement (GLWQA). Contaminated sediments were targeted by two of the three programmes under GLAP (*i.e..,* Cleanup Fund and Preservation Fund) as a major priority. Through the provision of funding by both Cleanup and Preservation Funds, Environment Canada has developed assessment methods for contaminated sediments and is in the process of developing biological sediment guidelines for the Great Lakes. One of the pilot projects of the Cleanup Fund was to address the issue of contaminated sediments and the need for remediation in Collingwood Harbour.

The National Water Research Institute (N.W.R.I.) of Environment Canada, with the support of the Cleanup Fund and the Collingwood Harbour RAP team, conducted extensive sampling of the sediments in Collingwood Harbour in 1992 and 1993. This report presents the results of those investigations and provides a spatial description of the state of the sediments in Collingwood Harbour together with an assessment of the degree of contamination.

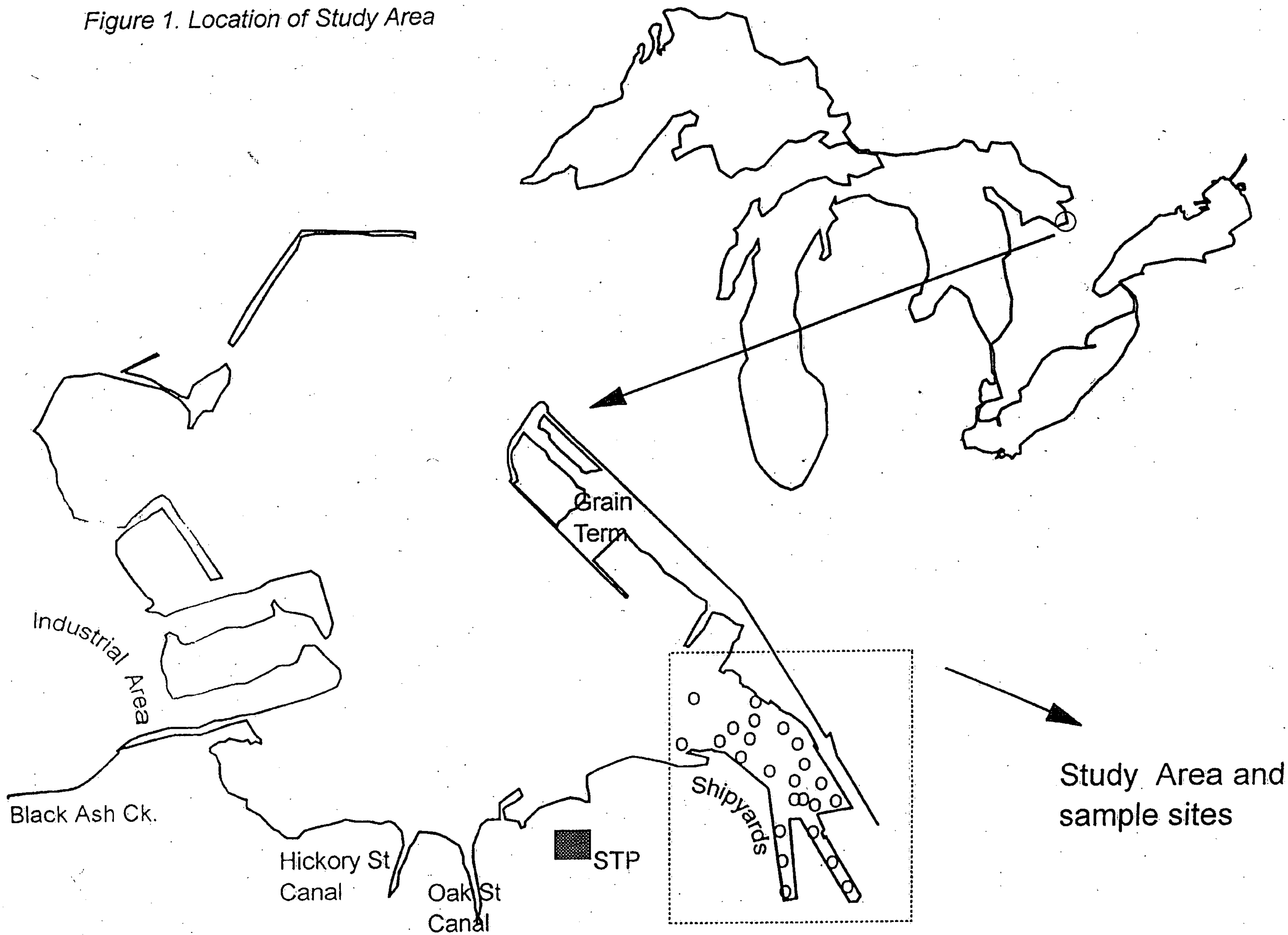
Description of Collingwood Harbour

Collingwood Harbour is situated on the south shore of Nottawasga Bay which forms the southernmost portion of Lake Huron's Georgian Bay (Fig 1). The harbour has an area of 0.8 km^2 with a maximum depth of 6.4 m and an estimated volume of 28.7 x 10^{-4} km^3. Three tributaries drain into Collingwood Harbour, the largest of which is Black Ash Creek which empties into the south east corner of the harbour. Oak Street Canal and Hickory Street Canal are intermittent sources which receive storm sewer discharge and enter at the south shore of the harbour. The harbour is surrounded by the town of Collingwood (1988 pop'n. 12,200) and the land use pattern in the vicinity of the harbour has changed in recent years from industrial to residential/recreational. The primary industry was shipbuilding which was a source of metal contamination in the harbour. However, the shipyards are no longer in use and shipbuilding has been discontinued. Thus, the major source of sediment contamination in the harbour is largely historical and the major contributors are no longer active.

Overview of Current Methods for Sediment Assessment

A number of approaches using either chemical or biological measures or both have been used to assess sediment contamination in North America. Chemical analyses are most frequently performed on bulk sediment for total concentrations of contaminants (*e.g.*, total PCBs, PAHs, etc.). Analyses are less frequently conducted on pore or interstitial water and differentiation of chemical species or congeners is infrequently performed either in bulk sediments or interstitial waters. To assess the extent of contamination and need for remedial action chemical concentrations are compared to background concentrations (pre-historic), predetermined objectives, criteria or standards for water or sediment quality, or to levels known to produce biological effects. The various methods used to assess sediment contamination include the equilibrium partitioning approach (EqP), the screening level concentration (SLC) approach and the apparent effects threshold (AET) (Persaud *et al.* 1992, Zarull and Reynoldson 1992). While each of these approaches to developing sediment objectives or criteria have their own advantages and disadvantages (Giesy and

Figure 1. Location of Study Area



Hoke 1990), they collectively suffer from the need to infer biological impact rather than direct measurements of biological effect (Reynoldson and Zarull 1993). Although most of these approaches will provide a high degree of protection to the aquatic ecosystem, they are unlikely to prove adequate justification for the expensive removal of extensive volumes of sediment, the treatment of in-place pollutants, or other methods of sediment mitigation, as part of a comprehensive remedial action plan for nearshore areas. It has been shown by Painter (1992) that many sediments in areas of the Great Lakes which are well removed from source(s) of contamination, contain concentrations of priority pollutants which exceed both the low and severe effects concentrations for aquatic biota. Most municipal, provincial or federal budgets do not have the capacity for the extensive remedial action which would be required to deal with this amount of sediment nor is there necessarily a need for such a degree of remedial action.

Since the fundamental reason for the development of guidelines, objectives or criteria for sediment is to provide and protect a sustainable and reproducing aquatic biota, alternative methods for developing such numerical data using biological assessments are needed. Biological assessments usually employ either structural or functional measures of impact or both. In the case of sediments, benthic invertebrates are the most appropriate group of organisms to use. Their populations are relatively stable in time with life cycles of one or more years and their taxonomy can be determined at least to the level of genus, and, in some cases, species. In addition, their response(s) to environmental change have been extensively studied.

Several authors have proposed the use of benthic invertebrates in the assessment and management of contaminated sediments. For example, Chapman and co-workers (Chapman and Long 1983; Long and Chapman 1985; Chapman 1986) have described the sediment triad approach, which incorporates aspects of sediment chemistry, toxicity testing and analysis of the community structure of the benthos. The International Joint Commission has suggested a sediment management strategy that incorporates both assessment and remediation (IJC 1987a, 1988). The United States is embarking on a programme to examine various approaches to sediment assessment and remediation and Canada is also addressing these issues via the Great Lakes Action Plan. However, there are still major objections to the employment of the structure of the benthic community to set sediment criteria or objectives. The main criticisms are their lack of universality (*i.e.*, they are, of neccessity, site-specific) and the inability of researchers to establish quantitative objectives for their application (*i.e.*, what should the community `look' like?). Universal guidelines may not be possible due to the very nature and complexity of sediment-contaminant-biota relationships and the diversification of biological and geological components over a large regional area.

Recent developments in the analysis of biological data using multivariate statistical techniques have shown extremely promising results in interpreting changes in community structure based on simple environmental parameters. As a first step towards the identification of the best achievable community for a specific type of habitat, there is a need to define reference communities based on chemical, physical, geological and geographical features in areas free from contamination. The greatest effort in this direction has been made in the United Kingdom (Wright *et al.* 1984; Armitage *et al.* 1987; Moss *et al.* 1987), but similar studies have been conducted in North America (Corkum and Currie 1987), continental Europe (Johnson and Wielderholm 1989) and South Africa (King 1981). An extensive collection of data at non-polluted sites in the U.K. has resulted in the identification of a number of natural communities (Wright *et al.* 1984). Environmental variables such as latitude, substrate type, temperature and depth, were used to correctly predict the benthic communities at 75% of 268 sites, and, at more than 90% of the sites, the observed community was similar to either the predicted, or the next most similar predicted community. At lower levels of community detail, even greater accuracy of prediction was observed. Other studies have shown a similar predictive capability (see Table 1) and the accuracy of prediction in these studies ranges from 68% - 90%, in habitats varying from large lakes to small streams and on three continents. In other words, based on a few physical variables, the expected community assemblage(s) at a location can be defined from a predictive model. Such predicted communities or key species in the communities can be used to establish site-specific guidelines, which may be compared with the actual species composition. Thus, determinations on whether or not the guideline is being met can be made.

Table 1. Ability to predict benthic invertebrate community structure.

Habitat	No. sites	No. Community Groups	Predictor variables	Prediction success rate	Reference
Rivers	286	16	28	76.1	Wright et al 1984
Streams	79	5	26	70.9	Corkum & Currie 1987
Rivers	45	6	5	68.9	Ormerod & Edwards 1987
Rivers	54	5	9	79.6	King 1981
Lakes	68	7	11	90.0	Johnson & Wiederholm 1989
Rivers	35	5	6	75.4	Reynoldson & Zarull 1993
Lakes	95	6	9	90.0	Reynoldson et al in press

To date, this approach has only been used on community assemblages with species as the classifying variables. There is no reason why the same approach cannot be used with both structural and functional variables such as reproduction or growth being included in the multivariate design to classify both predictable communities and responses to toxicity. This is the approach that we have used in the following study.

