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Great Lakes Dredging in an Ecosystem Perspective - Lake Erie: A Report Submitted to the Dredging Subcommittee of the Water Quality Board

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GREAT LAKES DREDGING IN AN ECOSYSTEM PERSPECTIVE - LAKE ERIE

A REPORT SUBMITTED TO THE
DREDGING SUBCOMMITTEE
OF THE (Great Lakes) WATER QUALITY BOARD
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BY

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THE VIEWS EXPRESSED IN THIS REPORT ARE STRICTLY THOSE OF THE AUTHORS AND DO NOT NECESSARILY REFLECT THE OPINION OF THE MEMBERS OF THE DREDGING SUBCOMMITTEE

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Executive Summary

I. Introduction

Although Lake Erie is the second smallest of the Great Lakes in surface area, and smallest in volume, it serves both as a vital link for shipping and as a productive fishing ground. Maintenance of harbors and shipping lanes required the dredging of over 4 million cubic meters place material during each of the years 1975 through 1979, representing well over half of the dredging for all of the Great Lakes. The magnitude of dredging, and the value of the lake for both commercial and sport fishing, has necessitated the careful monitoring of dredging activities. The purpose of this report is to assess the amount of dredging and its biological impact on the Lake Erie ecosystem.

II. Contaminant Status of Water, Sediment and Biota in Lake Erie.

1. Water

The Detroit River is the major tributary of Lake Erie representing 92.1% of the annual fluvial input of water and 84% of the annual total input. The Niagara River serves as the major fluvial outflow. Assessment of the contaminant status of Lake Erie water is made difficult due to the numerous studies which have been conducted on the lake for many purposes employing different sampling techniques, with varying analytical methods, and performed at different times of the year. Zinc and nickel are the most abundant of the contaminants studied in this report and cadmium and PCB's the least abundant. Table A summarizes and Appendix 1 details the available data.

Table A. Summary of contaminant concentrations in the sediment, water and fish of Lake Erie. Details may be found in appendices 1,2 and 4.

	Water (ppb)	Sediment (ppm)	Fish (ppm)
Arsenic	0.35 (0.1-0.6)	2.28 (0.2-60.0)	0.05 (0.01-0.12)
Cadmium	0.20 (N.D.-0.20)	2.65 (0.1-13.7)	0.13 (0.19-0.78)
Chromium	1.60 (N.D.-14.0)	54.22 (3.4-362.0)	0.22 (0.01-0.39)
Copper	1.75 (1.0.-17.0)	39.53 (3.0-207.0)	0.53 (0.18-1.56)
Lead	1.50 (0.15-12.0)	73.02 (4.6-299.0)	0.21 (0.09-0.52)
Mercury	0.61 (N.D.-0.61)	0.59 (0.01-7.5)	0.56 (0.01-3.6)
Nickel	3.50 (2.0.-4.0)	48.42 (9.0-121.0)	1.16 (0.12-7.53)
Zinc	5.50 (2.0-9.0)	129.22 (12.0-536.0)	4.84 (4.27-6.05)
PCB's	0.03 -	0.14 (0.004-0.80	1.31 (N.D.-14.0)

2. Sediment

Assessment of the contaminant status of the sediments suffers in the same manner as the water due to the varying methods used in the various studies reviewed for this report. As the sediment serves as a sink for contaminants it is not surprising that the sediments contain the contaminants in concentrations 2 orders of magnitude greater than the water. Zinc and lead are the most abundant contaminants in the sediments and arsenic and PCBs the least abundant. The available data is summarized in Table A and detailed in Appendix 2.

3. Pollutants Bound in Organisms

Assessment of the mass of contaminants in the biota, representing the biotic reservoir, was not possible due to the lack of overall estimates of lake-wide biomass for each of the trophic levels. Analysis of the mass of contaminants leaving the lake via fish harvest suggests that the biota are not a major store of the contaminants.

III. Contaminant Budget for Lake Erie

Mass balance studies attempt to identify the principle points of material storage and movement within a system. In this instance the mass balance approach was used to put dredging into perspective with the other major sources to, and exports from, contaminants of Lake Erie.

1. Inputs

Point source loadings are not a major contribution to the contaminants directly entering the lake, representing no more than 4.5% of any contaminants total input (Table B). Shore erosion data are available only for the Canadian portion of the lake shore. The shore serves as the source of 47% of the total arsenic load annually, but is an insignificant source for all other contaminants. However, the absence of data on contaminant loading from the United States portion of the shore may alter the perception of the shore as a contaminant source and should be corrected. Atmospheric loading is an important source of cadmium, lead and arsenic, and of least significance for nickel. Though atmospheric deposition is the major source of PCB to Lakes Michigan and Superior, Lake Erie receives a far smaller proportion due to its much smaller surface area and dependence on tributary inflow from the Detroit River. The Detroit Rver, and to a much smaller extent the other tributaries, represents the major source of most contaminants into Lake Erie. As examples, 99.8% of the mercury, 89% of the zinc, and 96% of the PCB's enter the lake from its tributaries each year. The sole exception is arsenic, with tributaries contributing only 31% of the total annual load.

Table B: Inventory of sources of contaminants in Lake Erie (fluxrate kg/yr).

Contaminant	Point Source	Tributaries	Shore Erosion	Dry Wet Fall	Dredge Material	Discharged into Lake Erie
Arsenic	7.87×10^3	5.62×10^4	8.50×10^4	2.00×10^4	1.10×10^4	7.02×10^4
Cadmium	3.56×10^2	1.05×10^5	1.10×10^4	8.00×10^4	5.82×10^3	1.76×10^5
Chromium	7.54×10^3	2.02×19^6	3.29×10^5	-	7.49×10^4	1.76×10^5
Copper	2.33×10^4	1.66×10^6	1.33×10^5	1.50×10^5	6.45×10^4	5.27×10^5
Lead	1.07×10^4	1.06×10^6	1.50×10^5	7.50×10^5	6.30×10^4	1.76×10^5
Mercury	6.20×10^1	5.23×10^5	1.65×10^2	4.00×10^3	6.80×10^2	1.76×10^3
Nickel	1.80×10^5	3.88×10^6	1.60×10^5	8.00×10^4	7.94×10^4	5.27×10^5
Zinc	4.73×10^4	1.43×10^7	3.98×10^5	100×10^6	2.43×10^5	1.05×10^6
PCB's	-	1.07×10^5	-	3.10×10^3	8x10	8.78×10^2

-data not available

When placed in the perspective of the total loading from all sources dredging can be seen to be a relatively minor source. Dredging is the source for no more than 6.5% of the total annual load for any of the contaminants studied in this report. However, this is not to discount the importance of a source responsible for 5.43×10^5 kg of contaminants in Lake Erie each year. Although representing a small portion of the total volume of contaminant loading, dredging may best be regarded as a point source which may result in significant local impacts.

2. Exports

Of the total contaminants which leave Lake Erie each year, most leaves via the Niagara River, the major hydraulic outflow. The Niagara River accounts for most of the export of cadmium, nickel and copper but only a relatively small percentage of PCB's. The other major export is by on-land or confined dredge material disposal, accounting for significant exports of PCB's, lead, chromium, and zinc. Export of contaminants through fish harvest is not significant, accounting for less than 1% of the contaminants lost. Mercury is also lost from the lake through volatilization, responsible for almost 15% of the annual export of the metal.

3. Contaminant Retention Time

Most of the contaminants entering Lake Erie become bound in the sediments, such that over 99% of the mercury and 95% of the PCB's entering annually remain in the lake. Arsenic, and especially cadmium, are not retained to the same extent. The values calculated for the retention of the contaminants in the waters of Lake Erie are all less than one year, ranging from arsenic, with a retention time of 333 days, to 43 days for PCB's. Although this is a seemingly low value for PCB's, the relative retention, that is, the ratio of retention time of the contaminant to the retention time of the lake water, is nearly identical with that for Lake Superior. The relative retention time for all contaminants is less than 1 indicating that they are readily incorporated into sediments and become unavailable.

4. Weaknesses in the Data Base

Apart from the problems related to sampling and analytical procedures, data for this analysis were absent or contained few useable values for non-point source loading, loading due to shoreline erosion in the United States portion of Lake Erie shores, and municipal point source loadings from the United States portion of the lake.

IV. The Fate of Contaminants in Lake Erie

The ultimate fate of contaminants in Lake Erie is the result of integrated processes involving physical, chemical, and biological mechanisms. Within each of these broad categories of mechanisms, there are a number of factors which are responsible for the form the contaminant takes in the lake.

Such factors include pH, redox potential, and methylation. The form of the contaminant exists is important as it strongly influences its toxicity and mobility.

V. The impact of dredging on the Lake Erie ecosystem.

Studies have documented a change in benthic invertebrate community structure following dredging activities in Lake Erie. These studies have concluded that the alteration in community structure is a negative impact whose effects have been observed to last at least 5 years. However, these studies lack certain data which weaken their conclusions. First, while all document a change in numbers of individuals, none document the change in biomass of the benthos, a critical measure of productivity. Second, none partition the effects due to burial and contaminant release making it impossible to determine which process is responsible for the observed alterations. Third, these studies make conclusions which are not based on ecological concepts regarding community structure. A change in the distribution of species abundances of the benthos over a relatively small area is not necessarily a damning flaw. There are no guidelines for "quality of benthos" indicating desirable species, community structure, or productivity. There are no studies on the impact to the rest of the ecosystem resulting from a change in the species composition of the benthos.

The sediments of Lake Erie may hold potentially toxic levels of many contaminants, with many of the more often dredged areas, particularly harbors, being the most contaminated. Before dredging disposal decisions can be made the protection of the ecosystem should be ensured with carefully designed and monitored pilot studies which will determine if there are deleterious effects to the ecosystem including changes in productivity and the release of harmful contaminants to the higher trophic levels.

Introduction

1. Purpose of the study

The goal of this report is to assess the amount of dredging and its biological impact on the Lake Erie ecosystem. To this end four objectives have been established:

1) Determine the contaminant status of the sediments, water and biota in Lake Erie;
2) Describe the contaminant budget for a defined set of pollutants (arsenic, cadmium, chromium, copper, lead, mercury, nickel, zinc, PCB's, oil and grease) previously identified as particularly harmful to humans and the aquatic ecosystem. Ascertain the importance of dredging in relation to other sources of these pollutants;
3) Detail the fate of contaminants in Lake Erie from both a physico-chemical and biological perspective; and
4) Describe the toxicity of contaminants and the adequacy of the tests used for toxicity determinations as well as elutriate tests.

CONCLUSIONS

1) The sediments of Lake Erie represent a sink for the majority of contaminants entering the lake each year. Regardless of consideration as a redistributor of in-place contaminants or an original source, on the basis of available data dredging activities play a relatively minor but controlable role as a contaminant source. However, on-land and confined disposal represents a significant means of removing the contaminants from the lake.

2) While advances have been made in our knowledge of physical and chemical parameters associated with contaminant availability the relationship between these factors and dredging activities remains poorly understood.

3) It is widely recognized that bioaccumulation of contaminants occurs at all trophic levels. However, the role of dredging in the release of contaminants is uncertain and conclusions regarding the impact of dredging activities are tentative, but suggest that dredging plays a minor part in bioaccumulation of contaminants. While the contaminants have all been shown to have potential deleterious effects on the biota of Lake Erie, the weakness of the data base allows for uncertainty on the role of dredging activities in the release of potentially toxic concentrations of contaminants.

4) Dredging activities have a direct impact on the immediate areas involved, i.e., the dredge site and dredge material disposal site. While there is some evidence that the dredge disposal site may exhibit alterations in community structure even after 5 years of disuse, there is no evidence regarding the change in quality of the benthos or whole lake ecosystem due to dredging activities.

5) The fact that there is limited evidence relating dredging to any sort of ecosystem impairment does not mean that dredging has no impact. Rather, there has simply been a lack of scientific studies which have been designed to show the extent of the impact.

RECOMMENDATIONS

The lack of data regarding contaminant release and accumulation by the biota directly due to dredging activities should be addressed. To that end it is recommended that an experimental dredging project be established which will:

1) monitor important abiotic factors, such as pH, redox potential, alkalinity, and dissolved oxygen, at both the dredge and dredge disposal sites.
2) monitor the change in concentration of selected contaminants and their possible alternative forms.
3) monitor the change in the benthic flora and fauna, including
   a) Alteration in bacterial flora and possible impact of methylation of mercury, arsenic, lead and cadmium, and biodegradation of PCB's.
   b) Alteration in the productivity of the macroinvertebrates.
   c) Alteration in the contaminant status of the macroinvertebrates.
4) Determine if bioaccumulation of contaminants released directly from dredging activities is significant to phytoplankton, zooplankton, and fish.

It is recommended that a study be commissioned which will determine the proper protocol for a sediment leachate bioassay. Such a study should investigate:

1) The specific conditions under which tests can be run which will minimize variability, e.g., length of experiment, type of water used, amount of sediment used, reductive or oxidative states, etc.
2) The use of test organisms which are closely related taxonomically to the biota of the sites in question. This may include the use of populations derived from organisms collected on site.
3) The use of microcosms of increasing complexity to determine possible ecosystem effects beyond those observed for single species.

It is also recommended that the collection and processing of biological materials for contaminant analysis should be standardized in a manner similar to water and sediment analyses.

II LAKE ERIE - AN INTRODUCTION

Lake Erie is the second smallest of the Great Lakes in surface area at 25,657 km^2 and smallest in volume at only 483 km^3. These values represent 10% of the total surface area of the Great Lakes and only 2% of the total volume. It is the shallowest of the lakes with an average depth of 19 m. The drainage basin for Lake Erie is relatively small, 58,800 km^2, but includes areas of intense agricultural and commercial activities and is heavily populated. Bordered by four states and the province of Ontario, several large cities rest on the banks of Lake Erie including Toledo, Cleveland, and Buffalo along with smaller population centers such as Erie, Ashtabula, and Leamington. Lake Erie serves as a link between these cities and as a thoroughfare to the upper lakes. The heavy use of Lake Erie for shipping, the shallowness of the lake, and the heavy siltation load from its tributaries has necessitated dredging activities surpassing that in all the other Great Lakes.

In 1979, 66.3 million tonnes of cargo moved through the Welland Canal at the eastern end of the lake, 40% of which was grain with an approximate value of $2 billion (Misener, 1981). Iron ore and other bulk material comprised most of the rest of the cargo. Between 1979 and 1985 shipping is expected to increase by almost 35% (Misener, 1981). These figures do not take into account the value of shipping entering Lake Erie through Lake St. Clair or intra-lake shipping. The cargo industry of Lake Erie is responsible for the direct employment of more than 15,000 people. Shipping expenses for the four major commodities (iron ore, coal, limestone, and grain) accounted for almost $4,340 million in 1970 (International Working Group on the Abatement and Control of Pollution from Dredging Activities, 1975). Dredging in the Great Lakes is a major industry employing 2,500 people with annual payroll of approximately $26 million (International Working Group on the Abatement and Control of Pollution from Dredging Activities, 1975).