Overview of Study

We chose to sample 335 reference sites in the nearshore regions of the various basins of the Great Lakes to establish a reference data base for determining whether or not a benthic community is impaired and to establish the expected response in whole-sediment toxicity tests. This reference data base will be used to provide the control or background conditions for sites in Canadian Areas of Concern in the Great Lakes and will encompass the range of variability in data inherent in any ecological study. At the time of this report, data from 218 of 335 sites are available for comparison with the data from Collingwood Harbour.

For comparison of community structure, a multivariate approach has been used. With this method, we have shown that it is possible to predict with an accuracy of 90%, the expected benthic community at any site within the range covered by the reference sites (Reynoldson *et al* in press), from a few environmental variables. The predicted community represents the assemblage of organisms that should be found at an unimpacted site. From the responses of four species of benthic invertebrates (*Hyalella azteca*, *Chironomus riparius*, *Hexagenia* spp. and *Tubifex tubifex*) in whole-sediment exposure studes to field collected sediment from 158 reference sites, we have been able to develop sediment-specific targets for each test endpoint that indicates sediments which are non-toxic.

Methods

The following approach was used as a demonstration of the practical application of this multivariate approach to determining the need for remedial dredging in Collingwood Harbour, Georgian Bay, Lake Huron. This harbour has been identified as an "Area of Concern" (IJC 1987b), in part, because of contaminants in the sediments which exceed the Province of Ontario's sediment guidelines (Persaud *et al.*, 1992). Twenty-five sites were sampled and compared with the reference sites.

Reference Sites

The study area for the reference sites encompassed the entire basin of the five Great Lakes. The multivariate analytical approach requires a large number of sites to be sampled in order to perform the mathematical algorithms and to ensure that the range of habitat characteristics are adequately represented. A preliminary list of 250 (now 335) sites were identified and stratified among the 17 ecoregions described by Wickware and Rubic (1989) for the Great Lakes. These ecoregions are defined from characteristics such as climate, vegetation, bedrock geology, flora and fauna, *etc.* The reference sites were selected to represent "unpolluted" conditions within an ecodistrict and the inclusion of each site required it to meet the following criteria: location a minimum of 10 km away from known discharges as described in the Ontario Intake and Outfall Atlas (OMOE 1990); within 2 km from shore and at a depth of less than 30 m (with the exception of Lake Michigan) and; have known or suspected fine-grained substrate. These sites have been sampled over a three-year period (1991-93). In Collingwood Harbour, 25 sites were sampled (Fig. 1) using the same protocols as the reference sites.

The location of each site was established in the field using either Loran C or a hand-held Geographical Positioning System (GPS). At each reference site, samples were taken of sediment, water and pore-water for chemical and physical analysis; in addition, samples were collected for the community structure of the benthic macroinvertebrates and for whole-sediment exposures in the laboratory with selected species of benthic invertebrates. Each reference site was sampled once in late summer or early fall over a three-year period. In addition, a sub-set of sites (10%) were sampled in each of the three field years, and four sites have been sampled monthly over two years. This will allow a subsequent determination of the effects of both annual and seasonal variation on the outcome of the predictions of community structure and toxicity and will be the subject of later publications.

Environmental Variables

A list of the variables measured at a site is presented in Table 2. Samples for water chemistry were taken using a Van Dorn sampler from 0.5 m above the sediment-water interface. A one litre sample was stored at 4°C prior to analysis of total phosphorus, Kjeldahl nitrogen, nitrate-nitrite and alkalinity at the National Water Research Laboratory in Burlington, Ontario, Canada. On the remainder of the sample, pH, dissolved oxygen, and temperature were measured in the field.

Table 2. *Summary of measured environmental variables and abbreviations used (note: those in bold were used as predictors in Multiple Discriminant Analysis (MDA)).*

Field (5 variables)	Water (mg.l) (5 variables)	Sediment (ug.g dry wt) (32 variables)
Latititude Longitude **Water Depth (m)** Dp **Oxygen (mg.l)** OXW **Bottom Temp (oC)** TMW **pH** PHW	**Alkalinity (mg.l)** AKW **T. phosphorus (mg.l)** TPW **Kjeldahl Nitrogen(mg.l)**TKN **Nitrate-nitrite(mg.l)** NOW Ammonia(mg.l)	**Silica Si** Titanium **Alumimium Al** **Iron Fe** **Manganese Mn** **Magnesium Mg** **Calcium Ca** **Sodium NA** **Potassium K** **T. Nitrogen TN** **T. Phosphorus TP** **T. Org. Carbon TOC** **Loss on ignition** LOI Selenium **Vanadium V** Chromium Cobalt Nickel Copper Zinc Arsenic Strontium Yttrium Molybdenum Silver Cadmium Tin Lead **% Gravel GR** **% Sand SN** **% Silt SL** **% Clay** CL

Sediment and sediment pore-water were characterized from samples taken from a mini-box corer. The mini-box core takes a 40 X 40 cm section of sediment to a depth of 25-30 cm. Samples for geochemical analysis were taken from the surface 2 cm of the box core. After sampling, the sediment was homogenized in a glass dish with a nalgene spoon. The sample was divided as follows:

An aliquot of sediment for organic contaminants was placed into a hexane-prewashed glass bottle with a hexane rinsed aluminum foil liner. Samples were sealed and stored frozen (or at 4°C in the field) for subsequent freeze-drying and storage. These samples were not normally analyzed but were archived in the event of a site being suspected as contaminated.

Samples for the determination of particle size distribution were placed into a plastic pill jar and stored at ambient temperature in the field. Upon return to the laboratory, samples were freeze-dried and analyzed following the method of Duncan and LaHaie (1979).

The remaining sediment in the glass dish was stored in a 500 mL plastic container at 4°C in the field and shipped to the National Water Research Laboratory for freeze drying and analyses of metals, major ions and nutrients.

Invertebrate Community Structure

Samples for the identification and enumeration of benthic invertebrates were taken by inserting five 10 cm plexiglass tubes (i.d. 5.5 cm) into the sediment in the box core. Each core tube is considered a replicate sample unit and was removed and the contents placed into a plastic bag and kept cool until sieved. The contents of each bag were sieved through a 250μm mesh in the field as quickly as possible after sampling. If sieving could not be done in the field, 4% formalin was added to the bag and the replicate samples were stored at 4°C and sieved as soon as possible thereafter. After sieving the samples were placed in plastic vials (50 mL) and preserved with 4% formalin. Replicates with large amounts of organic material were placed in larger containers and again preserved with 4% formalin. After 24 h the formalin was replaced by ethanol.

Samples were sorted with a low power stereo microscope (100-400X) and identified to the species or genus level where possible using appropriate identification guides. As required (Chironomidae and Oligochaeta) slide mounts were made for high power microscopic identification. Type specimens of all identified specimens were submitted to experts (R.O. Brinkhurst for Oligochaeta; B. Bilyj and D. Oliver for Chironomidae; G. Mackie for Mollusca) for confirmation. The confirmed type specimens are being maintained as a reference collection.

Whole-Sediment Toxicity Tests

A mini-ponar sampler was used to obtain five replicate field samples of sediment for laboratory bioassays with four species of invertebrates. Each replicate sample was placed in a plastic bag and held at 4°C until tests could be conducted. Tests were conducted, in sets of six to seven, over a period of approximately six months. A clean control sediment from Long Point, Lake Erie, was also tested with each set of samples to provide biological quality assurance. Complete details of the culture of organisms and conditions for each toxicity test with *C. riparius* and *T. tubifex* are described elsewhere (Reynoldson *et al.* 1991, Day *et al.* 1994, Reynoldson *et al.* 1994). Culture of *H. azteca* was conducted according to the procedure described in Borgmann *et al.* (1989). Eggs of the mayfly, *Hexagenia* spp. (both *H. limbata* and *H. rigida*), were collected during late June and July in 1991 and 1992 according to the method of Hanes and Ciborowski (1992) and organisms were cultured using the procedure of Bedard *et al.* (1992).

Tests with *H. azteca*, *C. riparius* and *T. tubifex* were conducted in 250 mL glass beakers containing 60 to 100 mL of sieved (500 µm mesh), homogenized sediment with approximately 100 to 140 mL of overlying carbon-filtered, dechlorinated and aerated Lake Ontario water (pH 7.8 to 8.3, conductivity 439 to 578 µohms/cm, hardness 119 to 137 mg/L). Tests with the mayfly, *Hexagenia* were conducted in 1 L glass jars with 150 mL of test sediment and 850 mL overlying water. The sediment was allowed to settle for 24 h prior to addition of the test organisms. Tests were initiated with the random addition of 15 organisms per beaker for *H. azteca* and *C. riparius*, 10 organisms per jar for *Hexagenia* spp. and 4 organisms per beaker for *T. tubifex*. Juveniles of *H. azteca* were 3 to 7 d old at test initiation; *C. riparius* larvae were first instars and were approximately 3 d post-oviposition; *Hexagenia* nymphs were 1.5 to 2 months old (approximately 5 to 10 mg wet weight) and *T. tubifex* adults were 8 to 9 weeks old. Tests were conducted at 23±1°C with a 16L:8D photoperiod. Tests were static with the periodic addition of distilled water to replace water lost during evaporation. Each beaker was covered with a plastic petri dish with a central hole for aeration using a Pasteur pipette and air line. Dissolved oxygen concentrations and pH were measured at the beginning, middle and end of each exposure period. Tests were terminated after 10 d for *C. riparius*, 21 d for *Hexagenia* and 28 d for *H. azteca* and *T. tubifex* by passing the sediment samples through a 500 µm mesh sieve. Sediment from the *T. tubifex* test was passed through an additional 250 µm mesh sieve at test completion. Endpoints measured in the tests were survival and growth for *C. riparius*, *Hexagenia* spp. and *H. azteca* and for *T. tubifex* survival and production of cocoons and young. Mean dry weights of *H. azteca*, *C. riparius* and *Hexagenia* spp. were determined after drying the surviving animals from each treatment replicate as a group to a constant weight in a drying oven (60°C).

Data Analysis

Data analysis was based on the procedure described by Wright *et al.* (1984). Pattern analysis was used to describe the biological structure of the data at the reference sites and correlation and multiple discriminant analysis (MDA) was conducted to relate the observed biological structure to the environmental characteristics.

Classification of Biological Data

The biological structure of the data was examined using two pattern recognition techniques, cluster analysis and ordination. The mean values from the five replicates for the taxonomic numerical data were used as descriptors of the benthic invertebrate community. These community data are not transformed and the raw scores are used because numeric differences are thought to be important community descriptors. The Bray and Curtis association measure was used because it performs consistently well in a variety of tests and simulations on different types of data (Faith *et al.* 1987). Clustering of the reference sites was done using an agglomerative hierarchical fusion method with unweighted pair group mean averages (UPGMA). The appropriate number of groups was selected by examining the group structure and, particularly, the spatial location of the groups in ordination space. Ordination was used to reduce the variables required to identify the structure of the data. A multi-dimensional scaling (MDS) method of ordination was used, *i.e.*, Semi- Strong- Hybrid multidimensional scaling. Multi-dimensional scaling methods use rank order rather than metric information and, thus, avoid the problematic assumptions of linearity inherent in many ordination techniques (Belbin 1991). This is of particular value when relating ordination scores to environmental characteristics. All clustering and ordination was done using PATN (Belbin 1993), a pattern analysis software package developed by CSIRO in Australia.

Correlation of Biological Data with Environmental Characteristics

Of the 42 environmental variables measured in this study, 27 (Table 2) were examined for their relationship with the biological structure of the data. We excluded those variables most likely to be influenced by anthropogenic activity, particularly those associated with sediment contamination. Thus, all the metals were excluded from consideration as potential predictor variables. The variables used were general descriptors of sediment type such as the major elements, particle size and organic material as a potential indicator of nutritive quality. These variables, together with physical attributes such as water depth and general water chemistry, were considered as the most appropriate general habitat descriptors less likely to be modified by human activity.

The relationship of the environmental variables with the biological data was examined by correlation analysis of the environmental characteristics with ordination vector scores using the procedure CORR in SAS.

Prediction of Biological Groupings

Environmental variables were selected for use in multiple discriminant analysis (MDA) to relate the biological site groupings to the environmental characteristics of the sites based on correlation analysis. The SAS version of MDA was used with raw environmental data to generate discriminant scores, and to predict the probability of group membership.

Spatial Pattern of Sites in Collingwood Harbour

Sediment Chemistry

Surficial sediment samples were taken from 25 sites in Collingwood Harbour (Fig. 1) to establish the range of sediment conditions. The results are presented in Appendices 1a and 1b. Multivariate pattern analysis was used to identify the spatial distribution of contaminants and the existence of contamination gradients. From the cluster analysis (Fig. 2a), it is apparent that there are two groups of sites, a smaller group of 4 sites (#6706 - #6709) and a much larger group of 21 sites. This larger group has a small sub-group of 5 sites (#6705, #6706, #6710, #6712, and #6725) that appear to be distinct and have been investigated separately. When the sites are plotted in ordination space (Fig. 2b) using two vectors (stress 0.0361), the four sites comprising Gp 3 were clearly separate. The sites defined as being Gp 2 were separated from the Gp 1 sites which formed a tight group The close association of sites in Gp1 suggests that they are very similar in respect to their physiochemical characteristics.

The relative contribution of each of the variables to the ordination and each vector is indicated in Table 3. Iron, zinc, and lead are best correlated with the total structure of the data; however, the major elements and arsenic, silver and selenium, separate sites on the first vector (Fig 2b) and sand and silt on the second vector. The three groups of sites appear to be separated on the second vector with 'within group' differences being important on the first vector.

Figure 2a. Collingwood H. sediment chemistry dendrogram.

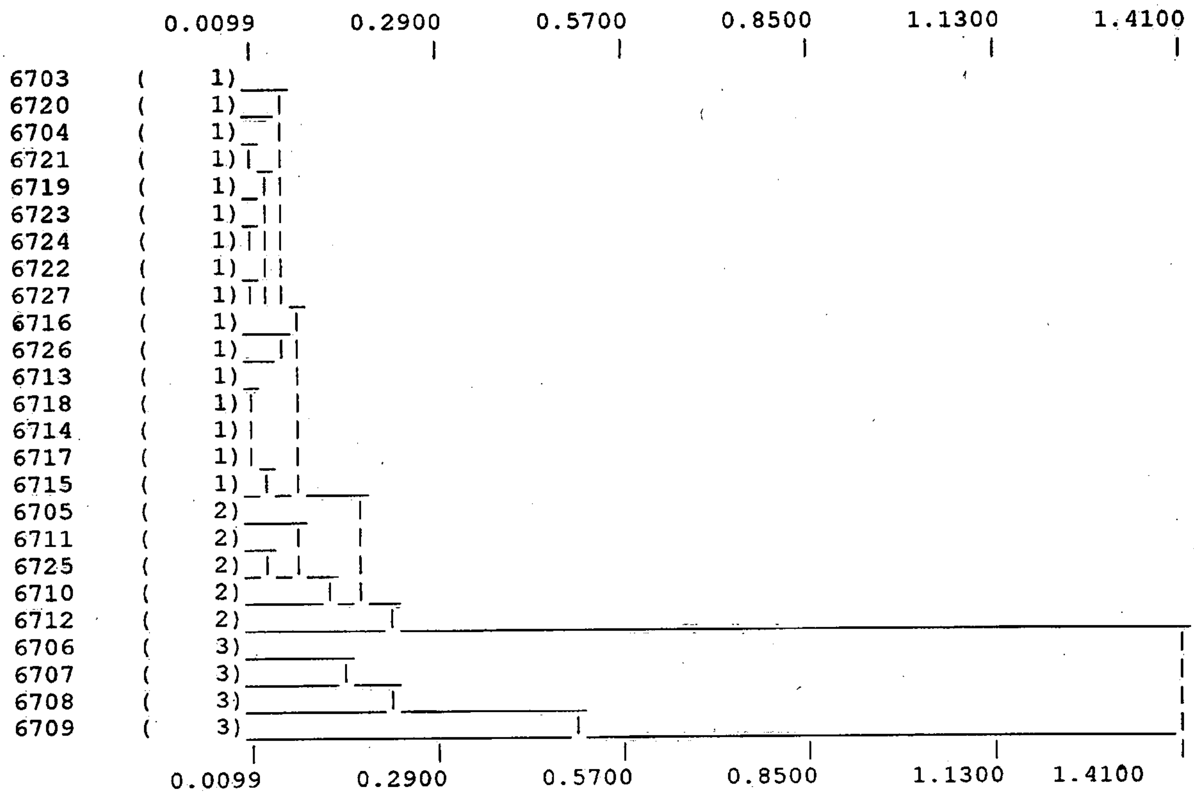


Table 3. Relative contribution of physio-chemical variables to SSH ordination.

Vector 1		Vector 2		Overall Correlation	
Mg	0.999	Sand	0.996	Fe_2O_3	0.986
Al_2O_3	0.995	V	0.927	Zn	0.981
SiO_2	0.991			Pb	0.979
Na_2O	0.986			Co	0.978
				Cu	0.976
Ag	-0.994	Silt	-0.997	Cd	0.974
As	-0.948	LOI	-0.955	Cr	0.970
Sn	-0.935	TN	-0.911		

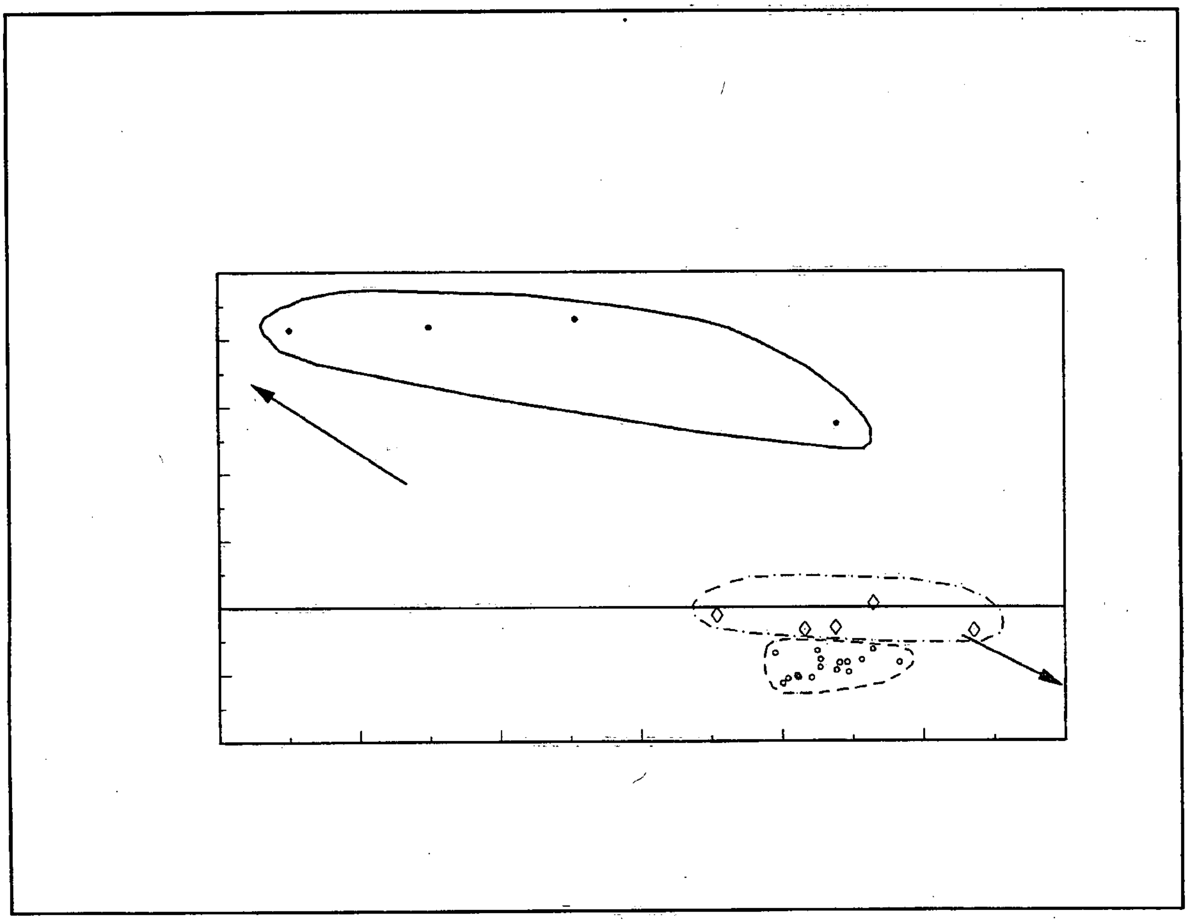
The mean values for the variables which contribute to the structure in the data are presented in Table 4. The sites comprising Gp 3 have high concentrations of zinc, copper, cobalt and lead, elevated concentrations of iron, chromium, tin, arsenic and barium and lower concentrations of total nitrate, phosphorus and loss on ignifition (LOI). The five sites comprising Gp 2 have somewhat elevated levels of zinc, lead, copper and barium.

The spatial distribution of the three groups of sites (Fig. 3a) shows that the sites with the highest metal concentrations (Gp 3) are restricted to the east boat slip and to one site in the mouth of the west slip. The Gp 2 sites are located in the inner west slip, the area of the harbour just outside both boat slips and one site (# 6712) on the western edge of the sampling area outside the boat slips. This chemical data suggests that the two boat slips are the major source of chemical (metal) contamination to Collingwood Harbour.

Table 4. Mean values for selected variables in each of three site groups.

Variable	Gp 1 (16 sites)	Gp 2 (5 sites)	Gp 3 (4 sites)
Fe_2O_3	3.3	4.1	15.0
Zn	161	429	8851
Pb	87	179	709
Co	7.2	11.8	156.8
Cu	41.8	111.2	2504.8
Cd	0.4	0.8	3.7
Cr	23.1	29.0	63.8
Sn	10.0	10.0	126.0
As	9.1	5.3	84.2
Ag	0.3	0.3	1.8
Ba	46.8	79.6	168.2
TN	2288	2129	1489
P_2O_5	0.3	0.3	0.1
Ni	20.7	22.4	31.8
LOI	22.9	22.2	18.1

Figure 2b. Ordination of Collingwood Harbour Sediment Chemistry.