Table 1. Annual commercial fish harvest in Lake Erie. (000's kg)

Year	U.S. Harvest	Canadian Harvest	Total	Value ($000's)
1976	4,107	11,570	15,670	5,990
1977	4,392	16,134	20,526	7,737
1978	*	18,072	*	9,884 +
1979	*	18,473	*	17,592 +
1980	*	19,490	*	14,024 +

sources: Canadian harvest - Ontario Ministry of Natural Resources, 1981

U.S. harvest - Baldwin et al., 1979

\+ Canadian harvest value

*data not available

The other major commercial utilization of Lake Erie is as a fishing ground. While there has been a dramatic decrease in the abundance of certain fish species common to oligotrophic lakes in favor of introduced and rough fish, the total harvest and value of harvest increased in the five year period, 1976-1980 (Ontario Ministry of Natural Resources, 1981) (Table 1). The Canadian harvest increased 68% during this period with perch and the introduced smelt comprising 87% of the commercial catch (Ontario Ministry of Natural Resources, 1981). During this same period the value of the harvest increased 134%. In all, 21 fish species were commercially harvested in 1980 with a value of $14 million (Ontario Ministry of Natural Resources, 1981). The American portion of the harvest decreased steadily during the last 25 years though it remained at a fairly constant level for the period 1970-1977. This portion of the total harvest is only 25% of the Canadian.

These figures represent an even more substantial increase considering that the total Canadian catch for all the Great Lakes was 21.9 million kg in 1973 (International Working Group on the Abatement and Control of Pollution from Dredging Activities, 1975). The Canadian fishing industry on Lake Erie employed 652 people in 1980 making use of $25 million worth of equipment (Ontario Ministry of Natural Resources, 1981).

The recreational value of the Great lakes perhaps equals or exceeds that of the commercial. It has been estimated that the tourist industry associated with the Great Lakes has a value of approximately $1 billion (International Working Group on the Abatement and Control of Pollution from Dredging Activities, 1975). In addition, in 1980 142,000 sports fishermen spent 1.2 million angler-days on Lake Erie catching over 5.2 million fish (Ontario Ministry of Natural Resources, 1982)..

Table 2.

DREDGING IN LAKE ERIE AND THE GREAT LAKES 1975-1979 (Cubic Meters Place Material)

Year	U.S. Lake Erie	Canada Lake Erie	Total Lake Erie	Total Great Lakes	Lake Erie %
1975	3,496,049	48,855	3,544,904	5,097,457	69.5
1976	3,043,328	59,000	3,102,328	4,524,463	68.6
1977	2,544,056	259,800	2,803,856	4,721,746	59.4
1978	1,902,733	377,732	2,280,465	4,664,147	48.9
1979	2,531,396	18,100	2,549,496	4,029,038	63.3

source: Guidelines and Register for Evaluation of Great Lakes Dredging Projects, 1982

Dredging plays a major role in maintaining the lake for the shipping industry. The dredging activity in Lake Erie represented over half the total volume dredged in the Great Lakes (Table 2). Historically, a vast majority (90%) of dredging in Lake Erie has been for navigational purposes (International Working Group on the Abatement and Control of Pollution from Dredging Activities, 1975). During the period 1975-1979 dredging activities decreased 63% in Canadian waters and 28% in U.S. waters for a total decline of almost 1 million cubic meters place material (Guidelines and Register for Evaluation of Great Lakes Dredging Projects, 1982). Considering the large volume of material dredged annually and the economic importance of the lake, the impact of dredging activities on the lake needs to be investigated thoroughly.

One of the critical factors in determining the extent of dredging in a given area is the cost of disposal options. Removing dredge materials to sites far removed from the dreging location can add $2 to $5 per m^3 to the cost of the dredging project (International Working Group on the Abatement and Control of Pollution from Dredging Activities, 1975). In addition, the cost of preparation or construction of disposal sites should also be considered. The decision of where to place dredged material must, therefore, balance economic and environmental concerns.

Dredging in the Great Lakes has come under close scrutiny in the last decade as awareness of the effects of this activity has increased. Dredging activities have a number of potential negative impacts including disruption of spawning sites, creation of turbidity, disturbance and destruction of aquatic organisms and habitats, resuspension of contaminated materials into the water column, dissolved oxygen depletion, release of nutrients and other material entrapped in the sediments, and the creation of floating scum and debris. Not all of these effects are necessarily harmful to the environment. Floating scum resulting from dredging activites offend our aesthetic senses more than they harm the environment, but the significance of some of the other effects remains largely unknown.

Because of the potential damage to the aquatic environment, guidelines have been established (Guidelines and Register for Evaluation of Great Lakes Dredging Projects, 1982) to regulate both dredging and disposal activities. These guidelines take into account the type of material dredged, contamination levels in the material, and the impact of the dredged material on the proposed dumping site. Depending upon these factors, dumping is suggested in the open lake, upland, in confined areas nearshore, or as beach nourishment.

III CONTAMINANT STATUS OF WATER, SEDIMENTS AND BIOTA IN LAKE ERIE

1) Introduction

Before understanding the impact of dredging on the Lake Erie ecosystem it is first important to determine the total contaminant load in the lake for those pollutants in question. These pollutants, in their elemental form and without regard to whether they are bound or available to the ecosystem, are found either in the water, sediment or biota. The total resident concentration of any given contaminant is then:

$$R = W + B + O$$

where R is the total resident mass of contaminant, W is the mass of the contaminant in the water, B is the mass in the sediment, and O is the mass of the material already taken up in the organisms in the lake.

2) Water

For the purpose of this report the volume of Lake Erie is taken to be 483 km^3, the equivalent of 4.83×10^{14} l (Great Lakes Basin Commission, 1976). The major tributary is the Detroit River with an average flow into the lake of 6146 m^3/sec or 1.938×10^{14} l/yr, representing 92.1% of the annual fluvial input (Table 3). The Maumee River is the second largest tributary but represents only 2.1% of the total annual input, a value slightly less than calculated by Sheng and Lick (1979). The total fluvial input is 2.104×10^{14} l/year. Outflow from the lake is via the Niagara River and Welland Canal with a flow of 5570 m^3/sec or 1.756×10^{14} l/yr. The other major factors in determining the water balance are input from precipitation and loss via evaporation. Precipitation is taken as 78.08 cm/yr (Derecki, 1976) over the surface of the lake (25,657 km^2) or 2.0×10^{13} l/yr. This value is 8.68% of the total inflow into Lake Erie (Table 3). Evaporation is 67.6 cm/yr (Derecki, 1981) or 1.73×10^{13} l/yr. Assuming that the water level in the lake is relatively stable, retention time for lake water is (Bowen, 1975):

$$R = A/(dA/dt)$$

where A is the total water in the lake and dA/dt is the total amount of water entering the lake annually, that is, the sum of the inflow and precipitation, or 2.304×10^{14} l/yr. The volume of the lake (4.83×10^{14} l) divided by this value yields a retention time of 2.10 years. Retention time may also be calculated from the outflow data (Bowen, 1975). In this instance dA/dt is the sum of the outflow and evaporation or 1.93×10^{14} l/yr. The result of this calculation yields a retention time of 2.5 years. Both of these values for retention time, 2.10 and 2.5 years, are in close concordance with other studies (e.g. Nriagu et al., 1979; Burns, 1976).

Determining the overall quantities of the various elements in the water is then a process of multiplying the concentration of heavy metals in the water by the total volume of water. However, as Appendix 1 demonstrates, there exist a wide range of values for each of the pollutants. These values were recorded for many purposes, for different portions of the lake, at different times of the year, using different sampling techniques, during different times of the last decade or so, and were no doubt determined using different analytical procedures. The concentration of each contaminant used in this report (Table 4) are the means of the open water vaues given in

Table 3. Fluvial inputs into Lake Erie. Data for U.S. rivers from Sonzogni et al, (1978) and Canadian rivers from Environment Canada surface water data (1982).

River	Avg. Annual Flow (l/yr)	% of tributary input
U.S.		
Black	3.673×10^{11}	0.17
Belle	1.063×10^{11}	0.05
Clinton	4.875×10^{11}	0.23
Rouge	2.526×10^{11}	0.12
Stony Cr.	6.780×10^{10}	0.03
Raisin	7.468×10^{11}	0.35
Huron (Mi.)	1.498×10^{11}	0.71
Ottawa	1.195×10^{11}	0.06
Maumee	4.456×10^{12}	2.11
Portage	3.876×10^{11}	0.18
Sandusky	1.043×10^{12}	0.50
Huron (Oh.)	2.848×10^{11}	0.14
Vermillion	2.223×10^{11}	0.11
Black	3.384×10^{11}	0.16
Rocky	2.510×10^{11}	0.19
Cuyahuga	8.940×10^{11}	0.42
Chagrin	3.144×10^{11}	0.15
Grand	1.164×10^{12}	0.55
Ashtabula	1.492×10^{11}	0.07
Conneaut	2.501×10^{11}	0.19
Cattaraugus	8.262×10^{11}	0.39
Buffalo	5.144×10^{11}	0.39
Tonawanda	6.269×10^{11}	0.30
U.S. Total	1.398×10^{13}	6.64
Canadian		
Sturgeon Creek	5.267×10^{9}	0.002
Kettle Creek	1.055×10^{11}	0.05
Catfish Creek	1.085×10^{11}	0.05
Big Otter Creek	2.611×10^{11}	0.12
Big Creek	5.046×10^{11}	0.10
Lynn River	5.046×10^{10}	0.02
Nanticoke Creek	4.226×10^{10}	0.02
Grand River	1.807×10^{12}	0.86
Canadian Total	2.588×10^{12}	1.23
Detroit River	1.938×10^{14}	92.11
Total fluvial input	2.104×10^{14}	91.32
Precipitation	2.0×10^{13}	8.68
Total Input	2.304×10^{14}	100.00

Appendix 1. In an attempt to establish a value the concentration in harbors or obviously contaminated sites were disregarded. The resulting values for resident pollutant loads must therefore come with a caveat that they represent only rough estimates, but are probably within the correct order of magnitude. The concentrations used in calculations are given in Table 4 along with each contaminant in the water.

Table 4. Resident load of contaminants in the water of Lake Erie

Contaminant	Concentration (ppb)	Resident in water (kg)
Arsenic	0.35	1.641×10^5
Cadmium	0.20	9.380×10^4
Chromium	1.60	7.504×10^5
Copper	1.75	8.208×10^5
Lead	1.50	7.035×10^5
Mercury	0.61	2.861×10^5
Nickel	3.50	1.642×10^6
Zinc	5.50	2.580×10^6
PCB	0.03	1.304×10^4

3) Sediment

Like the water, there exist many different values for the concentration of pollutants in the sediment. However, unlike the water the sediment types in the three different basins of Lake Erie are sufficiently distinct that they can be dealt with individually. The different sediment types have different densities and different depths to which the pollutants remain available to organisms and the water above. To calculate the amount of a contaminant which is available to the water and biota, the area of each basin (1.24×10^9, 1.01×10^{10}, and 4.29×10^9 m^2 for the western, central, and eastern basin, respectively) (Kemp et al., 1977) was multiplied by the working depth for each basin (11,6, and 3 cm for the western, central, and eastern basins respectively [Kemp et al., 1977]). This value was then multiplied by the average bulk density of the sediments of the three basins, 2.65 g/cm^3 (Kemp et al., 1977). The result is the total amount of active sediment in each basin, i.e. the amount of sediment that is available to

physical, chemical, and biological activities that might resuspend the sediments (Dolan and Bierman, 1982). This work is summarized in Table 5. The results of these calculations were then multiplied by the average concentration of pollutant in the sediment of each basin. Values of concentrations in the sediment were derived by averaging those values in the literature which did not represent harbors or ports where pollutant levels are generally much higher than typical lake sediments. Like the data for the water borne pollutants, these are weakened by the fact that they repesent many sources, originally collected for a variety of purposes, from many sites, by many methods, and analyzed by many means. The same caveat regarding a rough estimate must again be made for these values.

4) Biota

Data for this estimate were originally envisioned as coming from estimates of total phytoplankton, zooplankton, and fish biomass and levels of contaminants associated with these. Analysis of the mass of pollutant leaving the lake via fish harvest (Section III) suggests that the biota are not necessarily a major reservoir of these contaminants. A problem with an estimate of the biotic reservoir of heavy metals is that there are few estimates for either the overall biomass of the components or for the concentration of the metals in these components. Bioaccumulation will be discussed in Section V. Consequently, the importance of the biotic reservoir remains unknown.

Table 5. Resident load of contaminants in Lake Erie sediments.

Basin	Area(cm^2)	Working Depth (cm)	Bulk density (g/cm^3)	Weight of active sediment (g)
Western	1.24×10^{13}	11	2.65	3.61×10^{14}
Central	1.01×10^{14}	6		1.61×10^{15}
Eastern	4.29×10^{13}	3		3.41×10^{14}

Metal	Basin	Concentration (ppm)	Total basin mass (kg)	Total lake mass (kg)
Arsenic	Western	3.58	1.29×10^{6}	
	Central	2.06	3.32×10^{6}	
	Eastern	1.22	4.16×10^{5}	5.03×10^{6}
Cadmium	Western	3.75	1.35×10^{6}	
	Central	2.80	4.51×10^{6}	
	Eastern	2.40	8.18×10^{5}	6.68×10^{6}
Chromium	Western	73.66	2.66×10^{7}	
	Central	59.00	9.50×10^{7}	
	Eastern	30.00	1.02×10^{7}	1.32×10^{8}
Copper	Western	49.60	1.79×10^{7}	
	Central	46.00	7.41×10^{7}	
	Eastern	23.00	7.84×10^{6}	9.98×10^{7}
Lead	Western	73.16	2.64×10^{7}	
	Central	64.90	1.04×10^{8}	
	Eastern	81.00	2.76×10^{7}	1.58×10^{8}
Mercury	Western	0.98	3.54×10^{5}	
	Central	0.44	7.08×10^{5}	
	Eastern	0.34	1.16×10^{5}	1.18×10^{6}
Nickel	Western	51.75	1.87×10^{7}	
	Central	55.00	8.86×10^{7}	
	Eastern	38.50	1.31×10^{7}	1.20×10^{8}
Zinc	Western	167.66	6.05×10^{7}	
	Central	125.00	2.01×10^{8}	
	Eastern	95.00	3.24×10^{7}	2.94×10^{8}
PCB	Western	0.25	9.03×10^{4}	
	Central	0.07	1.12×10^{5}	
	Eastern	0.09	3.07×10^{4}	2.33×10^{5}

IV. CONTAMINANT BUDGET FOR LAKE ERIE

1) Introduction

Mass balance studies attempt to identify the principle points of material storage and movement within a system. Such knowledge allows for the calculation of such parameters as retention times and sedimentation rates. For this report the mass balance approach is used to put dredging into perspective with the other major sources and exports of contaminants of Lake Erie. It is not meant to be the definitive mass balance study of the contaminants of the lake.

For any lake, the simplest equation to describe the mass balance of an element within the system is:

$$T = R + I - C$$

where T is the total mass of the element in the lake, I is the loading from all sources, R is the resident mass of the element in the water and sediment, calculated in Section II, and C is the mass of the element leaving the lake in its outflow. For this project, the three variables I, R, and C, have been further subdivided so that all major processes in the pollutant cycle can be identified. In so doing, weaknesses in the data base have come to light, a matter which is discussed in the final section of this chapter.