Sediment Toxicity

Eight endpoints for four species of benthic invertebrates were used to examine the toxicity of the sediments in Collingwood Harbour. Survival and growth of the chironomid, *C. riparius*, the mayfly, *Hexagenia* spp. and the amphipod, *H. azteca* was measured as well as reproduction (the number of cocoons and young produced) in the oligochaete worm, *Tubifex tubifex*. These eight endpoints were analyzed in the same manner as the chemical data to determine whether or not a pattern existed in the overall distribution of sediment toxicity. Again from cluster analysis, three major groups of sites can be determined (Fig. 4a).

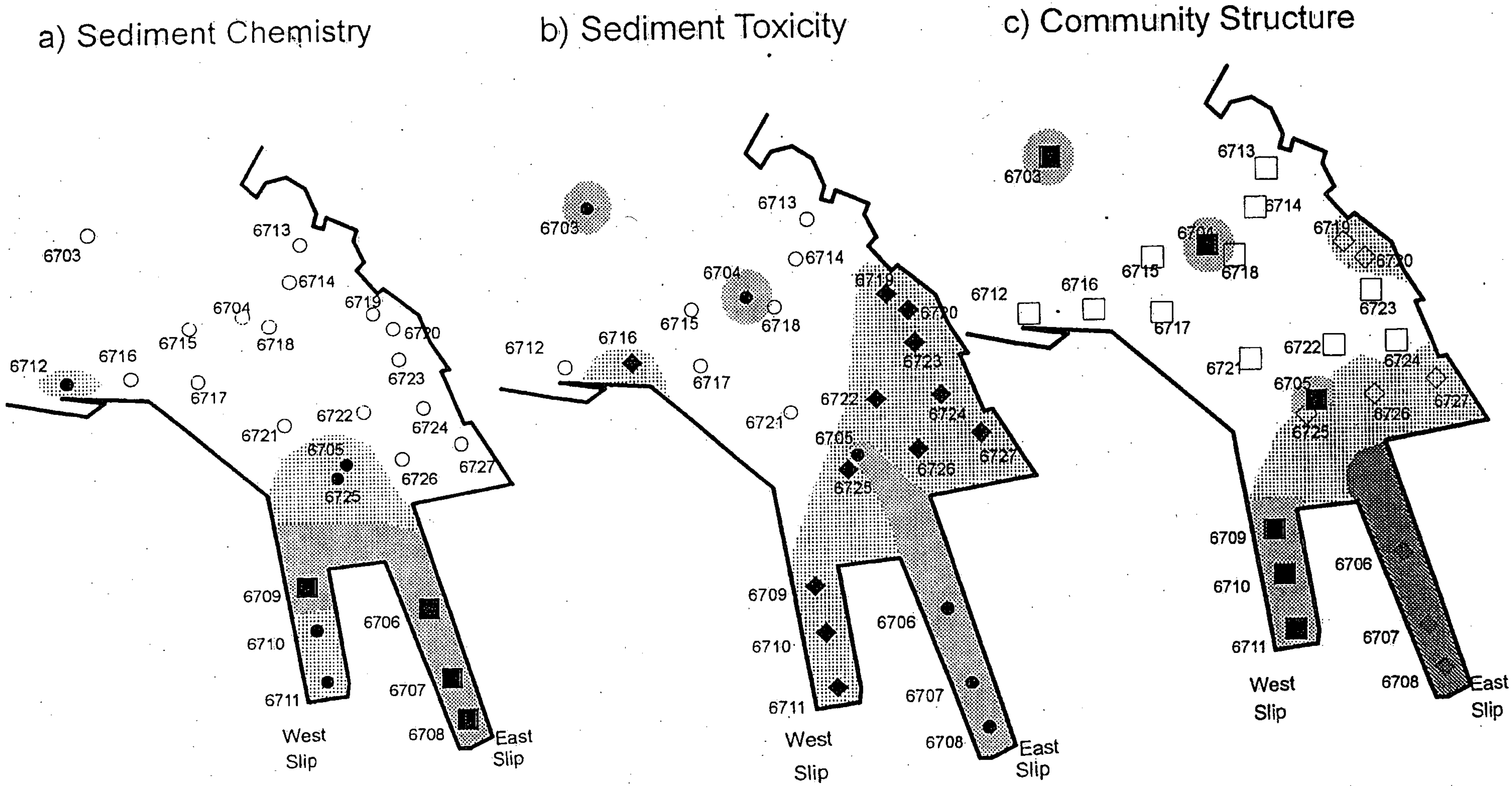


Figure 3. Spatial distribution of site groups formed from cluster analysis of sediment chemistry and toxicity and invertebrate community structure.

Fig 4a. Collingwood Harbour- Dendrogram showing the clustering of sites with related toxicity.

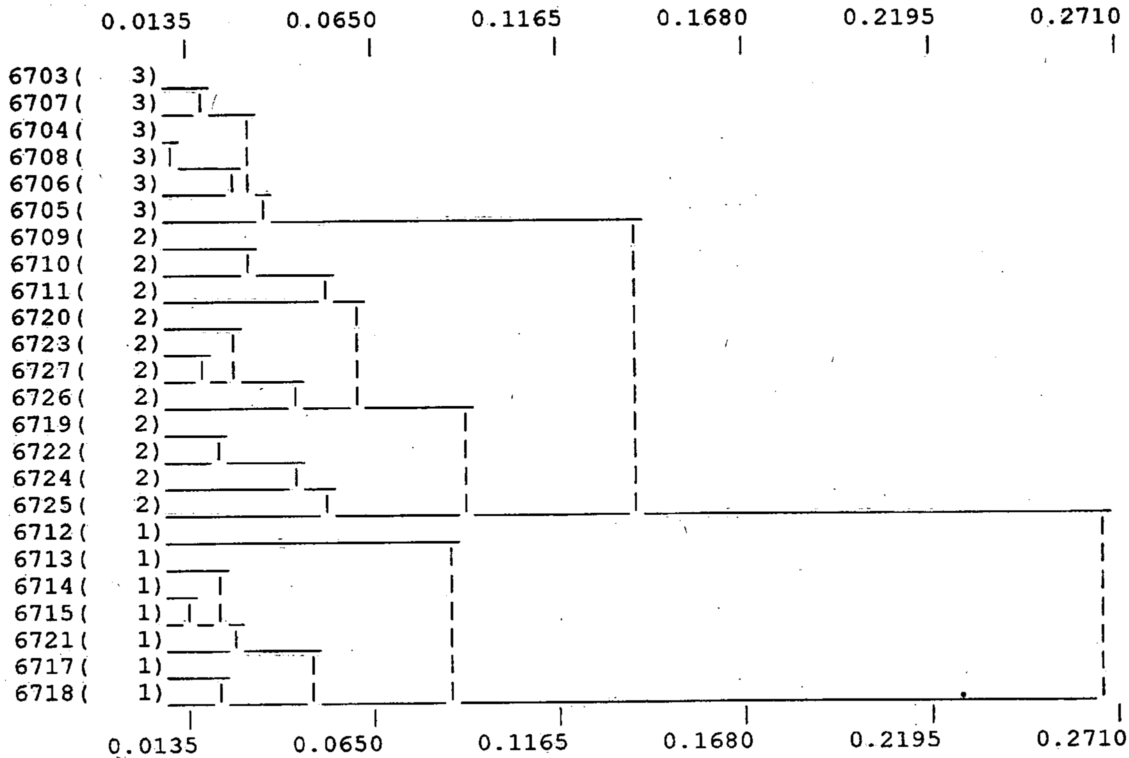


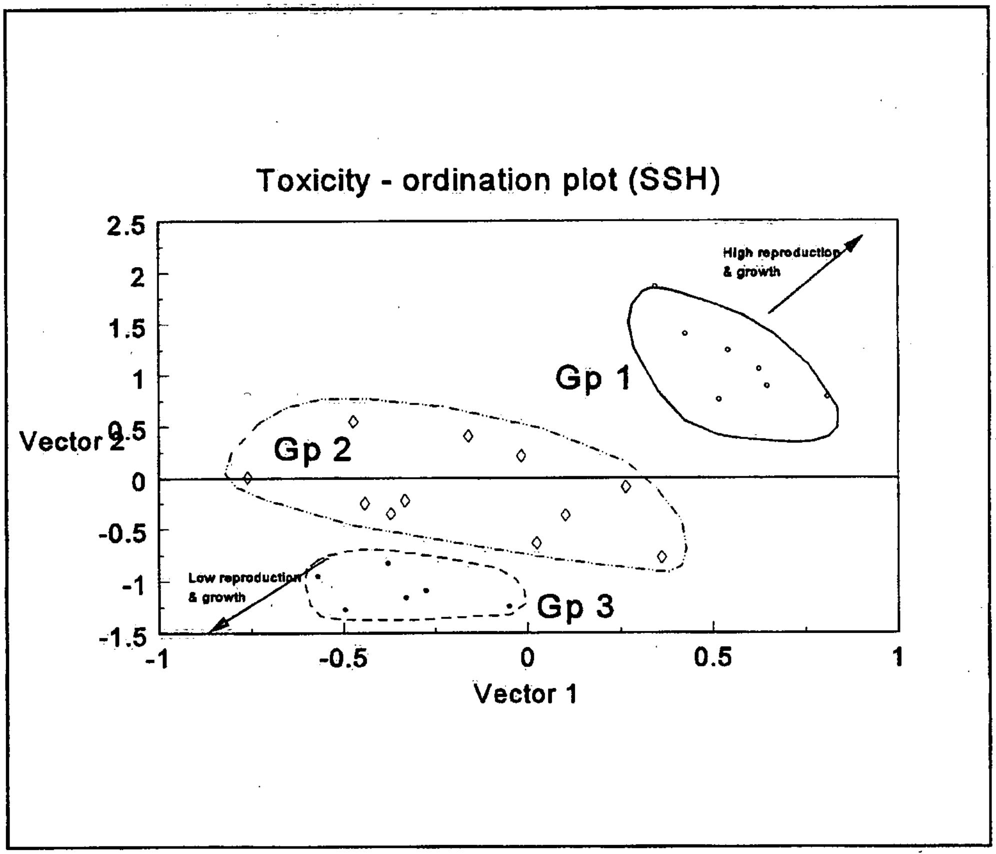
Table 5. Contribution of bioassay endpoints to ordination score.

Vector 1		Vector 2		Correlation	
Chironomus growth	0.912	*Hexagenia* growth	0.998	*Tubifex* young	0.989
Hyalella survival	0.861	*Tubifex* cocoons	0.988	*Chironomus* survival	0.859
		Hyalella growth	0.876	*Tubifex* cocoons	0.857
Chironomus survival	-0.897	*Tubifex* young	0.835	*Hexagenia* growth	0.845
				Chironomus growth	0.791
				Hexagenia survival	0.607
				Hyalella growth	0.598
				Hyalella survival	0.322

Most of the variation in response was in the number of young produced by *T. tubifex* (Table 5) which was the most important contributor to the results from the ordination analysis.

Of the remaining endpoints, all but survival and growth of the amphipod, *H. azteca*, and survival of the mayfly, Hexagenia, were important contributors to the ordination vector scores. The ordination showed (Fig. 4b) a clear distinction between the sites contributing to the three groups formed in the cluster analysis.

Figure 4b. Ordination of toxicity data from Collingwood Harbour.



The mean values for the endpoints measured in the whole sediment toxicity tests indicated that production of young by *T. tubifex* was the endpoint most affected by sediment contamination. For example, average number of young per four adult worms ranged from 139.4 in Gp 1 to only 24.3 in Gp 3. Cocoon production was similarly reduced at those sites comprising Gp 3 (Table 5). Survival of *C. riparius* was an important component of the first vector (Fig. 4b) but there was little difference in this endpoint among the three groups of sites. Growth of *C. riparius* as well as the mayfly, *Hexagenia* was reduced in sediments forming the Gp 3 sites. Survival and growth of *H. azteca* varied little among the three site groups

Table 6. Mean values for bioassay endpoints in three site groups formed from cluster analysis.

Endpoint	Gp 1 (7 sites)	Gp 2 (11 sites)	Gp 3 (6 sites)
C. riparius			
Survival	85.3	85.9	81.8
Growth	0.57	0.43	0.38
H. limbata			
Survival	95.7	97.6	98.0
Growth	10.59	7.00	6.32
H. azteca			
Survival	88.4	87.2	92.1
Growth	0.76	0.64	0.60
T.tubifex			
Cocoons	44.5	38.3	24.3
Young	139.4	54.8	22.2

The spatial distribution of sites in the three groups (Fig. 3b) is similar to that of the chemical groups. The most affected sites (Gp 3) are again those in the east slip, with an area in the west slip and the south east corner forming Gp 2.

We have been able to establish the expected range in responses of all four species of benthic invertebrates in 'clean' sediments from our reference data base (Table 7). For four of the endpoints (*i.e.*, survival and growth of *C. riparius*; growth of *H. azteca*; and survival of *Hexagenia*), a single value for each endpoint is sufficient as they did not appear to respond to sediment characteristics in the 168 reference sites examined to date. These reference sites show a wide range of sediment characteristics (Table 8) and therefore give an indication of the robustness of the responses in these bioassays. Other endpoints, *e.g.,* cocoon production and number of young in *T. tubifex,* survival of *H. azteca* and growth of *Hexagenia* spp., are affected by normal variations in sediment quality. However, these responses can be separated into several types of response which correlate with several environmental attributes (*e.g.* LOI or particle size distribution).

Table 7. Suggested criteria based on mean -1 SD from reference sites, test values should exceed value to be considered non-toxic.

Endpoint	Values for survival (%), growth (mg) and reproduction to be exceeded to indicate no sediment toxicity.	Values for survival (%), growth (mg) and reproduction to be exceeded to indicate no sediment toxicity.
C.riparius		
survival	75.5%	
growth	0.27 mg wet weight	
Hexagenia limbata	If LOI <12.5%, silt < 40%, sand > 25%	If LOI >12.5%, silt > 40%, sand < 25%
survival	89.9%	89.9%
growth	1.4 mg wet weight	3.9 mg wet weight
Hyalella azteca	If silt < 20%, LOI < 10%	If silt > 20%, LOI > 10%
survival	21.1%	81.0%
growth	0.34 mg wet weight	0.34 mg wet weight
Tubifex tubifex (unfed)	If LOI < 12%	If LOI > 12%
cocoons	27.2	27.2
young	25.4	58.6
Tubifex tubifex (fed)	If TN < 1700, TOC < 1.8	If TN > 1700, TOC > 1.8
cocoons	27.6	39
young	68.9	80.5

Table 8. Range of selected sediment characteristics in 258 reference sites.

Variable	Minimum	Maximum
Sand (%)	0.0	90.5
Silt (%)	0.0	86.3
Clay (%)	0.0	91.1
LOI (%)	1.0	38.7
TOC (ppm)	0.10	8.21
TN (ppm)	67	8911

When the results for the whole sediment exposure tests from Collingwood Harbour with four species of benthic invertebrates (Table 9) are compared with the criteria derived from the reference sites (Table 7), two of the test endpoints, *i.e.*, growth in both *C. riparius* and *H. azteca*, are well within the range observed for the reference sites and show no evidence of toxicity. Growth of the mayfly, *Hexagenia* spp., is just below the reference site value of 3.9 mg dry wt./ind. at one site (#6709). Survival of these three species may indicate a slightly toxic response at select sites. For example, % survival of *C. riparius* was reduced at sites #6706 and # 6709, two sites located within the east and west boat slips where concentrations of metals were above the severe effects limit set by the OMEE. In three species two sites have slightly reduced survival. However, the major effects of sediment contamination are on reproduction in *T. tubifex*. For example, cocoon production was reduced in sediments collected from 24% of the sites and total number of young indicated toxicity in sediments from 68% of the sites. The sites showing the most impairment relative to reference sediments, and the greatest number of endpoints not meeting the reference site criteria, are three in the east slip (# 6706, # 6707 and # 6708) and one in the outer portion of the west slip (# 6709).

Table 9. Collingwood H. sites, mean values for eight toxicity tests endpoints (values below criteria are shown in bold).

Site	CrSu	CrGw	HxSu	HxGw	HaSu	HaGw	TtCc	TtYg
6703	88.0	0.38	98	6.87	89.3	0.72	**22.4**	**22.0**
6704	81.3	0.38	98	8.11	94.7	0.75	**24.4**	**22.8**
6705	80.0	0.43	100	5.78	90.0	0.66	**21.8**	**14.6**
6706	**72.0**	0.39	98	5.62	93.3	0.50	**27.2**	**22.4**
6707	86.6	0.33	100	6.17	90.7	0.42	**25.8**	**28.8**
6708	82.6	0.36	94	5.35	94.7	0.53	**24.4**	**22.3**
6709	**68.0**	0.46	100	**3.86**	94.7	0.53	36.4	**42.4**
6710	78.6	0.40	100	5.04	88.0	0.60	31.8	**42.2**
6711	85.3	0.40	100	4.56	84.0	0.50	29.0	**55.8**
6712	89.3	0.52	**80**	10.12	94.7	0.70	44.7	171.7
6713	76.0	0.51	100	10.93	88.0	0.66	43.5	125.5
6714	86.6	0.61	100	10.59	84.0	0.74	45.6	135.4
6715	85.3	0.49	100	10.94	82.7	0.76	43.0	126.0
6716	90.6	0.65	100	10.58	89.3	0.79	na	na
6717	89.3	0.64	94	10.40	86.7	0.87	43.4	153.6
6718	86.6	0.66	98	11.16	93.3	0.82	46.6	146.8
6719	88.0	0.45	100	8.56	92.0	0.62	40.0	**74.2**
6720	90.6	0.56	100	6.70	89.3	0.74	43.8	**36.6**
6721	84.0	0.58	98	9.97	89.3	0.74	45.0	117.2
6722	88.0	0.55	98	9.11	**77.3**	0.71	42.8	**74.2**
6723	96.0	0.35	100	7.98	90.7	0.71	39.2	**42.8**
6724	76.0	0.51	98	8.53	90.7	0.66	42.4	**62.4**
6725	90.6	0.37	**84**	8.34	94.7	0.61	38.4	**77.8**
6726	92.0	0.34	96	7.98	**72.0**	0.75	39.4	**48.8**
6727	92.0	0.38	98	6.28	85.3	0.56	37.8	**45.8**
No. Exceedences	2	0	2	1	2	0	6	17

Community Structure

Forty-three species of benthic invertebrates have been identified from the 25 stations sampled in Collingwood Harbour (Appendix 2). The relative contribution of the classes found are shown in Figure 5. Two classes of oligochaetes (worms), the Naididae and the Tubificidae, are the dominant groups of organisms found in the area followed by the Porifera (sponges) and the Chironomidae (midge larvae). Each ofl the other groups of organisms comprise less than 5% of the total number of organisms found.

The distribution patterns of the community were examined for the 22 dominant species (those that each contributed to more than 0.05% of the community) using cluster analysis and ordination. From the cluster analysis, we examined the distribution of sites in the first four groups (Fig. 6) which were distinct in ordination space (Fig. 7). The four groups of sites are separated on the second vector and 5 taxa are important contributors to the separation of these groups. Variation between sites and within the groups distinguished by the classification analysis, occurs on the first vector where a large variety of species contribute to the community (Table 10).

Table 10. Contribution of species to ordination vectors.

Vector 1		Vector 2		Correlation	
Cladopelma spp	0.994	Platyhelminthes	-0.999	Tubificidae (no hr)	0.819
M. tardigradum	0.982	*L. cervix*	0.966	*V. intermedia*	0.796
Cryptochironomus spp	-0.956	*L. claparedianus*	-0.964	Tubificidae (hr)	0.754
S. ferox	0.949	*L. hoffmeisteri*	-0.951	*Cladopelma* spp	0.673
Cladotanytarsus spp	-0.945	*S. josinae*	-0.912	Platyhelminthes	0.654
A. leydigi	-0.940			*L. claparedianus*	0.620
A. pigueti	-0.924			*Procladius* spp	0.589
Procladius spp	-0.912				

The contribution of each species to the ordination (Table 10) shows that immature tubfiicidae (with and without hair chaetae), the naid oligochaete, *Vejdovskyia intermedia* and the midge larvae, *Cladopelma* spp., contribute the most to the ordination scores, but the vectors themselves are best represented by three chironomid species, the tardigrade,