2) Contaminant loading into Lake Erie

For this report the loading into Lake Erie is defined as (modified from Jennett et al., 1980):

$$I = P + A + E + D + S$$

where P is direct point source loading, A is tributary loading, E is loading from shoreline erosion, D is the amount of material entering the lake via dry and wet fall, and S is the amount of heavy metals introduced from dredge material. An important consideration in contaminant load assessment is the large dilution capacity of the lake. Consequently, large loadings are required before contaminants are analytically detected. Significant changes may therefore occur before a change is noticed.

a) Point Source

Point source loading for this report has been defined to include industrial and municipal discharges into Lake Erie. Unfortunately, current municipal data for the heavy metals in question are either not collected in the United States or are unavailable. Industrial loadings for the United States are found in an IJC report (1979). Point source loading estimates for the Canadian portion of the lake are available (Sudar, 1977) which include both municipal and industrial loading for all heavy metals with the exception of arsenic and mercury. The combined data from these sources can be found in Tables 6 and 7. Nickel is the most abundant heavy metal, primarily from industrial sources, followed by zinc and copper. Mercury and cadmium have the lowest levels of loading from these sources. For all heavy metals in question the importance of this loading source is minimal (Table 7) accounting for no more than 4.5% of the total loading. It is doubtful that the inclusion of municipal loadings from the United States would significantly affect the results. Data for PCB loading from point sources are not available.

b) Tributary

It has been estimated that the tributaries of Lake Erie contribute at least 95% of the contaminants into the lake (Nriagu et al., 1979). The importance of the Detroit River cannot be over-emphasized as the source for most of the heavy metal loading into Lake Erie. The Detroit River serves as the link from the upper Great Lakes and is consequently carrying the water borne pollutants from Lakes Superior, Michigan and Huron as well as receiving the industrial and municipal discharges from Detroit and the surrounding communities. Other tributaries contribute relatively minor amounts of the contaminants. This is not to say that they do not have localized impacts, but by comparison they do not add a significant amount of contaminants to Lake Erie; most contaminants enter Lake Erie via the Detroit River. The Detroit River carries 1.43×10^7 kg of zinc each year, some 89% of the total loading for this metal. Virtually the entire mercury load, 99.8% (Table 7), also enters via the river. The only exception is arsenic, with the river contributing only 31% of the total load. Over 96% of the PCB's enter the lake via the Detroit River.

c) Shore erosion

Data for Canadian shore erosion and its contribution of heavy metal loading into Lake Erie is detailed in Thomas and Haras (1978). Data for the United States shore erosion are not available. Only arsenic has shore erosion as a significant source, yielding 47% of the total load into the lake (Table 7), though only 8.5×10^4 kg are added each year. Zinc has the greatest mass loaded into the lake each year, 3.98×10^5 kg (Table 6), but this represents only 2.5% of its total load. Mercury has the lowest actual load, 1.65×10^2 kg (Table 6) and per cent load, 0.03% (Table 7), added from shore erosion.

d) Dryfall and wet fall

It is only within the last decade that the atmosphere has been recognized as a significant source of heavy metal loadings into lakes. Eisenreich (1982) has compiled most of the existing information on atmospheric loading into the Great Lakes. Data for arsenic loading are from Traversy et al. (1975) and data for zinc are from Kuntz (1978), while there remains no details on the loading for chromium. The atmosphere is a significant source of cadmium (39.6%), lead (37.12%), and arsenic (11.11%) loading into Lake Erie. Zinc has the highest mass loaded (1.0×10^6 kg) but this does not represent a significant source. Of the seven heavy metals for which data are available, nickel is least affected by the atmosphere as a loading source. Though atmospheric deposition is the major source of PCB to Lakes Michigan (Murphy & Rzeszutko, 1977) and Superior (Swain, 1980; Eisenreich, et al. 1981), Lake Erie receives a far smaller proportion due to its much smaller surface area, receiving PCB loading from the Detroit River.

e) Dredge material

The addition of contaminants into Lake Erie from dredge material was discussed in the recent "Guidelines and Register for Evaluation of Great Lakes Dredging Projects" (1982). Using dredge data found in that report in the present evaluation, the conclusion is that dredging is not a significant source of heavy metals into Lake Erie, with no metal having a percent load greater than 6.5% (Table 7). In comparison, the 1982 'Guidelines' found that

Table 6. Inventory of sources of contaminants in Lake Erie (flux rate, kg/yr).

Source	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Zinc	PCB	Total
Point Source (P)	7.87×10^3	3.56×10^2	7.54×10^3	2.33×10^4	1.07×10^4	6.20×10^1	1.80×10^5	4.73×10^4	-	2.77×10^5
Tributary (A)	5.62×10^4	1.05×10^5	2.02×10^6	1.66×10^6	1.06×10^6	5.23×10^2	3.88×10^6	1.43×10^7	1.07×10^5	2.36×10^7
Shore erosion (E)	8.50×10^4	1.10×10^4	3.29×10^5	1.33×10^5	1.50×10^5	1.65×10^2	1.60×10^5	3.98×10^5	-	1.30×10^6
Dry or wet fall (D)	2.00×10^4	8.00×10^4	-	1.50×10^5	7.50×10^5	4.00×10^3	8.00×10^4	1.00×10^6	3.10×10^3	2.08×10^6
Dredge material (S)	1.10×10^4	5.82×10^3	7.49×10^4	6.45×10^4	6.30×10^4	6.80×10^2	7.95×10^4	2.43×10^5	8×10^2	5.42×10^5
Total	1.80×10^5	2.02×10^5	2.43×10^6	2.03×10^6	2.03×10^6	5.28×10^5	4.38×10^6	1.60×10^7	1.11×10^5	2.78×10^7

- data not available

Table 7. Per cent contribution of souces of contaminants into Lake Erie. Total percentage may not equal 100% due to round-off errors.

Source	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Zinc	PCB	Average % Input
Point source	4.37	0.18	0.31	1.15	0.53	0.01	4.11	0.30	-	1.00
Tributary	31.22	51.98	83.47	82.17	52.47	99.05	90.02	89.37	96.40	84.90
Shore erosion	47.22	5.45	13.60	6.58	7.43	0.03	3.71	2.49	-	4.68
Dry or wet fall	11.11	39.60	-	7.43	37.12	0.76	1.86	6.25	2.79	7.49
Dredge material	6.11	2.88	3.10	3.19	3.12	0.13	1.84	1.52	0.72	1.95
Dredge material reported in Dredging Guideline 1982	-	13	9	11	6	10	10	8	-	-

- data not available

many of the contaminants contributed by dredging were in excess of 10% of the total load. The difference in estimate of importance relegated to dredging in this report and the 1982 report is due to the inclusion of more loading sources in the current study. When placed in the perspective of the total loading from all sources, dredging becomes a relatively minor source regardless of consideration as dredging as a new source of contaminants or re-suspender of in-place contaminants. However, this is not to discount the importance of a source responsible for 5.42×10^5 kg of heavy metals in Lake Erie each year. While representing a small portion of the total volume of heavy metal loading (Table 7) dredging might best be regarded as a highly controlable point source which may result in significant localized impacts.

3) Contaminant exports from Lake Erie

a) Tributaries

The major outflows from Lake Erie are the Niagara River and the Welland Canal. Kuntz and Chan (1982) have reported on the heavy metal concentrations in the water leaving the lake. Annual mass values were computed by multiplying the annual river discharge by the concentration of the heavy metal in the water. The Welland Canal was not considered due to its relatively small outflow in comparison with the Niagara River. The Niagara River is the major hydraulic outflow from Lake Erie for mercury (96%), cadmium (91%), nickel (80%), and copper (73%), and is significant for the other metals as well (Tables 8 and 9).

b) Fish harvest

One source of contaminant export not usually considered is the loss due to the harvest of fish from the lake. Mean values for the contaminant concentration in the flesh of each of the commercially important species (Appendix 4) was multiplied by the latest data for fish harvest. The results of the analysis (Tables 8 and 9) suggest that this is not a major source of contaminant loss from Lake Erie with most contaminants losing less than 0.1% in this manner. Mercury and PCB's are the sole exceptions with exports of 0.17% and 0.33%, respectively. In all cases but zinc the total mass loss was less than 30 kg.

c) On-land or confined disposal

Heavily polluted dredge materials are normally disposed of on land or in confined disposal sites in the water. The most recent data for this type of disposal including estimates of the mass of heavy metal exported are reported in the 1982 Dredging Guidelines. From these data, it can be concluded that of the contaminants removed from the lake each year, on-land and confined removal of spoils is a significant means of contaminant export from the lake. This is especially true for PCB's (82.89%), lead (57.69%), chromium (55.67%), and zinc (50.70%). Little mercury (3.99%, 7.3×10^2 kg) or cadmium (9.64%, 1.87×10^4 kg) is lost in this manner.

d) Volatilization

Mercury is unique among the heavy metals investigated in that it is relatively volatile (Bertine and Goldberg, 1971). Using an estimate of 1.7×10^{-5} g/m^2/yr (Environment Canada, 1981) as the flux of mercury from

Table 8. Inventory of exports of contiminants from Lake Erie. (flux rate, kg/yr)

Source	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Zinc	PCB
Niagara River & Welland Canal	7.02×10^4	1.76×10^5	1.76×10^5	5.27×10^5	1.76×10^5	1.76×10^3	5.27×10^5	1.05×10^6	8.78×10^2
Fish harvest	1.10×10^0	1.24×10^0	1.30×10^0	8.54×10^0	3.09×10^0	5.07×10^0	1.75×10^0	2.74×10^2	1.75×10^1
On land or confined dredge material removal	3.66×10^4	1.87×10^4	2.21×10^5	1.92×10^5	2.40×10^5	7.30×10^2	1.31×10^5	1.08×10^6	4.36×10^3
Volatilized						4.36×10^2			
Total	1.07×10^5	1.94×10^5	3.97×10^5	7.19×10^5	4.16×10^5	2.93×10^3	6.58×10^5	2.13×10^6	5.26×10^3

Table 9. Per cent contribution of export sources out of Lake Erie. Total percentage may not equal 100% due to round-off errors.

Source	Arsenic	Cadmium	Chromium	Copper	Lead	Mercury	Nickel	Zinc	PCB
Niagara River & Welland Canal	65.60	90.72	44.33	73.30	42.31	60.07	80.09	49.30	16.69
Fish harvest	0.00	0.00	0.00	0.00	0.00	0.17	0.00	0.01	0.33
On land or confined dredge material removal	34.20	9.64	55.67	26.70	57.69	24.91	19.91	50.70	82.89
Volatilized	0.00	0.00	0.00	0.00	0.00	14.88	0.00	0.00	0.00

open water, there is a loss of 4.36×10^2 kg from Lake Erie each year. While representing 14.9% of the total export of mercury, this is a very crude estimate whose value is dependant upon the concentration of mercury in the water and the water temperature.

4) Heavy metal and PCB retention time calculations

The mass of contaminant retained in a lake annually is calculated by:

$$M_T = I - C$$

where M_T is the mass of contaminant retained annually, and I and C are the masses of the contaminant loaded and exported annually, respectively. The results of this calculation give a value which is presumed to represent the amount of contaminant sedimenting each year. It is generally accepted that sediments act as sinks for contaminants (Mathis and Cummings, 1973; Mathis and Kevern, 1975; Enk & Mathis, 1977; and many others) and that the vast amount of contaminant entering the water accumulate in the bottom sediments (Gardiner, 1974). The results of this analysis are presented in Table 10. These values suggest that almost all of the mercury is retained annually in the sediments (99.41)%. This result is due primarily to the low concentration of mercury found in the headwaters of the Niagara River, the main discharge from the lake and is in agreement with the conclusion of Thomas (1974) who suggests that little of the mercury input leaves via the Niagara River. In addition, this value has been corrected for mercury leaving the sediments and entering the water due to volatilization (Environment Canada, 1981) at a rate of 1.73×10^{-3} g/m^2/yr. Cadmium, in comparison, has a very low retention value (3.96%) indicating that it is in near equilibrium in the lake. Values for the other metals fall within this range with most in the 65-85% retained range. In comparison Nriagu et al. (1979) calculated values of 35%, 50%, and 65% for zinc, copper, and lead while in this report values of 87%, 65% and 80% were obtained. Baier and Healy (1977) calculated a 70% retention level for lead in Lake Washington. The actual values for mass of heavy metal retained are very similar to those reported by Nriagu et al. (1979) who report mass values of 1.15×10^6 kg and 1.25×10^6 kg for copper and lead, respectively. The retained PCB's represent 95% of the input. This result is almost identical to the per cent value calculated from the data of Eisenreich et al. (1980) for Lake Superior. Nisbet and Sarofin (1972) also suggest that most of the PCB input to freshwater is bound to the bottom sediments.

The retention time of a contaminant in a lake is calculated by (Bowen, 1975):

$$\tau = W/I$$

where τ is the retention time, W is the total mass of contaminant in the lake's water (calculated in Section II), and I is the total annual input of the contaminant into the lake. The residence time of the contaminant relation to the lake water residence time is then

$$\tau_r = \tau/R$$

(Stumm and Morgan, 1970). The value of the residence time for lake water is 2.10 years, as calculated in Section II. The results of these calculations are presented in Table 11.

Values obtained range from that for arsenic of 333 days (0.91 yrs) to that for zinc of only 59 days (0.16 yrs). Most of the retention times range between 113 days (0.31 yrs) and 199 days (0.55 yrs). These values do not differ significantly from those calculated by Nriagu et al. (1979). As suggested by these authors, the short retention time of these metals compared to the flushing time of the lake's waters (2.10 yrs) indicates that the biogeochemical cycling of these metals is rapid. The relative resident time values all smaller than 1 indicate that these heavy metals are readily incorporated into sediments and become unavailable (Stumm and Morgan, 1970). The transfer at heavy metals into the sediments generally exceeds the transfer out (Jennett et al., 1980). Though the retention times differ markedly, the very low value for relative retention time of PCB is very similar to that of Lake Superior (0.03 vs. 0.056, calculated from Eisenreich et al., 1980). The value is influenced by the biota, mixing relationships, and exchange with the sediments (Leckie and James, 1974). It should be remembered that calculations of such characteristics are based on measures subject to a great range of error, so that the values for retention time are often imprecise (Bowen, 1975). Retention times within the same order of magnitude are generally recognized as being similar (Bowen, 1975).

5) Weaknesses in the data base

As suggested in the last chapter, the calculations can be no better than the data upon which they are based. In this instance there are a number of variables which have not been discussed or which have been poorly considered due to the paucity of data. One example of this is the lack of information on inputs via non-point source loading. Also absent are data on loading due to shoreline erosion on the United States portion of Lake Erie's shores. Considering the importance of loadings estimated from Canadian shores, the lack of data in this area may be critical. There is also a lack of data on point source loading, primarily from municipalities. This may not merit as much attention as estimates suggest that municipal loadings are not an important relative to the other loading sources.

Table 10. Mass of contaminants retained annually in the sediments of Lake Erie.

Contmainant	Mass retained (kg)	% of input retained	% of input lost
Arsenic	7.3×10^4	40.56	59.44
Cadmium	8.0×10^3	3.96	96.04
Chromium	2.0×10^6	83.67	16.33
Copper	1.3×10^6	64.58	35.42
Lead	1.6×10^6	79.51	20.49
Mercury	5.2×10^5	99.41	0.38
Nickel	3.7×10^6	84.97	15.03
Zinc	1.4×10^7	86.88	13.12
PCB	1.1×10^5	95.26	4.74

Table 11. Retention times of contaminants in Lake Erie.