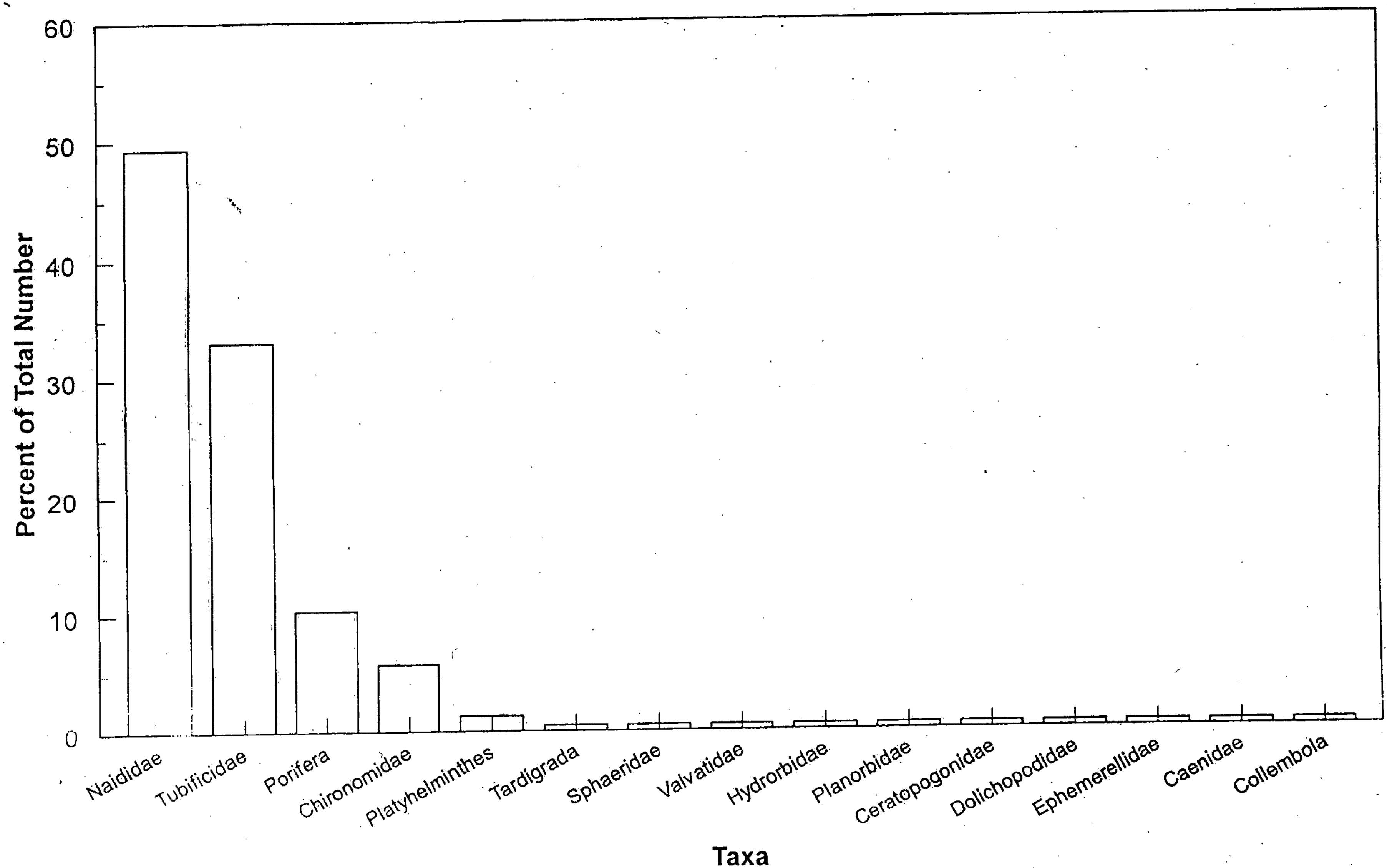
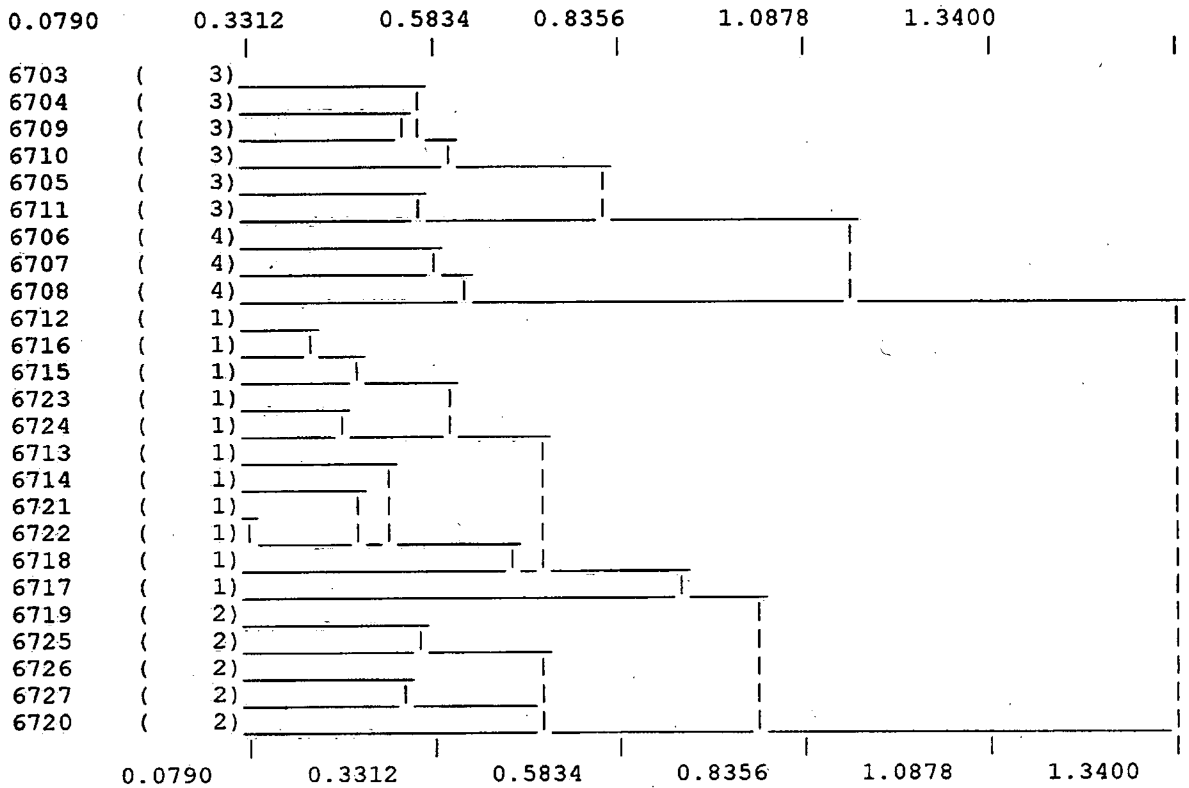


Figure 5. Relative abundance of families in the invertebrate community of Collingwood Harbour.

M. tardigradum, the Platyhelminthes and the tubificid worm, *Limnodrilus cervix*.

Figure 6. Collingwood - Dominant Species (22 spp)



The geographic location of the sites in the four groups shows a similar pattern to that of the toxicity and chemistry data (Fig. 3c). The sites in the east slip make up a single group (Gp 4) as do the west slip sites (Gp 3). A third group represents the area just outside the slip (Gp 2) and two sites on the east side of the harbour (# 6719 and # 6720). The final group is formed from the majority of outer harbour sites. The distribution of the ten taxa contributing most to the structure in the data (Table 11), shows both qualitative and quantitative differences in the four groups of sites. The east slip sites (Gp 4) have very few organisms and very low number. The Gp 1 sites (outer harbour) have the greatest number of taxa and the highest number of organisms. The other two groups are intermediate.

Figure 7. Ordination of Collingwood Harbour - community structure.

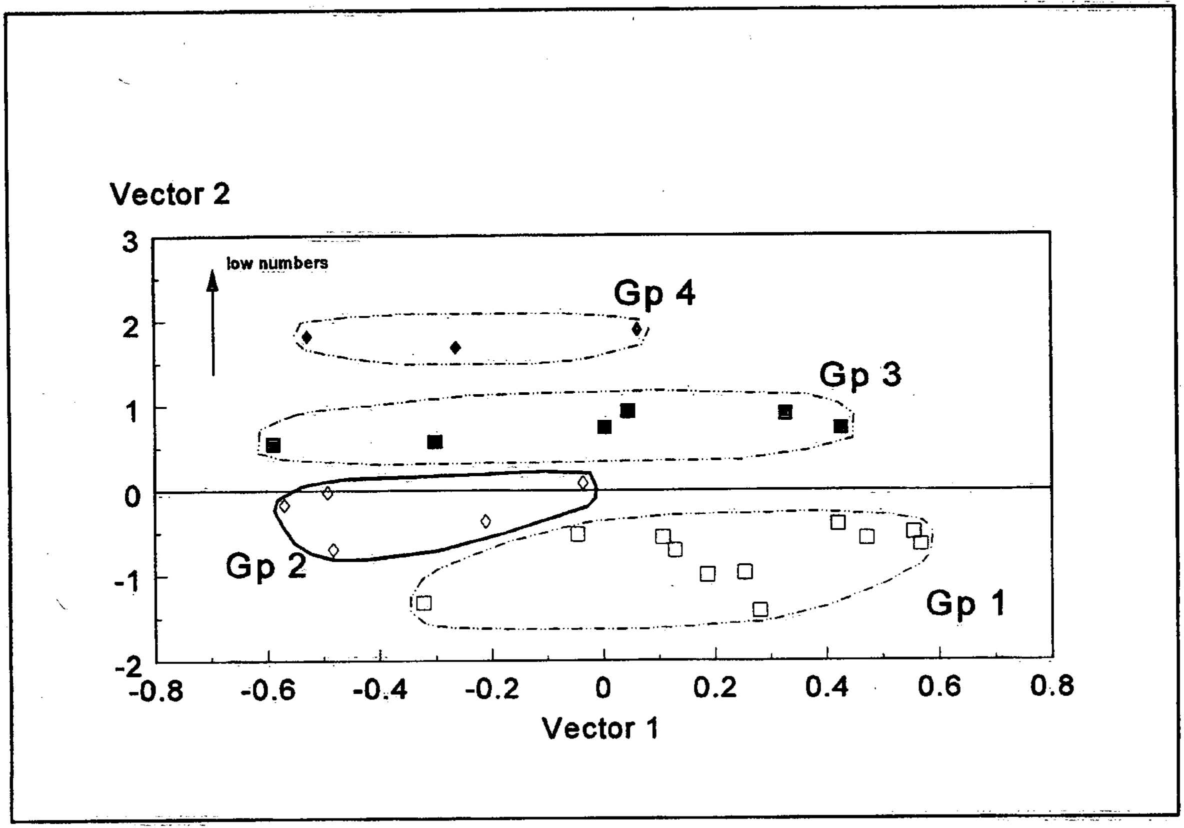


Table 11. Species composition of 4 groups formed by cluster analysis.

Taxa	Gp 1 (11 sites) Outer Harbour	Gp 2 (4 sites) Inner Harbour	Gp 3 (6sites) West slip	Gp 4 (3 sites) East slip
Tubificidae	51.2	38.8	10.4	1.1
V. intermedia	161.2	28.5	0.4	0.1
Tubificidae	19.9	15.7	2.9	0.5
Cladoplema sp	5.6	1.8	1.0	1.4
Platyhelminthes	4.4	0.8	0	0
L. claparedianus	0.4	0.1	0	0
Procladius spp	3.7	9.2	1.4	0.5
T. tubifex	2.4	0.5	0	0
Q. multisetosus	0.5	0.6	0	0
S. josinae	0.5	0.1	0	0

Table 12. Species indicative of five "reference communities" in the Great Lakes.

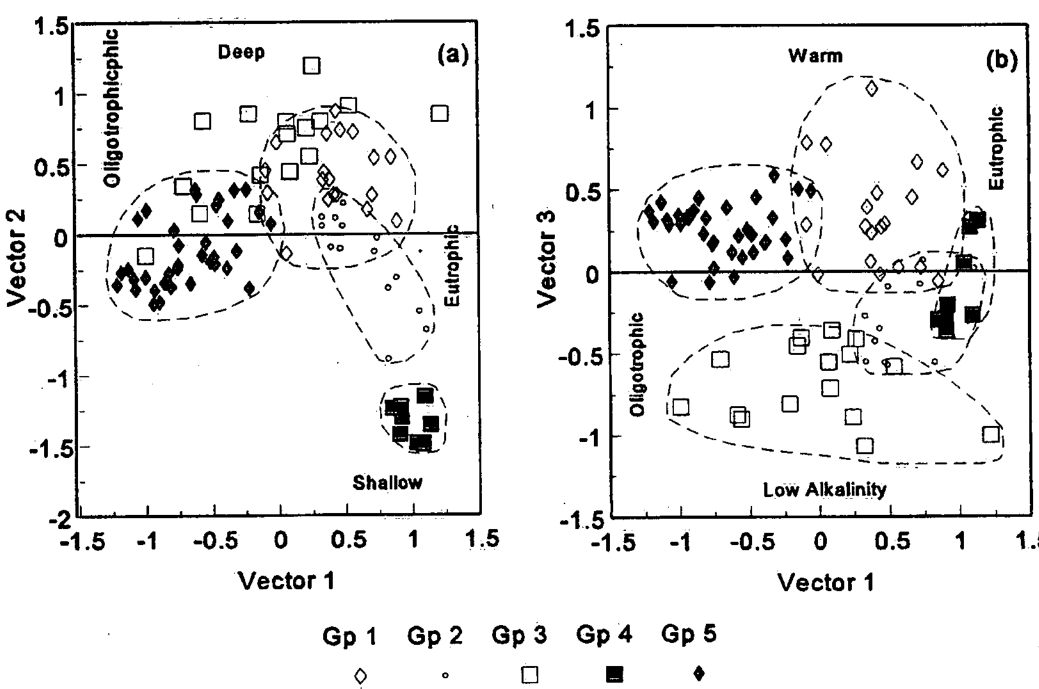
Community Type (Gp)	Abundant Species (>70%)	Common Species (>50%)
1	Tubificidae Tubificidae *Procladius*	*Pisidium* Porifera *Spirosperma ferox* *Dreissenia polymorpha* Platyhelminthes *Limnodrilus hoffmeisteri* *Chironomus* spp *Aulodrilus pigueti*
2	Porifera Tubificidae *Procladius* *Cryptochironmomus* *Chironomus* *Pisidium* Tubificidae *Tanytarsus*	*Valvata* *Aulodrilus* Platyhelminthes *Pisidium* *Polypedium*
3	Tubificidae	Tubificidae *Procladius* *Pisidium* *Diporeia hoyi* *Microspectra* sp
4	Porifera *Procladius* *Chironomus* *Dicrotendipes*	*Microtendipes* *Cryptochironomus* Tubificidae *Physella* *Polypedium* *Pisidium casertanum* *Pisidum nitidum* *Endochironomus* *Pseudochironomus* *Pisidium* spp *Aulodrilus pigueti* *Amnicola limosa*
5	*Diporeia hoyi* *Stylodrilus heringianus* *Pisidium sp* *Vejdovskella intermedia*	Platyhelminthes Heterotrissocladius *Pisidium casertanum* Tubificidae (no hr) Tubificidae (hr)

Based on data from reference sites located throughout the Great lakes, five types of communities have been identified in fine-grained nearshore habitats. The species which are characteristic of each type of community are identified in Table 12. Communities 1 and 2 are more typical of mesotrophic areas with the species in these two groups associated with more productive habitats such as sites found in the lower Great Lakes, particularly Lake Erie and lower Georgian Bay. When these communities are examined in ordination space (Fig. 8) there is also some overlap between communities 1 and 2. This suggests that these assemblages may represent a continuum rather than two discrete community types.

Community 3 is associated with less productive conditions. The substrate at the sites which comprise this type of community has a lower silt content and is characterized by species such as *Diporeia hoyi*, which is usually found in habitats which are classified as oligotrophic. Thus, this community is more typical of the upper Great Lakes. The fourth community type is represented by a small group of sites in very shallow water and is typified by large numbers of chironomid species. The last community (Gp 5) is associated with deeper waters, a sandy substrate and species such as *Diporeia hoyi* and *Stylodrilus heringianus*.

Using nine environmental variables (Table 13) in a discriminant analysis model we predicted which of these community assemblages should exist at each of the 25 sites sampled in Collingwood Harbour. With the exception of site 6711, all the sites were predicted as having a Gp 1 community assemblage (Table 12). This type of community is represented by the Tubificidae and the chironomid, *Procladius* spp., being the most common organisms but several other species are also frequently found (Table 12) and this reference community occurs most typically in Lake Erie, and some parts of Georgian Bay.

Figure 8 . Ordination of Great Lakes Reference sites in five Community Groups.



We compared the species which are characteristic of the two reference communities (Table 12) which were predicted at Collingwood Harbour (Table 13), and found that in most cases, that assemblage of species did occur at the Collingwood Harbour sites (Table 13). This assessment of the degree to which the communities compare with the the expected assemblage of organisms is strictly qualitative. A more rigorous approach is one that involves a comparison of the compositions of species in ordination space. Thus, we repeated the ordination and examined the location of the Collingwood Harbour sites relative to the Gp 1 reference sites in ordination space (Fig. 9). The results show, that of the 24 sites predicted as having a Gp 1 community, most fall within the boundaries of the group of reference sites comprising community 1. The variation found at these Collingwood sites is no greater than that observed among reference sites and, thus, these Collingwood Harbour sites can be considered to be clean sites. Exceptions were sites # 6708 and # 6717 on the second vector (Fig 9a), and sites # 6706, # 6707, # 6708 and # 6709 on the third vector. One site (# 6711) was predicted as having a community represented by Gp 4. This site was well outside the range observed in the reference sites (Fig. 9c) on vector 2.

Table 13. Probability of Collingwood Harbour sites being a member of one of five reference commuities from MDA with nine environmental variables (depth, nitrate, alkalinity, pH, silt, sodium, calcium, aluminium and LOI).

Site	Prob of being member of Predicted Gp	Predicted Community	Site	Prob of being member of Predicted Gp	Predicted Community
6703	0.791	1	6715	0.974	1
6704	0.826	1	6716	0.769	1
6705	0.906	1	6717	0.972	1
6706	0.885	1	6718	0.933	1
6707	0.748	1	6719	0.989	1
6708	0.924	1	6720	0.991	1
6709	0.893	1	6721	0.986	1
6710	0.839	1	6722	0.993	1
6711	0.904	4	6723	0.994	1
6712	0.993	1	6724	0.992	1
6713	0.987	1	6725	0.990	1
6714	0.987	1	6726	0.965	1
			6727	0.992	1

Table 14. Presence of predicted species at Collingwood Harbour sites at two levels of probability

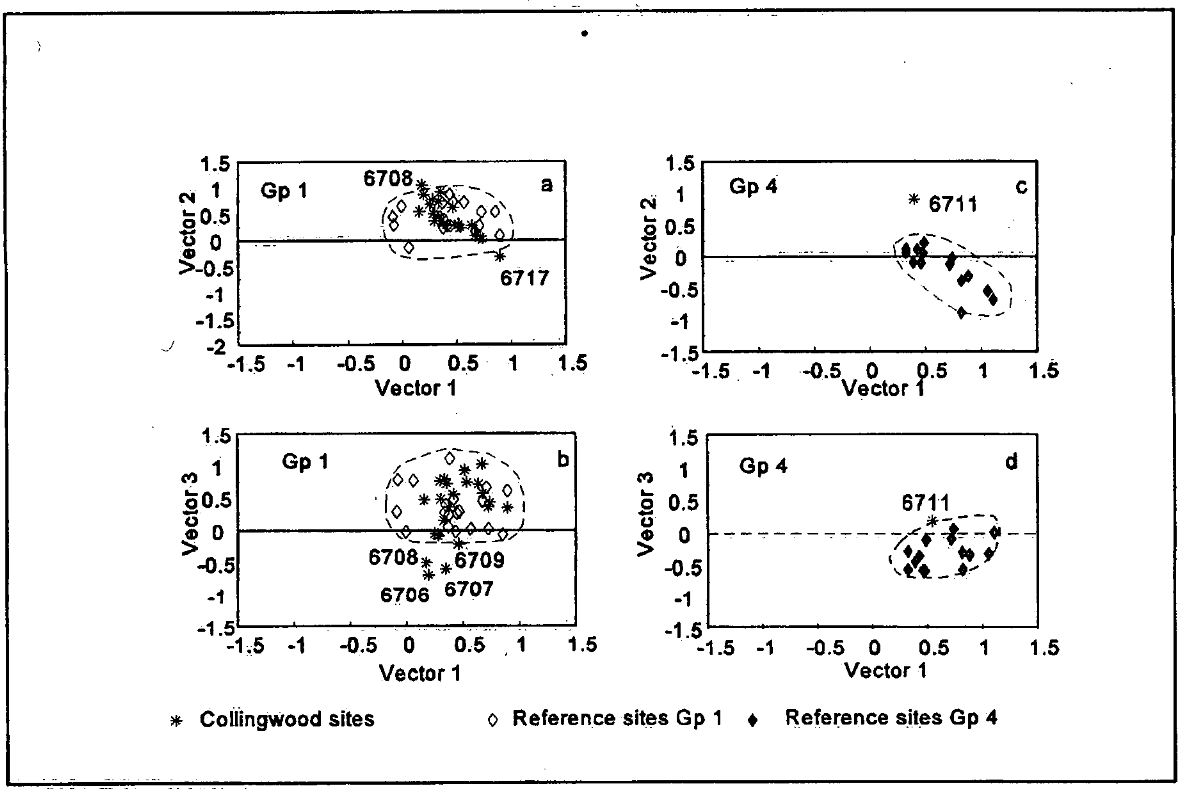
Site	Species with Prob > 0.7 (max=3)	Species with Prob >0.5 (max=7)	Site	Species with Prob > 0.7	Species with Prob >0.5
6703	2	5	6715	3	5
6704	3	5	6716	3	5
6705	3	3	6717	3	5
6706	2	4	6718	3	5
6707	3	4	6719	3	5
6708	3	3	6720	3	6
6709	3	3	6721	3	5
6710	3	3	6722	3	5
6711	3 of 4	1 of 12	6723	3	6
6712	3	5	6724	3	4
6713	3	5	6725	3	4
6714	3	5	6726	3	4
			6727	3	4

Examination of the data on the spatial pattern of the site groups enhances the interpretation of sediment contamination in the harbour. As with the chemical and whole-sediment toxicity test data, the ordination of the sites differentiates between the two boat slips (east and west) and also separate the inner and outer harbours into localized areas where sediment contamination is a potential problem (Fig 3c).

Discussion

There is good agreement between the results from chemistry, whole-sediment exposure tests and in situ community structure (Fig. 3a, b, c). All three measurements indicate that three to four zones of contamination exist in the harbour, i.e., the east and west boat slips, the inner harbour area just outside the slips and the outer harbour. In each case, the east slip is shown to have the highest levels of contamination, the poorest performance in the toxicity tests and the most impaired benthic community. The data on the structure of the bentic community structure distinguish the west slip from the area outside the slips; the chemical and toxicity data did not provide this distinction but rather included the west boat slip sites with the inner harbour sites. However, the toxicity data indicate an expanded area of sites along the east wall of the harbour where at least 2 of 8 endpoints indicate sediment toxicity. Finally, all three methods of assessment suggest that the outer harbour sites are either the least contaminated (chemistry) or are not causing an mpact on the benthic invertebrate community (toxicity and community structure).

Figure 9. Collingwood Harbour sites compared with reference sites in ordination space.