Contaminant	Total Annual Load (kg)	Total Resident Mass (kg)	Retention time (days)	Relative retention time
Arsenic	1.80×10^5	1.64×10^5	333	.434
Cadmium	2.02×10^5	9.38×10^4	169	.220
Chromium	2.43×10^6	7.50×10^5	113	.147
Copper	2.03×10^6	8.21×10^5	148	.193
Lead	2.02×10^6	7.04×10^5	127	.166
Mercury	5.28×10^5	2.86×10^5	199	.260
Nickel	4.38×10^6	1.64×10^6	137	.179
Zinc	1.60×10^7	2.58×10^6	59	.077
PCB	1.11×10^5	1.30×10^4	43	.056

V. THE FATE OF CONTAMINANTS IN LAKE ERIE

1). Introduction

The ultimate fate of contaminants in Lake Erie, or any lake, is the result of integrated processes involving physical, chemical, and biological mechanisms. Within each of these broad categories of mechanisms, there are a number of processes which need to be considered. These are detailed in Table 12 (modified from Jennett, et al., 1980). The physical and chemical factors have been studied with increased intensity in the last decade so that, for instance, the aquatic chemistry of many metals is fairly well understood. Table 13 lists the possible forms in which metals may occur in the aquatic environment. The form in which the heavy metals occurs strongly influences its toxicity and mobility (Saxena and Howard, 1977). Because of the increasing concern for the possible impact of contaminants on ecosystems it is desirable to understand the interaction of dredging and the processes involving contaminant cycling.

Sly (1977) outlined the influence of dredging on the Great Lakes. He concluded that "as a result of particle settling and dilution, elevated concentrations [of contaminants] decreased rapidly and background conditions in the overlying water were generally re-established within a few hours. Because of wave activity in Lake Erie the dumped materials were rapidly redistributed and no evidence was obtained to indicate a long term influence on water chemistry". The same conclusion regarding the effects of dredging has been reached by Lee (1977). Sly (1977) also states "Dredging effects and ship turbulence are undoubtedly significant at the local level in the Great Lakes; however, the scale of events remains small in comparison to sediment resuspension resulting from wind-wave action." The purpose of this section will be to review physical, chemical, and biological processes with which dredging activities may interact and attempt to put these considerations in perspective to Sly's comments.

2) Physico-chemical processes

a) Particulate matter and adsorption

Much of the following analysis will emphasize the role of fine-grained sediments and their relationship with contaminants. Such sediments compose 90% of the deposited material in Lake Erie (Kemp et al., 1976). Fine-grained particles are of importance due to their large surface area and hence large adsorptive capacity (Lee, et al., 1981). Dredging resuspends large quantities of such particles at both the dredging site and, in the case of open lake disposal, at the disposal site. This is an important fact as smaller-sized particles have the slowest settling time and are therefore subject to greater dispersion at both the dredging and disposal sites (Weber et al., 1982).

While it is generally accepted that many contaminants adsorb onto particulate matter, or carrier substances (Förstner and Whittman, 1979), especially smaller particles (Hart, 1982; Lee et al., 1981; Vuceta and Morgan, 1978; Jaquet et al., 1978; and many more) there remains much to be understood about many of the details regarding adsorptive processes (Vuceta and Morgan, 1978). Such knowledge is important if the fate of contaminants is to be

Table 12. Mechanisms regulating the fate of contaminants in Lake Erie.

Physical Factors
Temperature
Hydrodynamics and mixing
Chemical Factors
Acid-base (pH)
Complexation
Oxidation-reduction (redox, E_h)
Sorption-desorption
Precipitation-dissolution
Biological Factors
Methylation
Degradation
Bioturbation
Accumulation

Table 13. Forms of heavy metals present in the sediment-water system (Gambrell et al., 1976)

Soluble free cations
Soluble organic or inorganic complexes
Easily exchangable cations
Precipitates of metal hydroxides
Precipitates with ferric oxyhydroxides
Insoluble organic complexes
Insoluble sulfides
Residue metals

understood and if control over contaminants via adsorptive or interfacial reactions (James and MacNaughton, 1977; Vuceta and Morgan, 1978) is to be achieved.

The vast majority of particles of importance to contaminant-particle interactions are those in the size distribution of clay. There is increasing evidence suggesting that the particles are likely to act primarily as substrates coated with metal oxides or a thin film of natural organics (Hart, 1982). Consequently, the understanding of adsorption is dependent upon knowledge of the interaction between metal oxides and organic matter (Davis and Leckie, 1978). In water, metal oxides are coated with surface hydroxyl groups (Stumm and Morgan, 1981). The specific adsorption of cations and anions can be treated as reactions with surface bound hydroxyl groups at the water-oxide interface. The rate of adsorption is initially rapid with equilibrium established within hours.

Although there are many explanations for metal sorption-desorption processes (Westall and Hohl, 1980) there is general agreement that adsorption can be considered analagous to the formation of soluble complexes with the surface sites thought of as ligands that can bind ions to the extent dependent on (Vuceta and Morgan, 1978; James and MacNaughton, 1977):
1) the types and concentrations of reacting species;
2) types of adsorbants and available surface area;
3) pH; and
4) types and concentrations of organic and inorganic ligands present.

Vuceta and Morgan (1978) have demonstrated that these four factors are interactive. The extent of adsorption of dissolved Cu(II) species depends strongly upon the availability of organic ligands and surfaces. When complexing agents are absent, Cu(II) is substantially removed by adsorption. The presence of complexing agents results in partial or total release of adsorptive copper. The distribution of Pb(II) depends on the availability of surface area. Ni(II) and Zi (II) are expected to be present in solution as Ni^{2+} and Zn^{2+} with an increase in available surface area and organic ligand concentration resulting in some adsorption and complexation. Nickel is usually found in a colloidal state (Hem, 1967) although nickel salts are highly soluble in water. Most Cd(II) is present as Cd^{2+} ions though at proper alkalinity and pH some will precipitate as $CdCO_3$. Particulate matter in water often contains 5 to 25 times more mercury than the water itself (Hinkel and Learned, 1969) as mercury is rapidly adsorbed to alluvium (Dall'Aglio, 1971; D'Itri, 1972). Most of the mercury binding is due to low molecular weight ligands (Ramamoorthy et al., 1977). The same is true for cadmium (Nordberg, 1974) with most cadmium adsorbed to the humus component of the sediment (Babich and Stotzky, 1978). PCB's are known to adsorb strongly and rapidly onto the surfaces of suspended particles and at the water-sediment interface (Eisenreich et al., 1980).

In an oxidizing environment Fe and Mn are oxidized to Fe(III) and Mn(IV) and precipitate as $Fe(OH_3)$ (s) and MnO_2 (s) with the presence of adsorbing surfaces or complexing agents only slightly decreasing the degree of precipitation. It is significant to note that both of these precipitates adsorb significant amounts of trace metals (Stumm and Morgan, 1970) and thus act in a scavenging role.

The importance of pH in the process of adsorption has been investigated (James and Healy, 1972; MacNaughton and James, 1974; James and MacNaughton, 1977). The major conclusions are:

1) Uptake is strongly dependent on pH, occurring over a narrow pH range;
2) The dependence of the fractional adsorption on pH is similar to the dependence of the fractional formation of soluble and insoluble hydrolysis products on pH;
3) The shape of the fractional adsorption-pH curve shows that OH^- is consumed or H^+ released as adsorption increases; and
4) The location of the adsorptive region on the pH scale is characteristic of metal ion and its complexes, and relatively insensitive to the adsorbent.

Jaquet et al. (1982) emphasizes that the assumption that fine-sized particles carry contaminants must be refined as different metals have different affinities for different sized particles, at least in their studies of Lake Lemans. They found a significant correlation between a chemical variable and a particular size of particle, suggesting preferences for different fractions. In Lake Erie, zinc has been found to have significant correlation with the size fraction of particles (Alther and Wyeth, 1981).

b) pH and redox potentials

The role of pH and redox in heavy metal availability cannot be overemphasized. Hem (1975) demonstrated the importance of the relationship between these factors and the chemical state of the heavy metals being considered. In addition, the alkalinity of the water acts as a third interacting variable so that 3-dimensional plots are required to demonstrate the full range of chemical states of heavy metals possible. Conditions found in Lake Erie aqueous and sediment environments limit what metal states may occur amongst all possible states so that not all states occur in the natural environment.

In Lake Erie, and the Great Lakes, dredging activites most likely have little effect on pH-redox potentials of the bulk solid materials (Gambrell et al., 1976). In addition, there is little opportunity for most of the disturbed, but not removed, reduced bulk material to significantly oxidize before resettling to the bottom at an open lake disposal site (Gambrell et al., 1976). Some changes in dredged material pH and redox potential may occur in the interstitial water components and solids when mixed with oxygenated surface water for extended periods and to bulk solids transported to land for disposal.

During disposal the mixing of large quantities of oxygenated surface water with reduced sediments and interstitial water may effect pH and redox potentials and subsequently the solubility of trace and toxic metals. In addition, the pH of surface waters and dredged solids may be altered due to mixing of materials of different pH during dredge material disposal or by chemical transformations of predominant reactive compounds affected by change in the oxidation levels. Changes in surface water pH during dredge material disposal has been noted to vary from slight increases to slight decreases (May, 1973; 1974; O'Conner, 1976; Weber et al., 1982). The oxidation of reduced material was found to cause approximately 12% of the oxygen depletion in the central basin of Lake Erie (Burns and Ross, 1972). This figure would

probably be increased with the addition of reduced metals from dredged materials and demonstrates that metals already in the active sediment layer may play a significant role in the oxygen concentration dynamics of Lake Erie. Such effects should be a consideration in the location of open water disposal sites.

c) Hypolimnetic anoxia

The study of hypolimnetic anoxia in relation to dredging is important not because of the direct biological effects of oxygen depletion, but rather because reducing conditions may mobilize heavy metals in the sediment. Redox and anoxia are separable phenomena; while anoxic environments will always have a reducing atmosphere, a reducing atmosphere is not necessarily an indication of anoxia. For such conditions to be important to water chemistry the oxygen concentration at the sediment surface must fall below 1-2 mg/l (Mortimer, 1971). In nature, the western basin of Lake Erie usually has sufficient mixing, due to wind and river inflow, to prevent anoxic conditions and the resulting reductive atmosphere (Potos, 1970). The eastern basin, though stratifying, typically has no problem with anoxia as it has a large hypolimnetic volume and low oxygen demand exerted by the sediments (Leutheuser, 1981). The central basin, however, stratifies with a thermocline near 17 m resulting in a shallow thermocline of only 2.5 m (Burns, 1976; Dobson and Gilbertson, 1972). The hypolimnion is possibly depleted of oxygen primarily by the sediments due to the large sediment surface area to hypolimnion volume ratio (Lucas and Thomas, 1972).

Anoxic waters overlying sediments generally contain higher heavy metal concentrations than do oxygen rich waters as a consequence of dissolved polysulfides and organic complexes. Interstitial waters of reducing sediments also often contain high heavy metal concentrations, with diffusion from interstitial waters to overlying anoxic waters a possibility (Presely et al., 1972). Any anthropogenic or natural process by which reducing sediments and the overlying water become oxidized leads to pronounced heavy metal mobilization. Consequently, activities such as dredging, with the mixing of anoxic, reduced sediments with oxygenated water, may constitute a local hazard. However, factors such as pH, redox, and alkalinity are critical in models predicting the forms of metals present in an anoxic atmosphere. The question thus becomes: How similar are the conditions of these factors (redox, pH, and alkalinity) during anoxic conditions and during dredging operations (both dredging and open lake disposal)?

Burns (1976) found that during periods when the hypolimnion was oxic in the central basin, the redox potential was very near that of the epilimnion with the pH approximately one unit less (ca. 8.6 vs. ca. 7.3). Under anoxic conditions the redox potential was considerably depressed, reaching a low value of 71 mV and high of 298 mV compared to 400 mV in the epilimnion. Thus, anoxic release and change in metal availability in this reducing water and sediments is a possibility. That sediments have a reductive atmosphere, especially in often dredged areas such as harbors, has been shown by Weber et al., (1982), and is taken for granted. Many of the reactions described for reductive conditions most probably occur in the sediments. However, no studies have attempted to monitor changes in redox potential during dredging activity, though the assumption is made that as dredging material is disposed

in open water it becomes oxidized. pH has been monitored (Weber et al., 1982) at dredge sites and the data indicate very little if any change even while dredge material was being disposed. So eqilibria of heavy metal states with redox-ph considerations need take into account only changes in redox and not pH during dredging activities.

d) Summary

Table 14 summarizes the possible alterations in availability due to dredging activities of the heavy metals considered in this report. In the study by Weber et al. (1982) zinc showed the greatest potential for release in their open water dispersal simulation with cadmium showing the least potential. Zinc and arsenic did not show any additional release under anoxic conditions though arsenic may show a slight increase. An important finding was that most of the heavy metals did not show additional release in merely a reducing atmosphere; the presence of reductive conditions is not solely responsible for the release of these metals. Finally, all of these metals show a considerable tendency for sorption.

Current data do not suggest that Sly (1977) was inaccurate in his assessment of the impact of dredging on the chemical status of Lake Erie. However, the proper monitoring of dredging operations has not yet been conducted whereby the critical factors of pH, redox potential, and alkalinity are measured in relation to the release and potential availability of heavy metals. Such a study would certainly aid in assessing the extent of heavy metal release during dredging activities.

3. Biological Factors

a) Methylation

The metabolic activity of micro-organisms plays a significant part in determining the availability of contaminants in the aquatic environment (Wood, 1974). The methylation of heavy metals, chiefly mercury, came under intense study when methylmercury compounds were found in fish in the mid-1960's. Studies soon implicated bacteria and fungi (loosely called "microbes" in the literature) in the transformation of heavy metals into either reduced (Nelson and Colwell, 1975; Summers and Silver, 1972) or oxidized (Titus et al., 1980), and methylated or demethylated, states. Microbes may also degrade organometallics, such as organomercurials (Billen et al., 1974; Clark et al., 1977; Nelson et al., 1973; Spangler et al., 1973). The importance of these transformations is suggested by the fact that methylmercury (CH_3Hg^+) is 50 to 100 times more toxic than inorganic Hg^{2+} (Summers and Silvers, 1978). Mercury, arsenic, lead, and cadmium have all been found to be methylated by microbes (Saxena and Howard, 1977).

Mercury is transformed to either mono- or dimethylmercury (Jensen and Jernelöv, 1969: Fagerström and Jernelöv, 1971; Wood et al., 1968; Landner, 1971) by nonenzymatic methylation via methylocobalamin which serves as a methyl donor. Other means of methylation are possible (Silver et al., 1976; Vonk and Sijpesteijn, 1973). Dimethylmercury is highly volatile; upon reaching the atmosphere it undergoes photolysis (Gomer and Noyles, 1949) to monomethylmercury which then undergoes decomposition to metallic mercury. In

Table 14. Summary table of the availability of heavy metals under real and potential conditions at dredge and dredge and dredge material disposal sites (Data from Weber et al., 1982).

	As	Cd	Cr	Cu	Pb	Hg	Zn
open water disposal simulation (a)	++	+	++	++	++	*	+++
rank in availability in simulation	2	5	4	6	3	*	1
anoxic release	nd(b)	*	*	*	*	nd	*
redox (c)		?	nd(d)	nd	nd	*	nd
sorption tendency (f)	+++	+++	+++	+++	+++	*	++(e)

(a) dispersion of 1% sediment suspension; +++= major trace; ++ = minor trace; + = subtrace

(b) may favor slight increase

(c) subject to change with ranges typified for Great Lakes aqueous and sediment environments (ca. -150mV to 500mV)

(d) above 350 mV Cr can form toxic Cr(VI) but may not due so due to Cr(III) forming non-labile complexes

(e) dependent upon slurry:water ratio

(f) +++ = strong tendency to sorb; ++ = slightly less tendency

nd = no difference between oxic and anoxic conditions or oxidative and reductive conditions

* = not investigated by Weber et al. (1982).

one sense the methylating organisms may serve a useful purpose as they maintain the environmental methylmercury concentrations at a minimum by turning available methylmercury into volatile elemental mercury (Hg^0) and methane (Saxena and Howard, 1977).

Arsenic similarly undergoes methylation by microbes (McBride and Wolfe, 1971; Cox and Alexander, 1973). Reduction and methylation results in the formation of extremely toxic dimethyl and trimethylarsines. Alkylarsines are very volatile and are rapidly oxidized in the atmosphere.

The methylation of lead and cadmium are not well known. Though it has been suggested that neither can be methylated naturally (Wood, 1974), organic and inorganic lead compounds have been found to be methylated to the volatile and more toxic tetramethyllead by microbes (Wong et al., 1975). Inorganic cadmium is presumably methylated to more volatile forms by Pseudomonas species (Huey et al., 1974).

A number of factors regulate the methylation of the heavy metals in the aquatic envrionment. These factors include:

1) Concentration of the metal in question. Aerobic mercury methylation increases up to concentrations of 100ug/g sediment but then decreases due to inhibition (Jensen and Jernelöv, 1969). This is also the case for anerobic mercury methylation (Bisogni and Lawrence, 1973).

2) Microbial activity. Both the types and their population size are important in determining the extent of methylation though apparently many bacterial species have the capacity for methylation of mercury to some extent (Saxena and Howard, 1977).

3) Microbial acclimation. The rates of methylation increases gradually when metals are initially added to bacterial cultures suggesting that a certain amount of time is required for acclimation (Cox and Alexander, 1973).

4) Adsorption and chelation of heavy metals. The strength of adsorption of heavy metals to particles may slow methylation but does not appear to prevent it (Jensen and Jernelöv, 1969; Olson and Cooper, 1974).

5) Presence of other chemicals. There are three possible effects that other chemicals may cause:
   a) May effect metabolic activity of microbes (e.g., sulfides) (Fagerström and Jernelöv, 1971);
   b) May react with the metal ion rendering them unavailable for methylation; and
   c) May be preferentially methylated.

6) Physical parameters. Three parameters are of importance in regulating methylation. They are:
   a) pH. pH may alter enzyme reactions in the microbes, and as has been discussed previously, may play a role in the availability of particular ions. The optimum pH for the methylation of mercury is 4.5.

b) Temperature. The growth rates of many microbes is temperature dependent and this will inturn effect methylation rates. The rate of methylation does seem to be constant over the range of 10-30°C (Bisogni and Lawrence, 1973) but sediments are often cooler than 10°C.
c) Redox potential. At low redox potentials (-200 mV) mercury ions combine with sulfide forming mercury sulfide which is unavailable for methylation.

Methylation of mercury is accomplished by the normal bacterial flora of the gills and guts of fish (Summers and Silver, 1978). However, the rate of synthesis of methylmercury in the sediments does not have to be very rapid for fish to accumulate dangerous levels (Wood, 1973) from the water. In locations where sediments were polluted with both inorganic mercury and chlorinated hydorcarbons fish were found to accumulate very little methylmercury as the enzyme systems responsible for methylmercury synthesis are inhibited (Wood et al., 1968). Fish, therefore, may accumulate methylmercury from within and without, a fact which should serve to re-emphasize the importance of understanding the impact of contaminants in the environment.

Dredging serves primarily to resuspend mercury and distribute inorganic mercury over a wide area (Wood, 1972). However, in Lake Erie, dredging acts as a major export (See Section III). The relevant question is thus whether or not dredging activities in some way affect the methylation of mercury or other heavy metals in Lake Erie. Data for answering this question are unavailable though they should be an important determinant in dredging decisions. However, the determination of the concentration of methylmercury in the sediment is not sufficient as it does not reflect the rate of synthesis (Wood, 1974). Instead, total mercury in the sediment and rate of methylmercury accumulation in fish are necessary for meaningful decisions (Wood, 1974).

b) Biodegradation of PCB's

The degradation of PCB's by oxidation has been reported to be accomplished by the fungus Rhizopus japonicum (Wallnöfer et al., 1973) and strains of the bacteria Achromobacter (Ahmed and Focht, 1973; Wong and Kaiser, 1975), Alcaligenes (Furukawa et al., 1978; Furukawa and Matsumura, 1976; Yagi and Sudo, 1980), Acinetobacter (Furukawa et al., 1978), Pseudomonas (Sayler and Colwell, 1976; Wong and Kaiser, 1975; Baxter et al., 1975) and Norcardia (Lunt and Evans, 1970; Baxter et al., 1975). In two studies (Wong and Kaiser, 1975; Clark et al., 1979) mixed assemblages of microorganisms were utilized and found to be capable of degrading PCB's. Wong and Kaiser (1975) concluded that less than 1% of the bacteria in lake water were capable of utilizing PCB's for growth. The use of mixed cultures results in more rapid degradation than pure cultures perhaps due to the presence of micro-organisms that could aid the degradation of PCB's by removing potentially inhibitory intermediates (Clark et al., 1979). Moreover, the mixed culture method is probably a much more accurate representation of the activities in a lake than pure culture studies. Both of the later studies concluded that PCB degrading microbes are commonly present, though in limited abundance, in soils and lake sediments.

The general conclusions of PCB degradation studies are:

1) Lower chlorinated biphenyls (less than or equal to 3 chlorine atoms/molecule) are readily degraded while those with higher chlorination are degraded slowly or not at all;
2) The pattern of chlorination will influence the ease of degradation with the position of the chlorine atom in the benezene ring being the determining factor for the rate of degradation; and
3) Many PCB isomers show increasing amounts of degradation as the duration of the experiments increased suggesting that time is required for microbial acclimation.

While it is clear that there are microorganisms present in the aquatic environment capable of degrading PCB's there has been no research which estimates the extent to which this activity is occuring in any lake. Such research is necessary to determine the ultimate fate and duration of PCB's in Lake Erie. In addition, the bacterial flora of fish have been implicated in PCB degradation (Califano, 1979). This process also requires further research.

c) Bioturbation

Bioturbation is the stirring of sediments by the activity of burrowing benthic organisms, primarily inveterbates such as oligochaete worms, insect larvae, crustaceans, and bivalve molluscs (Petr, 1977). The important bioturbaters in Lake Erie are listed in Table 15.

Bioturbation activites include (Petr, 1977):

1) Pumping the enriched interstitial water out of the sediment (release of pore water) and bringing in water to the sediment richer to oxygen;
2) Active transport and exchange of particulate material to the surface and into deeper layers; and
3) Discarding faecal pellets onto the sediment surface.

These activities result in alterations in (Petr, 1977):

1) Chemical and physical changes in the intersititial water and the overlying waters;
2) Removal of nutrients and decomposition processes;
3) Release of heavy metals from sediments; and
4) Deposition of metals (via faecal pellets) into sediments.

Table 15. Principle organisms responsible for bioturbation in Lake Erie (Robbins, 1980; Krezoski et al., 1978; McCall et al., 1979).

Oligochaetes	
Tubifex tubifex	Limnodrilus hoffmeisteri
T. kessleri americanus	Vejdovskyella intermedia
Naididae	
Insect larvae	
Chironomidae	Hexagenia spp.
Heterotrissocladius oliveri	
Crustaceans	
Pontoporeia hoyi	Mysis relicta
P. affinis	
Molluscs	
Sphaerium spp.	Lampsilis radiata siliquoidea
Pisidium spp.	Proptera alata

Irrigation activities of tubificid worms reduced pH in the upper 1-2 cm and increased pH below 2-3 cm in sediments (Davis, 1974). As previously described such pH changes in conjunction with other factors such as redox and dissolved oxygen can result in changes in heavy metal availability. Oxygen exchange is minimal during periods of low oxygen concentration, such as stratification, and activity of tubificids is minimal. With an increase in oxygen concentration activity resumes. The mass transport of oxygen into the sediments by bioturbation can be of considerable importance where large densities of bioturbating organisms exist. At 50,000 tubificids/m^2 the oxidized zone of the sediment extends at least 15 cm into the sediment (Schumacher, 1963). Increases in redox in the sediment have also been reported where benthic organisms are very active (Edwards, 1958; Hargrave, 1972) with the oxidized zone deepened by 0.3 to 1.6 mm (Davis, 1974). Clearly, bacterial activity, such as methylation of heavy metals, will be effected. Both tubificid worms and molluscs have been shown to release the methylmercury formed by microbes in the sediment as they burrow through the sediment (Jernelöv, 1970). Bioturbation in productive areas will result in higher methylmercury concentrations in the overlying water (Petr, 1977) as both inorganic mercury and methylmercury have high affinities for organic substances (Jernelöv, 1975).

Bioturbation is a significant event in the Great Lakes (Robbins and Edgington, 1975). Robbins (1980) suggests that the benthos can play an important role in the cycling of silica, a nutrient critical to limiting the development of diatoms. In Lake Erie, Tubifex tubifex have been found to feed over a range of 0-10 cm in the sediment though most feeding occurs between 5-8 cm (Fisher et al., 1980), with mixing occuring from 6-9 cm as a layer of sediment above the zone of peak feeding moves downward. The process causes a significant amount of sediment mixing wherever the downward velocity of the sediment-water interface caused by tubificid feeding is greater than its upward velocity due to sedimentation. Consequently, tubificids alone are capable of thoroughly mixing the sediments of the western basin, most of the central basin, and some regions of the eastern basin (Fisher et al., 1980). In addition, unionid clams have been estimated to mix 2-6% of the top 10 cm layer of sediment in the western basin each week at 19°C (McCall et al., 1979). The uniformity of profiles of such heavy metals as mercury, chromium, lead, and zinc, to a depth of 20 cm in some instances (Kovacik and Walters, 1973) is taken as evidence that burrowing activity is an important phenomenon in Lake Erie. Mixing profiles are seldom below 5 cm where unionid clams are absent suggesting their importance in deeper sediment mixing (McCall et al., 1979). However, it is not possible to separate the reworking effects of the different benthic components from physical factors such as waves and currents.

d) Bioaccumulation

Having discussed physical, chemical, and biological factors which play a role in determining the availability of contaminants, the next relevant question is whether or not organisms are capable of accumulating these pollutants. The effects that the contaminants may have is discussed in Section VI. Bioaccumulation data for species found in Lake Erie may be found in Appendix III. The majority of these studies, while dealing with taxa found in Lake Erie, were not carried out in Lake Erie.

One significant point in much of the literature concerns semantics, in particular the use of the terms bioaccumulation and biomagnification. While often used to describe the same events, they are actually separable processes. Bioaccumulation is the extent to which an organism collects a compound from its surrounding environment by all processes, while biomagnification indicates that a compound is concentrated through the consumption of lower by higher food chain organisms with a net increase in tissue concentration (Isensee et al., 1973). The two processes will be discussed in the following sections. Bioconcentration factor (BCF) is the concentration of a chemical in an organism, or in the tissue of an organism, divided by the concentration in the water (Kenega and Goring, 1980).

i) Bioaccumulation

A critical point is that many studies, on all trophic levels, have been performed solely to measure concentrations of contaminants in the organisms. Such data are insufficient to determine the ability of organisms to accumulate contaminants via absorption, adsorption, or biomagnification. What is required for the BCF to be calculated is the concentration in the water, sediment, and the organism in question. Without knowing these parameters in the environment the concentration in the organism has limited value. Concentration values for contaminants in fish are presented in Appendix 4. Information was not available to allow the calculation of BCF's of these fish.

Both the mass balance data and physico-chemical data suggest that heavy metals and PCB's are not evenly distributed throughout Lake Erie but are instead associated with suspended particles or with the sediments. Bacteria and algal cells represent living suspended particles that have been demonstrated to be effective bioaccumulators. The ability of microorganisms to bind heavy metals is well known (Adams et al., 1973). Ramamoorthy et al. (1977) demonstrated that the bacteria Pseudomonas fluorescens is actually, per unit surface area, much more active in adsorbing mercury than were the sediment particles. They found that the sediment contained a relatively small number of very high affinity binding sites whereas bacteria contained many more binding sites. Pseudomonas sp. has been shown to accumulate 10 times the amount of cadmium as sediment on a dry weight basis (Titus and Pfister, 1982). Remacle (1980) demonstrated that attached bacteria are 5 times as efficient as free floating bacteria in accumulating cadmium. While he points out that cadmium removal is dependent on bacterial productivity it has been shown (Titus and Pfister, 1982; Guthrie et al., 1977; Cherry and Guthrie, 1977) that the accumulation of cadmium is affected by the physico-chemical condition of the water in the same way that sediment particles are; i.e., lowered redox values accelerate cadmium uptake while high pH values allow cadmium to form insoluble complexes so that accumulation is limited.

Tornabene and Edwards (1972) demonstrated that the accumulation of lead by bacteria was actually due to adsorption rather than absorption. Aquatic fungi similarly adsorb cadmium, lead, and zinc (Duddridge and Wainwright, 1979). Of the lead associated with Micrococcus luteus and Actobacter sp., over 99% was associated with the cell wall and membrane (Tornabene and Edwards, 1972). They also showed that the bacteria were

adsorbing at a rate dependent on the physico-chemical characteristics of the medium. All of the work on bacterial accumulation has been conducted under laboratory conditions. There is a clear need to assess the impact of contaminant adsorption on bacteria in field situations.

Like the bacteria, algal accumulation of heavy metals is dependent on the physico-chemical conditions of the environment (Hassett et al., 1980). Chylamydomonas sp. accumulates cadmium, lead, and mercury more efficiently than either of the blue-green algae Nostoc sp. or Oscillatoria sp. (Hassett et al., 1980). It was suggested that the flagella act as sites of lead accumulation in Chlamydomonas and Platymonas (Hessler, 1974). The accumulation, as in bacteria, appears to be due to adsorption, at least for the planktonic species including greens, blue-greens, and diatoms (Hassett et al., 1980; Denny and Welsh, 1979; Conway and Williams, 1979; Laube et al., 1979). Mercury, lead, and cadmium are all accumulated (Hassett et al., 1980) but neither zinc and arsenic are, though it has been shown that Chlorella will quickly adsorb zinc to its surface in amounts dependent on the concentration in solution (Broda, 1972). Asterionella formosa adsorbs arsenic also as a function of its concentration (Conway, 1978) although this is not the case with PCB's (Lederman and Rhee, 1982). Algae also have the ability to adsorb various chlorinated hydrocarbons (Sikka et al., 1977) as demonstrated by the fact that dead cells take up similar levels as live cells (Urey et al., 1976). Chlorinated organics may also be accumulated by absorption or partitioning (Södergren, 1968; Rice and Sikka, 1973; Reinert, 1972). Bioaccumulation of these compounds varies with the type of compound and the alga in question (Keil et al., 1972; Khan and Khan, 1974; Hollister et al., 1975; Neudorf and Khan, 1975; Schauberger and Wildman, 1977; Paris and Lewis, 1976; Wright, 1978; Harding and Phillips, 1978; Hansen, 1979). Benthic algae may also act as bioaccumulators. Cladophora, common in the Great Lakes, accumulates zinc, cadmium, lead, and copper (Keeney et al., 1976).