The Province of Ontario has developed a set of chemical sediment guidelines for defining sediment impairment (Persaud *et al.* 1992) at severe and low effect levels to biota based on data derived from screening level criteria (SLC) and laboratory toxicity tests with the chironomid, *C. tentans*, the mayfly, *Hexagenia* spp. and the fathead minnow, *Pimephales promelas*. A comparison of the concentrations of chemicals measured in sediments at the 25 sites in Collingwood Harbour to the Province of Ontario's criteria (Table 15) indicates that none of the sites meet all the criteria and a number of sites in the east slip (# 6706, #6707, #6708) and one in the west slip (# 6709) exceed the low effects criteria for all variables and the severe effects criteria for some of the variable(*e.g.*, copper, zinc, arsenic, *etc.*). A few sites in the outer harbour meet the criteria for a few variables, most notably, iron (16 sites), chromium (15 sites) and cadmium (15 sites).

Using the methods developed in this study, there was good concordance between the chemical and biological data for the most severely contaminated sites *i.e.*, both the *in situ* data on community structure and the data from the laboratory toxicity tests indicated toxicity at the sites with the highest concentrations of contaminants. The most contaminated sites (# 6706, # 6707, # 6708, # 6709 - Table 15) had the most depauperate benthic invertebrate communities with only a few but abundant organisms present (e.g., Oligochaeta). Other sites in the outer harbour with lower concentrations of contaminants required the biological data to define whether an impact was occurring. Two sites exceeded the severe effects level based on chemical criteria yet biological effects were not demonstrated at these sites. Both site # 6710 (which had a high copper concentration; 210 ppm) and site # 6705 (which had a high lead concentration;329 ppm) had reduced numbers of organisms but these numbers were within the range of variation found at reference sites.

Table 15. *Sediment Chemistry for contaminants for which guidelines are available (Values in ppm dry wt).*

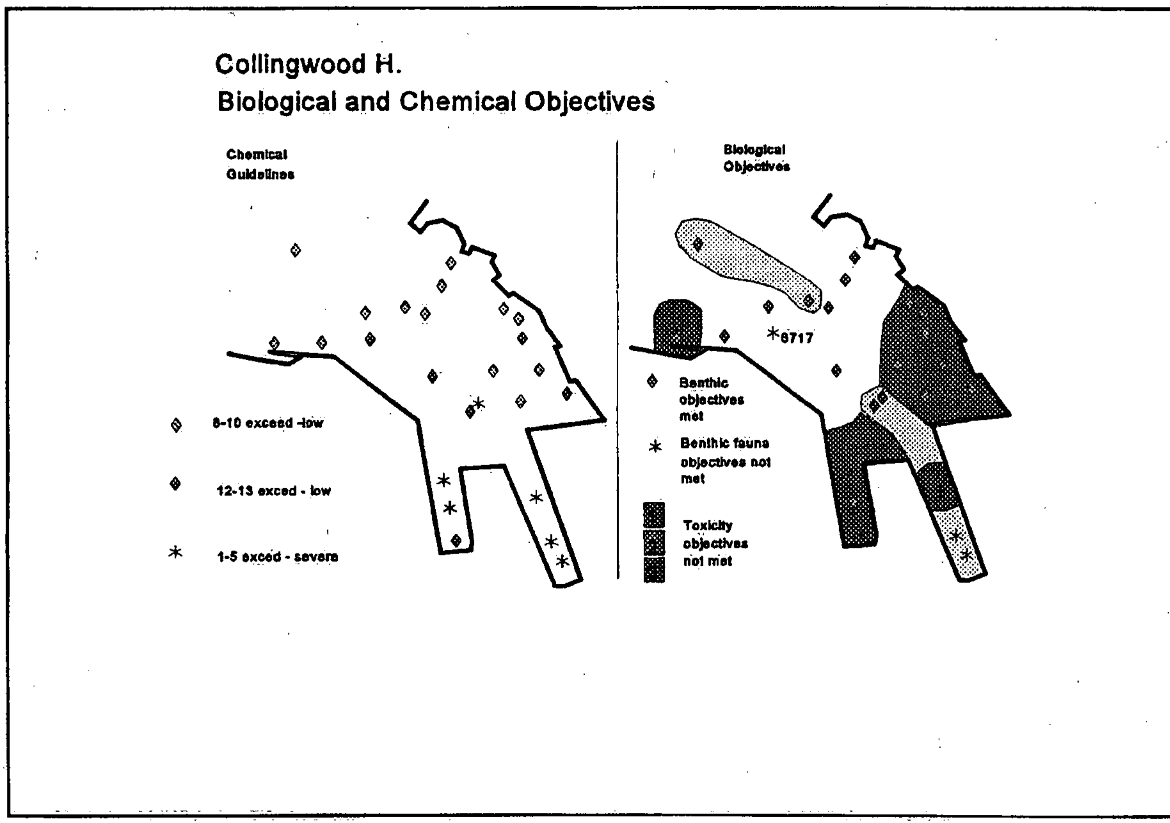
	Total N	Total P	TOC (%)	Total Fe (%)	Mn	Cr	Ni	Cu	Zn	As	Cd	Pb
Low	550	600	1	2	460	26	16	16	120	6	0.6	31
Severe	4800	2000	10	4	1100	110	75	110	820	33	10	250
SITE												
6703	975	1095	1.77	1.64	522	25	21	37	129	2.5	0.6	76
6704	2205	1290	2.11	1.52	480	24	21	35	123	2.5	0.5	63
6705	2042	1335	3.48	2.17	570	29	24	101	411	2.5	1.2	**329**
6706	1524	1230	2.34	**11.2**	594	65	33	**2835**	**10780**	**105**	3.9	**709**
6707	1244	945	1.85	**14.43**	642	85	36	**4170**	**13943**	**137**	4.9	**974**
6708	1364	1080	2.08	**8.87**	646	58	31	**2121**	**7527**	**70**	4.2	**724**
6709	1825	1200	2.44	**4.98**	**1114**	47	27	**893**	**3154**	25	1.8	**430**
6710	2441	1515	2.66	2.98	647	40	29	**201**	669	2.5	1.5	200
6711	2211	1560	2.65	2.19	583	32	25	109	380	2.5	0.4	149
6712	1661	607	2.08	1.52	355	18	12	49	216	7	0.4	105
6713	2499	866	2.59	1.73	491	23	19	36	145	12	<0.2	71
6714	2439	821	2.45	1.72	495	22	20	35	141	9	0.3	71
6715	2371	801	2.52	1.55	460	20	19	32	125	<5	0.6	57
6716	2299	1032	3.06	1.61	405	25	17	65	247	6	0.4	88
6717	2431	850	2.93	1.53	444	20	17	35	151	10	0.3	55
6718	2534	988	2.63	1.56	453	21	20	34	138	8	0.5	59
6719	2128	902	2.28	1.78	487	23	22	50	161	13	0.3	94
6720	2082	1124	2.83	1.98	480	25	23	54	204	11	0.3	115
6721	2239	779	2.06	1.66	472	22	20	36	145	11	0.8	68
6722	2315	949	2.22	1.77	489	23	21	39	155	18	0.4	92
6723	2194	1029	2.44	1.89	525	25	23	41	165	12	<0.2	91
6724	2230	868	2.34	1.85	507	25	23	41	169	<5	0.4	92
6725	2293	998	2.36	2.23	508	26	22	96	467	12	0.5	111
6726	2368	630	2.72	1.67	464	20	19	54	198	10	<0.2	195
6727	2305	852	2.26	2.04	526	26	26	45	182	15	0.9	105

Bold values exceed the severe effects level established by Province of Ontario.

Remediation of sediment in Collingwood Harbour may not be warranted at all harbour sites based on the data from this analysis. A comparison of sites exceeding the chemical criteria with those exceeding the biological criteria (Fig. 9) indicates that where the

However, at many of the sites which exceed the low effects criteria, the biological data do not indicate any impairment.

Figure 9. Comparison of chemical and biological objectives in Collingwood Harbour.



When a map of Collingwood Harbour is examined (Fig. 9), the data on community structure show good correspondence with the chemical data. For example, the structure of the benthic invertebrate community was only outside the range of species established for reference sites at sites where the levels of metals exceed the severe effects levels established by the province. Only one site in the outer harbour was identified as being outside the range of our reference data base (*i.e.*, # 6717) but the levels of contaminants at this site were all below the severe effects limits and this outlier can be attributed to the high numbers of freshwater sponges (Porifera) at this site rather than toxicity. The results from the whole-sediment laboratory tests also do not indicate a problem with sediment toxicity in the outer harbour with the exception of the results for reproduction of the oligochaete worm, *T. tubifex*. Total production of young was reduced at three sites in the outer harbour and in the south-east corner of the harbour, as well as in the east boat slip. The reduction in reproduction in the east slip may be attributed to the high levels of metals at these sites. Oligochaetes are known to be particularly sensitive to metal contamination (Hynes 1960, Brinkhurst and Jamieson 1971, Aston 1973, Chapman *et al.* 1980, Wachs 1980). In addition, the number of tubificids in the field-collected samples for benthic invertebrates were very low at these sites. The reason(s) for the lowered reproductive output of tubificids at sites in the outer harbour and in the south-east corner of the harbour is unknown at this time and an impact due to sediment contamination and/or substrate type cannot be discounted. However, natural populations of oligochaete worms including tubificids are present at these sites so the effect (if any) of sediment toxicity appears to be minimal in the real world.

Based on the above results, removal of contaminated sediment appears to be warranted in the east and west boat slips. The remainder of Collingwood Harbour appears to support a sustainable and reproducing benthic invertebrate community similar to that predicted from reference sites. This is despite the fact that several of the provincial chemical sediment criteria are exceeded for metals at the severe effects level. In addition, there is little additional weight of evidence from the laboratory toxicity tests for an extensive remedial action programme in the outer harbour. This case study emphasizes the value of a combined laboratory and field approach using both chemical and biological data in interpreting the effects.

Conclusions

Using a combined chemical and biological approach as detailed in this study, we conclude that the sites in the east and west boat slips of Collingwood Harbour are clearly in need of remediation. This is based on both evidence from whole-sediment toxicity tests with four species of benthic invertebrates and impairment of the *in situ* benthic community. An area of impairment that extends into the southerly portion of the inner harbour could also be included in this remedial action.

There is some agreement between the Province of Ontario's severe effects criteria for several metals and the biological effects demonstrated in both the field and laboratory studies. However, it should be noted that in some of these cases of agreement, the severe effect criteria were exceeded by several orders of magnitude especially in the case of copper, zinc and arsenic.

It is also noteworthy that at several sites in the harbour where the severe effect levels were exceeded, there was no evidence of toxicity demonstrated for the chronic endpoints in three of the four toxicity tests. Reduced reproduction in the tubificid worm, *T. tubifex*, was evident at a number of sites in the outer harbour but this reduction could not be attributed to any specific chemical compounds. There was no evidence of acute toxicity in any of the tests.

Any remedial action in Collingwood Harbour should include a re-sampling component at the same sites to establish if a) re-colonization by benthic invertebrates has occurred b) if there has been recovery of the benthic invertebrate communities to levels similar to reference sites. Once it has been demonstarted that there are no further biological effects due to new contamination of the sediment, and, if based on comparison to reference sites, it can be shown that the benthic community is returning to levels indicative of reference conditions, we would recommend that Collingwood harbour be delisted as an area of concern with regard to sediment contamination.

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