Molluscs, especially clams, have been considered as environmental monitors due to their ability to accumulate contaminants. Mathis and Cummings (1973) found three species of clams to concentrate 8 different heavy metals, in amounts exceeding their concentrations in water, but less than their sediment concentrations. Their methods were somewhat irregular in that whole clams were analyzed, including shells, sediments and undigested food in the intestine. Anderson (1977a) similarly used whole ashed animals in his study an concluded that such a technique led to low values for accumulation as the shell contains low heavy metal concentrations. In a more detailed study (Anderson, 1977b) it was found that body concentrations of heavy metals were higher than the shells, but that the shells were possibly acting as an adsorbing surface. The gills were found to have the highest heavy metal concentrations, serving as a site for both adsorption and absorption. Copper, cadmium, and lead were all in lower concentration than in the sediment, whereas zinc was significantly higher. Adams et al. (1981) obtained similar results regarding zinc accumulation and the importance of the gills as the site of highest heavy metal accumulation. Mathis et al. (1979) found that snails and fingernail clams accumulated higher concentrations of cadmium and lead than sediments.

Despite the importance of the zooplankton component in aquatic food webs there is a paucity of information on contaminant accumulation in these organisms. Only Daphnia magna has received attention specifically in terms of bioaccumulation (Biesinger et al., 1982; Poldoski, 1979), and this only for mercury. Accumulation of mercury was found to be highly dependent on the form of mercury in the water with methylmercuric chloride accumulated 20 times more than mercuric chloride. However, about half the mercury was lost within four days of exposure. Denny and Welsh (1979) looking at the zooplankton fraction of their field samples found lead to be significantly accumulated but give no indication of the mode of accumulation. Mathis and Kevern (1975) in their zooplankton samples found cadmium to be accumulated but also included gut contents of the zooplankton in their analyses. It is clear that further work is needed on biaccumulation in the zooplankton, especially in previously ignored groups as the copepods.

Tubificid oligochaetes are an important part of the Lake Erie benthic community. Mathis and Cummings (1973) found that tubificids were accumulating metals at levels very similar to those in the sediments, a finding supported by Mathis et al. (1979). Chironomid larvae, who share a similar niche, were found to accumulate copper, lead, and zinc in levels greater than found in the sediments, while chromium was accumulated less so (Namminga and Wilhm, 1977). It was suggested that zinc accumulation is by surface adsorption and that low values for zinc in this study may be the result of frequent molting and the consequent elimination of adsorbed metals. In both these studies, and those of the zooplankton, the animals were not given an opportunity to clear their guts before analysis. The gut contents of these animals undoubtedly make up a high proportion of their total weight and apparent bioaccumulation may represent nothing more than ingested sediment.

Fish, though intensively monitored for contaminant levels, have not been studied to yield bioaccumulation data. Data on the concentration of the contaminant in the water where the fish have been caught is usually absent so that the means of accumulation of the contaminants cannot be determined. Mathis and Cummings (1973) found that fish were accumulating heavy metals in their muscles at levels above that found in the water but less than those in sediments. Omnivorous and carnivorous species accumulated lead and cadmium at similar levels, but omnivorous fish had significantly higher concentrations of copper, nickel, chromium and zinc. Murphy et al. (1978) found a similar difference between omnivorous buegill and carnivorous largemouth bass with the bluegill having higher concentrations of heavy metal than adult bass. Juvenile, omnivorous, largemouth bass similarly had higher concentrations than did adults. Mathis and Kevern (1975) found that mercury was accumulated in fish but were unable to determine the relative importance of the source, water or food. Essentially all mercury accumulation is as monomethylmercury (Kamps et al., 1972) although inorganic mercury can be absorbed through the gills, intestine or skin (Jernelöv and Lann, 1971). While mercury concentrations in fish were correlated with length no such correlation was present with cadmium or lead (Mathis and Kevern, 1975). Accumulation of mercury is also highly dependent upon temperature (Cember et al., 1978). Similar results have been obtained for cadmium (Lovett et al., 1972) and zinc (Mount, 1964). Arsenic (Sorensen, 1976), cadmium (Mount and Stephan, 1969), and zinc (Mount, 1964)

are all accumulated directly from the water at a concentration dependent upon that of the contaminant concentration in the water. The uptake of PCB's has been noted to be very rapid with 80% equilibrium reached in only 12 hours by larval striped bass (Califano et al., 1980). Other fish have been found to require longer periods, up to 100 days, before reaching equilibrium (Defoe et al., 1978; Stallings and Mayer, 1972). The ability to accumulate PCB's directly from the water has not been compared to the importance of dietary uptake.

ii) Biological Magnification

Studies of biological magnification attempt to elucidate the difference in the role of acquiring contaminants via dietary means and absorption or adsorption directly from the water (partitioning). Such studies fall into two general categories: 1) laboratory ecosystems consisting of a simplified food web or food chain or 2) modelling approaches.

Jernelöv and Lann (1971) estimated that dietary uptake accounted for less than 25% of the methylmercury in bottom feeding fish and 60% in pike. As predatory fish accumulate only 10 - 15% of the mercury found in their prey, (Jernelöv, 1968) the prey of pike must have more mercury than the prey of bottom feeding fish. Benthic invertebrates have been found to contain low percentages of methylmercury supporting this contention (Huckabee and Hildebrand, 1974). However, this result is in contrast to the studies of Mathis and Cummings (1973) and Murphy et al. (1978) who found that omnivorous fish had equal or higher levels of heavy metals. However, neither study determined mercury concentration and it has been shown (Mathis and Cummings, 1973) that uptake of metals is variable.

In a laboratory food chain study it was found that zooplankton accumulated methylmercury 10 to 15 times faster than fish (Huckabee et al., 1975). It was also determined that partitioning was of greater importance for zooplankton than dietary uptake. For fish, however, the relative importance of the water and food is a function of ingestion rate. The conclusion was that feeding rates of fish significantly influence assimilation efficiency. In contrast, it was found that under specific laboratory conditions Daphnia magna accumulated all of its mercury from its dietary source, Chlorella vulgaris, which was virtually 100% efficient at the uptake of mercury from its media. The Daphnia were more efficient at the uptake of methylmercury than mercury chloride with the efficiency increasing with temperature. In addition, the Daphnia retained 10 times the amount of methylmercury than mercury chloride.

The contrasting results of these and other studies, indicates that laboratory research suffers from the limited number of trophic levels investigated. A compounding problem is the fact that the ecosystem exists as an intricate feeding web rather than a linear food chain. In addition, laboratory studies fail to take into account environmental variability in such abiotic factors as temperature and daylength which may seasonally alter behavior and feeding habits. Consequently, the results of such studies must be viewed with caution.

The ecosystem modelling approach has the advantage of allowing the investigator to look at all trophic levels of the ecosystem and approximate the interactions between levels. Such models have been used to (Thomann et al., 1974):

1) investigate the structure of the build-up of potentially toxic substances in the food web;
2) determine the types of data required for a verification of the model; i.e., point out weaknesses in the available data set; and
3) determine the utility and application of linear food chain models in broad scale ecosystem planning.

In such models, the ecosystem is viewed as a series of discrete trophic levels, with inputs and outputs of the various contaminants described by mass balance equations (Thomann et al., 1974). The accuracy of the predictions from such a model, however, will be dictated by the quality of the information used for the variables. The upper trophic levels, especially bird biomass, are admittedly "constructed on tenuous grounds" (Thomann et al., 1974) as the data are simply not available for projections. Also required are partition coefficients and bioaccumulation data in general. Thus these models have served the second use described previously very well. The conclusion is again that the results of the studies must be viewed with the assumptions used in the model in mind.

The difference in the results of the laboratory and modelling approaches can be illustrated by the work on biomagnification of PCB's. It has been reported that the process of biomagnification is relatively unimportant (Scura and Theilacker, 1977; Macek et al., 1979; Harvey and Steinhauer, 1976). These are laboratory results from simplified food chain studies. Weininger (1978) used a modelling approach and determined that partitioning was responsible for only 2-3% of the observed PCB's in lake trout. Another modelling effort gave similar results for a generalized fish top predator (Thomann, 1981). It was concluded that laboratory studies suffered from small sample size and from a failure to use realistic concentrations of PCB's in the experimental environments.

Dredging has been taken into account in modelling studies as well. The accumulation of PCB's by zooplankton has been estimated by a modelling approach that demonstrated the rapid entry of PCB's from suspended sediment desorption into soluble and food web components (Brown et al., 1982). Ninety five percent equilibrium was established among soluble, suspended particle, and food web components within four hours after perturbation with the phytoplankton responsible for most of the entry of PCB's into the food web. The PCB's adsorbed to phytoplankton, bacteria, and other zooplankton food sources, was responsible for increasing the body burden of PCB's in the zooplankton over that which would be expected by sorption and desorption processes alone. Thus, this model likewise suggests the importance of biomagnification in contrast to the laboratory research.

VI. The toxic effects of contaminants on the biota of Lake Erie

1) Weaknesses of toxic effects studies

Biossays and their associated problems have been recently discussed in great detail by Maciorowski et al. (1981; 1982). Problems associated with toxicity tests are primarily associated with the lack of correspondence between the laboratory test system and nature. Laboratory studies, on all trophic levels, suffer from this weakness with such problems as lack of proper control of environmental chambers (e.g., constant light or temperatures, or unrealistic temperatures), starvation of test organisms, and improper monitoring of test media for similarity with natural conditions, common in these studies. The last point is especially important in that the physico-chemical conditions in the test media may alter the availability of the contaminant in a manner unrelated to the natural system. It has been found that even the time that the media should sit prior to use is important for chemical speciation to occur (Allen et al., 1976).

Dutka and Kwan (1981) in a multi-laboratory study of the toxicity of five heavy metals and PCB's found that the test method employed indicated toxicity and end point concentrations varying 100 to 1000 times between laboratories. It was believed that the problem of reproducibility both within and between laboratories was related to the nature of the cell suspensions. A significant conclusion was that the result of any one procedure cannot be readily correlated with the results of other procedures. In addition, no single biological testing procedure can predict the presence of all toxicants which might affect aquatic organisms or be eventually accumulated and affect the food web or chain.

Another highly significant point is the choice of the organisms. The recent literature makes two points in this regard, i.e. 1) the use of laboratory stock cultures is highly unrepresentative of the real world and 2) the use of 'representative' taxa is almost meaningless. The first point is clearly demonstrated by Seyfried (1978) who found that bacteria cultured from areas receiving heavy metal pollution had greater resistance to these contaminants than those cultured from collections in relatively clean sites. In addition, the use of planktonic species would give a false impression as well considering that two-thirds of the mercury resistant strains were found in the sediment. It has been pointed out by Whitten and Say (1975) that the existance of organisms in contaminated water raises the question as to whether organisms belonging to the species are genetically tolerant to a particular contaminant, or that particular organism has evolved mechanisms to tolerate the high concentrations of contaminants as have been documented in the bacteria (Beppu and Arima, 1964; Billen et al., 1974) and algae (Whitten, 1975; Foster, 1982a, b).

The second point has been emphasized in both the phytoplankton and zooplankton. In the phytoplankton it has been found that sensitivity to contaminants varies between algal divisions. As an example, Euglena, in the division Euglenophyta, is more resistant to cadmium than Chlamydomonas, in the division Chlorophyta (Fennikoh et al., 1978). Differences may exist within a division as well as has been demonstrated between Asterionella formosa and

Fragilaria crotenensis, both members of the Bacillariophyta, for toxicity of cadmium (Conway and Williams, 1979). Differences may even extend to the genus level with different species within a genus having different sensitivities (Wong et al., 1979). Again, there may also be differences within species and between habitats. Zooplankton toxicity tests frequently feature Daphnia magna as suggested by the EPA. However, this organism is not representative of its genus (Winner and Farrell, 1976) for sensitivity to contaminants, and certainly not representative of the freshwater crustacean zooplankton in general (Wilson, 1980; Marshall, 1979; Marshall and Mellinger, 1978).

Wilson (1980) has summarized the problems of toxicity tests as:

1) Little research is done on naturally occurring species with extrapolation between species difficult or impossible (Borgmann et al., 1980);
2) A lack of information on temperature effects exists; e.g. the effects of zinc on bluegill with a 10°C temperature shift noted by Cairns and Scheier (1957) versus the results of Rehwoldt et al. (1972);
3) The preponderance of static bioassays;
4) The extreme overcrowding in test chambers represents unnatural conditions for organisms ranging from bacteria to zooplankton; and
5) The lack of replication, and has been previously described, both within and between laboratories.

Hendrix et al. (1982) have suggested that current toxicity tests attempt only to determine the effects of toxicants on particular organisms but fail to evaluate the ecological effects that may 1) arise due to multispecies and species-environment interactions, or 2) propogate through natural systems with consequences far removed from the test organism. Such consequences include altered grazing and predation rates, elimination of key species, food chain magnification, and inhibition of microbial processes. Rather than use single-species assays, they propose that more realistic and ecologically meaningful results might be expected from a testing protocol that builds upon bioassay data with experiments in multispecies microcosms. Increasing complexity of microcosms could be the appropriate scheme, i.e.,

1) Relatively small, static microcosm;
2) Flow-through microcosm; and
3) Detailed but selective studies in more complex microcosms.

2) Evaluation of the elutriate test

For dredging projects, the most important tests have been those which determine the release and availability of contaminants from sediments. To this end, three types of tests have been designed. The bulk chemical sediment analysis of the EPA and elutriate test of the U.S. Corps of Engineers have been discussed in detail by Lee and Plumb (1974) and Prater and Anderson (1977a, b) as well as by numerous other authors. Biologically, the bulk chemical sediment test may not be directly related to the impact of the sediment on the aquatic environment because a substantial fraction of the constituents that it measures are in forms that are unavailable to the ecosystem (Prater and Anderson, 1977a, b). The utility of the elutriate test for the evaluation of possible ecological effects due to dredge material disposal on benthic fauna is suspect as well (Hoke and Prater, 1980). The

elutriate test only simulates the immediate impact of disposal events on the water column (Hoke and Prater, 1980). A third type of sediment bioassay has been employed in different forms by Seeyle et al. (1982) and Hoke and Prater (1980). In this type of test, organisms are placed in a recirculating system with water flowing over a given amount of sediment to determine how much of the contaminants are available and their impact on the test organism. Such tests typically run for 96 hours. Seeyle et al. (1982) slightly modified this approach by keeping the sediment in suspension. They make four recommendations to maintain the quality of toxicity studies of sediment:

1) The test organisms should be of consistent size and origin. The importance of origin has been stressed in the previous section.
2) The sediments to be tested should be handled in a consistent manner both in terms of oxidation and reduction of sediments and in the amount of sediment used;
3) The water used in the test should be of the same hardness and salinity as water from the dredge site. In addition, such factors as pH should be regulated as well; and
4) Both bioaccumulation tests and toxicity tests designed for the use of regulation of dredging and dredge material disposal, particularly where long-term impacts are being examined, should be conducted for substantially longer periods than 10 days, or time series data should be collected to enable estimation of steady-state concentrations based on uptake patterns over a short period.

3) Specific effects of contaminants on the biota of Lake Erie

In the following discussion of toxic effects on the Lake Erie biota the reader should be aware of the problems associated with these types of studies detailed in the previous sections. Many studies, on all trophic levels, have either contradictory results or indicate of effects at orders of magnitude differences in concentration.. This section will focus on the typical biota of large bodies of water for the bacteria, algae, and zooplankton, and on the fish of Lake Erie in particular. Appendix 5 presents an account of the toxicity of the contaminants studied in this report with an emphasis on the Lake Erie biota. Because very few studies have been performed specifically on organisms from Lake Erie, studies using the same species from different lake systems have been included in Appendix 5 as well. Even with these inclusions, there remains a paucity of data on the species of most economic importance, the yellow perch. While data on trout are abundant, they have very little economic impact on the Lake Erie fishery.

Studies of contaminant toxicity of heavy metals to bacteria present contradictory information. Bauer et al. (1981), measuring the bacterial decrease of dissolved oxygen, concluded that cadmium was the most toxic of the metals tested, followed by nickel and copper. At low concentrations copper was even found to be stimulatory. Seyfried (1978), however, found that copper and nickel were amongst the most toxic heavy metals and cadmium the least. Seyfried used cultures isolated from areas of varying heavy metal pollution. Cultures resistant to cadmium, arsenic, and lead were isolated most frequently, whereas less than half were mercury resistant. Of those with mercury resistance two-thirds were from the sediment with those isolates from mercury polluted areas having the highest percentage that were resistant.

Toxicity tests on algae present a variety of results using a variety of methods and species. Again, it must be emphasized that comparisons between procedures, and species, may be tenuous. In a study employing Euglena and Chlamydomonas species found at a local pond, but obtained from a culture collection, it was found that Euglena was more resistant than Chlamydomonas to cadmium while diatoms were unaffected at 1000 ug Cd/l (Fennikoh et al., 1978). Different forms of cadmium were tested for their toxicity to five species of Chlorophyta, all laboratory stocks, by Wong et al. (1979). It was concluded that Scenedesmus was the most resistant species and that two Chlorella species were quite different in their sensitivity. Cadmium toxicity values range from 10 ppb (Conway, 1978) to 10 ppm (Sparling, 1968) with the differences associated with variations in test conditions, species, and method of bioassay (Wong et al., 1979). Nickel is known to inhibit algal growth (Hutchinson and Stokes, 1975; Fezy et al., 1979) at concentrations as low as 1.7 umol Ni/l when water chemistry favors the presence of free Ni^{2+}, its most toxic species (Spencer and Greene, 1981). The free metal ions are most toxic species of copper (Cu^{2+} - Anderson and Morel, 1978; Petersen, 1982), cadmium (Cd^{2+} - Sunda et al., 1978, and zinc (ZN^{2+} - Petersen, 1982). In the presence of Ni^{2+} diatoms decrease in abundance while filamentous greens and blue-greens algae increase in abundance and diversity (Spencer and Greene, 1981). Blue-green algae are more tolerant perhaps due to the production of extracellular organics which can detoxify nickel, a process previously identified as used by Aphanizomenon flos-aquae to complex copper (McKnight and Morel, 1979). Mangi et al. (1978) cultured algae from natural collections to study the effects of Cr(6+). While inhibition occurred at 10 ppm, some cells were apparently healthy after 2 weeks of exposure. The uptake of chromium was found to vary with the chromium concentration and it was suggested that adsorption was responsible for the uptake, even on dead cells, proving to be a means of detoxifying the cultures. The effects of zinc have been studied on the green alga Chlorella vulgaris (Rachlin and Farran, 1974; Coleman et al., 1971), Scenedesmus quadricauda (Petersen, 1982), and Pediastrum tetras (Coleman et al., 1971) as well as the euglenaphyte Euglena virdis (Coleman et al., 1971). While Coleman et al. (1971) showed that the dry weight of algae increased at concentrations up to 4.2 ppm during a 3 week test, Rachlin and Farran (1974) found that 2.4 ppm retarded growth by 50% after only 4 days. In nature the problem of zinc toxicity is complicated by possible additive or synergistic effects with other contaminants such as copper (Petersen, 1982; Anderson and Weber, 1975; Sprague and Ramsay, 1965; Brown and Dalton, 1970; Lloyd, 1961). Synergistic effects will be discussed more fully in the following part of the report.

A natural phytoplankton assemblage was subjected to various heavy metals, singly and in mixture, followed by either sediment or bog water (a natural chelator) to determine the effects on photosynthesis (Hongve et al., 1980). In this instance mercury toxicity was the greatest followed by copper, cadmium, lead and zinc, a result similar to Gächter (1976) who reversed the ranking of lead and zinc. The addition of sediment caused the greatest reduction in the toxicity of mercury, the metal most readily sorbed, and the least reduction in cadmium and zinc, the metals least readily sorbed.

PCB's have been established as toxic to a number of freshwater algae. Lichetenstein et al. (1973), Sivalingan et al. (1973) and Ewald et al. (1976) have determined that the initial perturbation is inversely related to the chlorine content of the PCB; i.e., higher chlorinated PCB's have decreasing effects. Though arriving at a similar conclusion using Chlorella pyrenoidosa, Hawes et al. (1976a) found that PCB's exerted temporary depressing effects and only at concentrations far higher than are likely to occur in nature. It has also been estalished that sublethal effects are directly related with chlorine content whereas lethal effects are inversly related (Peakall and Lincer, 1970; Hawes et al., 1976b). Hawes et al. (1976b) warn that laboratory results may also be influenced by algal culture density and age. Using Euglena gracilis, Ewald et al. (1976) found that there was no inhibition of population growth at 100 ppm and could not get enough AROCLOR 1242 into solution to establish an LD_{50}. There was a decrease in carbon fixation and chlorophyll content with AROCLOR 1221 though no difference for oxygen consumption was observed. They suggested that population growth inhibition was the result of depressed photosynthesis or chlorophyll production perhaps due to an alteration of cell wall permeability as PCB's have been shown to bind to membranes and alter membrane organization at these binding sites (Roubal, 1974).

A number of oils have been found to have negative effects on algae. Euglena gracilis is tolerant in solutions up to 10% lubricating or diesel oil and is found at oil polluted sites while Scenedesmus quadricauda exhibits reduced growth and population size at similar oil concentrations and is found primarily at unpolluted sites (Dennington et al., 1975). A one minute immersion in oil of their substrate was found to reduce the population of red algae and increase the green algae in a study of a stream ecosystem (Lock et al., 1981). However, the response of the algae was highly variable and it was concluded that the results might be dependent upon the species or even strains present, a conclusion supported by others (Kauss et al., 1973; Pulich et al., 1974; Kauss and Hutchinson, 1974; Parsons et al., 1976). In general, no great shift in community stucture was found. Gaur and Kumar (1981) found that all test oils exerted detrimental effects on algal growth, a result similar to Soto et al. (1975), Pulich et al. (1974), and Hsiao (1978) though stimulation of algal growth by a surface oil scum has been observed by Gordon and Prouse (1973), Dunstan et al. (1975), and Parsons et al. (1976). The inhibition of growth is thought to be due to a number of factors including alteration of cell membranes, inhibition of enzyme systems, photosynthesis, nucleic acid synthesis, or other metabolic processes (Baker, 1970; O'Brien and Dixon, 1973; Karydis, 1979).

Daphnia are the most frequenty used zooplankton species for toxicity tests. In a study of chronic cadmium stress of D. galeata mendotate it was found that the test populations exhibited increased turnover rates, temporal variability, probability of extinction, and proportion of ovigerous females (Marshall, 1978). Individuals had increased prenatal mortality, average weight, and brood size. It was concluded, however, that barring synergistic interactions with other factors in a lake, 0.15 ug Cd/l would probably not have a detectable effect on the population dynamics of this species. An interesting observation was that since cadmium stress favors increased size,

it may have a synergistic relationship with fish predation on larger zooplankton (Hall et al., 1976). Bertram and Hart (1979) found that cadmium did not affect time to maturity or frequency of broods in Daphnia pulex but did affect the percent of adults producing young, the number of broods per adult, the number of young per brood, total reproduction per female, the rate of population growth, and the generation time. Cadmium in solution or adsorbed onto Chlorella had the same effect. Kettle et al. (1980) found that Daphnia pulex increased their oxygen consumption in the presence of cadmium while another caldoceran, Simocephalus serrulatus, decreased oxygen consumption, again illustrating the difficulty of the 'representative' species. Daphnia pulex had increased longevity and fecundity in the laboratory portion of this study but did not show a response under field conditions. Simocephalus serrulatus was much more sensitive with no reproduction and limited life span both in the lab and field. Winner and Farrell (1976) concluded that larger species of Daphnia (magna and pulex) are significantly less sensitive to cadmium stress than smaller species (parvula and ambigua) though the four species did not differ in their susceptibility to chronic stress. As another caveat, Canton and Adema (1978) found no difference between three species of Daphnia in their toxicity tests but did find that the results of experiments run in two different laboratories were more divergent than the results of multiple replicates in one laboratory.

Temperature may play an important role in determining the toxicity of cadmium and lead to Daphnia and the copepods Cyclops bicuspidatus thomasi and Diaptomus sicilis (Wilson, 1980). Copepods have the additional complication of having differential sensitivity between sexes. Though Wilson (1980) found copepods to be more sensitive to heavy metals than Daphnia, other studies have shown the opposite to be the case, at least for cadmium (Marshall, 1979; Marshall and Melinger, 1978) and chromium (Baudouin and Scoppa, 1974), an element copepods do not seem to absorb (Baudouin et al., 1972). Another factor to be considered in field studies is the time of the year as Borgmann et al. (1980) found that copepods showed a cycle of sensitivity being most sensitive in late fall and winter and least sensitive in early summer, to cadmium, copper, mercury, and lead. It was also found that extrapolation of lab data to the field may be difficult due to the number of factors effecting toxicity in nature.

PCB's are relatively toxic to Daphnia pulex with concentrations as low as 0.02 ppm toxic to juveniles (Morgan, 1972). It was shown that PCB adsorbed to algae caused mortality to D. pulex in only 4 days indicating that PCB's may be passed along the food chain. Oil and water soluble fractions of such hydrocarbons as #2 fuel oil and coal tar creosote greatly depress growth and reproduction in D. pulex as well (Geiger, 1979; Geiger et al., 1980; Geiger and Buikema, 1981). Changes in filtering rate may be a sensitive indicator of sublethal stress by these contaminants (Geiger and Buikema, 1981). Wong et al. (1981) examined the effects of small oil particles (of phytoplankton size) on D. pulex. They found that the toxic effect increased with concentration, and that weathered oil, which had lost most of its volatile and toxic components, was less toxic than fresh oil.

The benthic macroinvertebrates include animals from at least four phyla (Platyhelminthes, Mollusca, Annelida, and Arthropoda) and so a variety of results have been observed. Whitley (1967) found that Tubifex tubifex and Limnodrilus hoffmeisteri had a high tolerance to lead and zinc. Nickel was toxic but did not seem to impair respiration. Methylmercury was suggested to be a neurotoxin of Dugesia dorotocephala by Best et al. (1981) who observed a non-lethal response of head resorption in this flatworm. Rehwoldt et al. (1973) concluded that mercury is more toxic to the benthos than to fish but that benthic invertebrates tend to be able to withstand more different heavy metal inputs than fish. The relative resistance of the benthos has also been documented by Warnick and Bell (1969), Arthur and Leonard (1970), Nehring (1976), and Spehar et al. (1978). As with Daphnia, the juvenile stages tend to be the most sensitive (Wier and Walters, 1976).

Toxicity studies on fish are very numerous and have been discussed in great detail by Spehar et al. (1982). This is both extensive and detailed so that an adequate review has not been attempted here. One very important point that has been made in a number of studies that may have a direct bearing on dredging decisions is the general agreement that most fish larvae are more sensitive to contaminants than embryos (Eaton et al., 1978) though some species, such as the fathead minnow, have more sensitive embryos (Pickering and Gast, 1972). Eaton et al. (1978) suggest that a 60 day exposure period is required for determining the larval sensitivity to cadmium. Dredging decisions may thus required that the breeding season as well as the season during which larval fish may be abundant be taken into consideration.

4) Synergistic interactions among contaminants.

In most natural waters a number of contaminants are present together in solution. When mixed together such combinations have often been determined to exhibit synergistic effects. Such interactions have been investigated for more than 30 years, primarily using algae as the test organisms, though protozoa, copepods, and fish have been tested. The conclusions from a number of these studies are presented in Table 16. Synergistic and additive effects have not been distinguished in this table. While it is recognized that these are separable interactions the results to the test organisms are the same. Separation of these effects is, however, important to management decisions (Anderson and Weber, 1975). In general, many metals have synergistic relations while in a few instances they are antagonistic. In no instance has synergism between PCB's and heavy metals been investigated.

An example of a typical laboratory study is that on Selanastrum capricornutum (Bartlett et al., 1974), where it was determined that combinations of copper, zinc, and cadmium, were similar in toxicity to equal concentrations of zinc. In addition, copper and cadmium mixtures resulted in greater growth than equal concentrations of copper indicating an antagonistic relationship. An important methodological observation was that the nature of the media was very important in determining the outcome of experiments with more alkaline media resulting in the metals having increased toxicity, a result confirmed by Anderson and Weber (1975).

In a study of a natural, mixed phytoplankton assemblage, it was found that the results were variable depending upon the dominant species in the sample (Pietilainen, 1975). This result suggests that certain species have more, or less, sensitivity to certain metals or metal combinations, a result supported by Wong et al. (1978). An interesting observation was that when lead was added to a solution with a higher concentration of cadmium there was a synergistic effect whereas the addition of cadmium to a solution with a higher concentration of lead led to an antagonistic effect. Such cross mixing is seldom done so that important interactions may be overlooked. While the approach of using natural assemblages for this type of testing, and toxicity tests in general, is attractive, Wong and Beaver (1980) have pointed out that analyses of the results of such studies requires extra caution due to algal species interactions such as competition for nutrients and the production of suppresive extracellular products.

Anderson and Weber (1975) provide a thorough analysis of how metals interact and the differences between additive and synergistic effects among heavy metals. They conclude from their study on guppies (Poecilia reticulata) that the toxicity of mixtures of metals can be predicted from the lethal response curve for each constituent. This is apparently the case for copepods as well (Borgmann, 1980). It was found in this instance that for a natural assemblage of cyclopoid and calanoid copepods collected from the Burlington Canal, that most of the toxicity was accounted for by the individual toxicites of the metals, as when present singly. The accuracy of this determination decreases as the number of metals tested together increases due to the accumulation of small synergistic effects. However, metals are probably most often found in combinations of greater than five. A combination of the 10 heavy metals for which there are guidelines established by IJC were toxic when together at their suggested maxima (Wong et al., 1978) to algae.

A recent study that has particular bearing on these types of studies is that of Foster (1982a; b) who has found that algae of the same species collected at sites with varying heavy metal concentrations had corresponding sensitivities to these metals. Not only were the algae found to be more tolerant in contaminated areas but the tolerance was found to be genetically based. The implication is that laboratory studies using algae from culture collections may not reflect events in nature whereby natural selection has resulted in algal populations of greater contaminant tolerance. While the use of natural assemblages may yield results of mixed value, algal clones derived from the aquatic habitat in question would seem desirable.

VII. The impact of dredging on the Lake Erie ecosystem.

Having reviewed the current status of opinion on the physical, chemical, and biological factors that may effect Lake Erie in terms of contaminant availability and dredging activities, the final and most pertinent question is that of the actual impact of dredging on the Lake Erie ecosystem. To what factors are the impacts, if any, due? Studies have been conducted to determine the role of heavy metals and the physical impact of sedimentation, but unfortunately, with only minor exception, not simultaneously. As an example, dredging in the relatively unpolluted San Antonio Bay, Texas, was found to have no significant impact on the heavy metal concentration in oysters or clams with the exception of copper (Sims and Presley, 1976) while benthic crustaceans were unaffected. Fish showed a high degree of overlap of contaminant levels both between species and on dredged and undredged sites. The conclusion then was that dredging in a clean harbor had no effect on the benthic community in terms of heavy metal accumulation. However, the impact of the sediment on the disposal site was not investigated.

Table 16. Synergistic and additive effects between heavy metals tested in pairs. + = synergistic or additive effect; - = antagonistic effect. Areas that are not marked indicated that the pair has not been tested. In some instances both effects have been noted depending upon the organism tested and the specific test conditions.

	As	Cd	Cr	Cu	Hg	Ni	Pb	Zn
As								
Cd				-			+	+
Cr								
Cu	+	-(+)			+	+	+	+(-)
Hg								
Ni								
Pb	+	-			+			
Zn	+	+(-)		+(-)	+(-)		+(-)	

References: Bartlett et al., 1974; Wissmar, 1972; Pietlainen, 1975; Hutchinson and Czyrska, 1972; Haberer and Norrmann, 1971; Gray and Ventilla, 1973; Liebmann, 1960; Laborey and Lavollay, 1967; Cheremisinoff and Habib, 1973; Miettinen, 1975; Borgmann, 1980; Sprague, 1964; Lloyd, 1961; Brandt, 1946; Doudoroff, 1952; Wong et al., 1978.

Another marine study, of Coos Bay, Oregon, investigated the physical impact of dredging but did not examine the role of contaminant release (McCauley et al., 1977). It was their conclusion that at the dredge site there was a rapid (one week) adjustment in the number of organisms (also noted by Harrison et al., 1964; Flemer et al., 1968) but that this did not necessarily indicate a long term re-establishment. An important observation was the wide degree of difference both within and between the control and study sites, the implication being that the benthos is a highly variable

habitat requiring a careful sampling scheme for accurate population estimates. This was also noted by Sweeney (1978) in Lake Erie. It was concluded that dredging caused:

1) Significant siltation in adjacent areas near the dredge site due to prop wash by the dredge and the inefficiency of the dredge to remove all suspended particles;
2) Removal of the benthic organisms by the dredge; and
3) Alternation of current patterns and sediment distribution.

Regarding the effects at the disposal site they concluded:

1) The population covered by disposed material recovered after 7 days;
2) There was a decline up to 50 m from the disposal site of the benthos due to siltation. This will vary with the specific dredge operation depending upon site depth, currents, and the type of dredge material; and
3) Most individuals of the benthic community were not grossly impaired.

Again, the effect of pollutants that may have been released by the dredge operation were not monitored in this study so there is no way to determine if this was a possible cause of any of the observed effects.

In a study of a Lake Erie disposal site out of use for 5 years it was found that the disposal site remained more polluted than the control sites and still exhibited "lower quality" than the control sites (Sweeney et al., 1975). It was found that heavy metals, with the exception of cadmium, and oil and grease, were in higher concentration at the disposal site. This would seem to be good evidence for the ability of the sediments to retain contaminants out of the water column. The benthic community remained disturbed with a greater evenness in species diversity occurring at the control sites, though the species composition at both sites were indicative of a sediment containing a high degree of organic matter. There was, however, no actual difference in the calculated species diversity index. It was concluded that a prohibition should be placed on open lake disposal for dredge material from polluted sources, specifically the Cuyahoga River and Cleveland harbor.

A more comprehensive survey conducted at the Ashtabula River disposal site (Flint, 1979) similarly found that the control sites were more diverse and their population densities more evenly distributed among the species (i.e., no dominant species). It was concluded that the benthic community at the disposal site had substantially changed after deposition representing a less stable community structure exhibiting larger population fluctuations than at the control sites. The disposal site changes were related to substantial increases in individuals of the oligochaete genus Limnodrilus (especially L. udekeminus and L. hoffmeisteri) and a decrease in individuals in species of the genus Aulodrilus. It was also found that river and harbor dredge material mediated different responses at the disposal site, perhaps due to differences in the sediment texture. With both types of dredge material there was an increase in the number of oligochaetes, particularly those species considered most opportunistic (such as Peloscolex multisetosus multisetosus). The predominance of oligochaetes at disposal sites has also been reported by Kinney (1972) in Lake Ontario.

Another study of the Ashtabula River disposal site is the highly detailed report by Sweeney (1978). This study monitored the requisite physical, chemical, and biological factors and concluded that though there was an immediate impact on the benthos, there were no long term deleterious effects on the ecosystem resulting from either the physical impact of sediment or the release of contaminants from the dredge material. Factors such as low oxygen concentration, low interstitial water content, increased compaction, and highly reducing conditions were all suggested as causes for the change in the benthic community. The warning is given that disposal operations should be co-ordinated in such a way as to minimize the impact on the life cycles and interactions of many of these organisms. Otherwise, dredging was concluded to not be an immediate danger to the Lake Erie ecosystem.

Certain data are absent from many of these studies which weaken their conclusions. First, change in biomass of the benthos caused by the disposal of dredge material is not documented. Changes in population sizes do not reflect changes in biomass, which is an important indicator of productivity. There is no question that there is a shift in species composition associated with dredge disposal but there is no indication that there is a loss of productivity. Second, these studies, with the exception of Sweeney (1978), did not differentiate between changes due to burial by sediments and the change in availability of contaminants. Because the two effects have not been partitioned no decision can be made on the specific effect of the disposal of polluted dredge material. Third, and perhaps most critical, many of these studies make conclusions which are not based on ecological concepts regarding community structure. A change in species distribution or population structure over a relatively small area is not necessarily a damning flaw. There are no guidelines for "quality of benthos" indicating desirable species, community structure, or productivity. There are no data regarding the impact to the rest of the ecosystem resulting from a change in the species composition of the benthos. This is not to say that there are no deleterious effects, just that they have not been critically examined. Until this is done it is difficult to assess the impact of dredging on the benthic ecosystem or the rest of the Lake Erie ecosystem.

While studies have been conducted which detailed alterations to the benthic community there are very few studies which show the direct effects of dredging on other components of the ecosystem. In one study of importance it was found that fish can definitely accumulate PCB's, zinc and mercury from non-aerated (reduced) sediments and arsenic and chromium from aerated (oxidized) sediments (Seeyle et al., 1982). This is the sole paper indicating that the resuspension of sediments, such as from dredging or from wind induced mixing, may have a critical impact on other parts of the ecosystem. However, this was a laboratory study with the fish forced to stay in contact with the suspended sediment. There is, again, very little information regarding the specific impact of dredging on higher trophic levels either directly or indirectly through the food web.

The sediments of Lake Erie may hold potentially toxic levels of many contaminants, with many of the more often dredged areas, particularly harbors, the most contaminated. Before dredging disposal decisions can be made the

safety of the ecosystem must be ensured with carefully designed and monitored pilot studies which will determine if there are deleterious effects to the entire ecosystem including changes in productivity and the release of harmful contaminants to the higher trophic levels. Without such detailed studies assessment of the impact of dredging, at this point, would have to agree with that of Sly (1977) and Sweeney (1978) that dredging plays a minimal role in the dynamics of the Lake Erie ecosystem.

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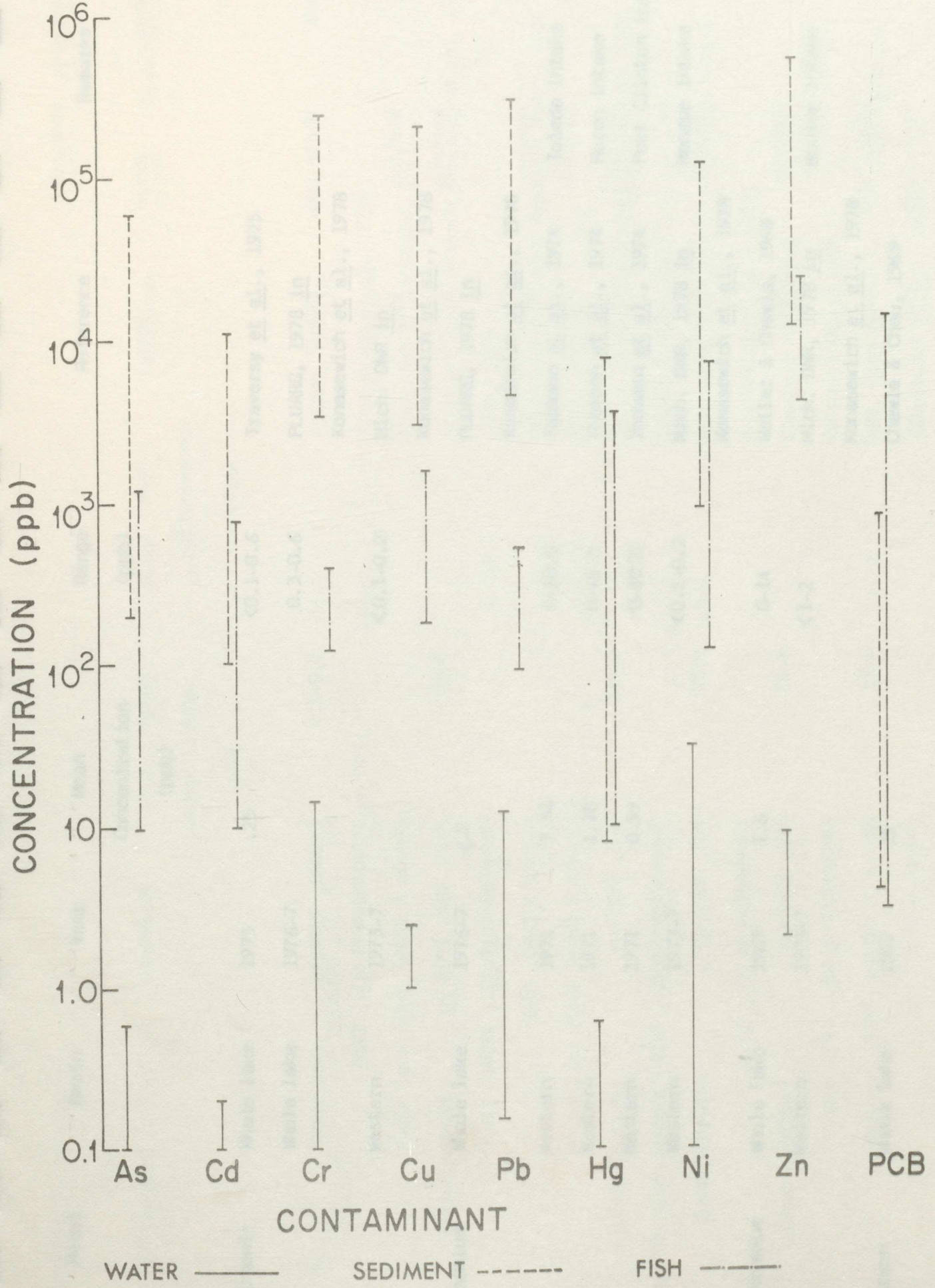
Appendix 1

a. Summary figure of contaminants in the Lake Erie system in parts per billion (ppb).

b. Concentrations of contaminants in the water of Lake Erie in parts per billion (ppb).


CONCENTRATION (ppb)
10^6
10^5
10^4
10^3
10^2
10
1.0
0.1
As
Cd
Cr
Cu
Pb
Hg
Ni
Zn
PCB
CONTAMINANT
WATER
SEDIMENT
FISH

Metal	Basin	Year	Mean Concentration (ppb)	Range (ppb)	Reference	Remarks
Arsenic	Whole lake	1975	.25	<0.1-0.6	Traversy et al., 1975	
	Whole lake	1976-7		0.3-0.6	PLUARG, 1978 in Konasewich et al., 1978	
	Western	1973-7		<0.1-0.2	Mich. DNR in Konasewich et al., 1978	
Cadmium	Whole lake	1976-7	<.2		PLUARG, 1978 in Konasewich et al., 1978	
	Western	1971	1.76	0-10.0	Thomann et al., 1974	Toledo intake
	Western	1971	1.18	0-10.0	Thomann et al., 1974	Huron intake
	Western	1971	0.59	0-10.0	Thomann et al., 1974	Port Clinton intake
	Western	1973-7		<0.1-0.2	Mich. DNR, 1978 in Konasewich et al., 1978	Monroe intake
Chromium	Whole lake	1967	1.6	0-14	Weiler & Chawla, 1968	
	Western	1973-7		<1-2	Mich. DNR, 1978 in Konasewich et al., 1978	Monroe intake
Copper	Whole lake	1967	15		Chawla & Chau, 1969	

Appendix 1. Continued

Metal	Basin	Year	Mean Concentration (ppb)	Range (ppb)	Reference	Remarks
Copper	Whole lake	1976-7		1.0-2.5	PLUARG, 1978 in Konasewich et al., 1978	
	Western	1967	17		Chawla & Chau, 1969	
	Western	1973-7		3-45	Mich. DNR, 1978 in Konasewich et al., 1978	Monroe intake
	Central	1967	15		Chawla & Chau, 1969	
	Eastern	1967	14		Chawla & Chau, 1969	
	Eastern	1973		11-17	MOE, 1973 in Konasewich et al., 1978	Port Colborne, Algoma-Inco effluent
	Eastern	1973		4-20	MOE, 1973 in Konasewich et al., 1978	Nickel Beach
	Eastern	1973		6-17	MOE, 1973 in Konasewich et al., 1978	Welland Canal
Lead	Whole lake	1967	2.8	1-12	Weiler & Chawla, 1968	
	Whole lake	1967	4		Chawla & Chau, 1969	

Appendix 1. Continued

Metal	Basin	Year	Mean Concentration (ppb)	Range (ppb)	Reference	Remarks
Lead	Eastern	1973	6		MOE, 1973 in Konasewich et al., 1978	Welland Canal
Mercury	Whole lake	1970-1	.17±.11	0-0.4	Chau & Saitoh, 1973	
	Whole lake	1974	.05	<.05	CCIW, 1974 (NAQUADAT) in Sherbin, 1979	
	Whole lake	1976-7		⩽0.5	PLUARG, 1978 in Konasewich et al., 1978	
	Whole lake	1978	0.61		IJC, 1978	
	Western	1973-7		<0.1-0.8	Mich. DNR in Konasewich et al., 1978	Monroe intake
Nickel	Whole lake	1967	3		Chawla & Chau, 1969	
	Western	1967	4		Chawla & Chau, 1969	
	Western	1973-7		<5-14	Mich. DNR, 1978 in Konasewich et al., 1978	Monroe intake
	Central	1967	2		Chawla & Chau, 1969	

Appendix 1. Continued

Metal	Basin	Year	Mean Concentration (ppb)	Range (ppb)	Reference	Remarks
Nickel	Eastern	1967	2		Chawla & Chau, 1969	
	Eastern	1973		140-320	MOE, 1973 in Konasewich et al. 1978	Port Colborne, Algoma-Inco effluent
	Eastern	1973		4-840	MOE, 1973 in Konasewich et al., 1978	Nickel Beach
	Eastern	1973		5-10	MOE, 1973 in Konasewich et al., 1978	Welland Canal
Zinc	Whole lake	1967	8		Chawla & Chau, 1969	
	Whole lake	1976-7		2-9	PLUARG, 1978 in Konasewich et al., 1978	
	Western	1967	7		Chawla & Chau, 1969	
	Western	1973-7		11-24	Mich. DNR in Konasewich et al., 1978	Monroe intake
	Central	1967	8		Chawla & Chau, 1969	
	Eastern	1967	7		Chawla & Chau, 1969	

Appendix 2

Concentrations of contaminants
in the sediments of Lake Erie
in parts per million (ppm).

Concentration of PCB's in ppb

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Arsenic	Whole lake	1971	1.3±4.3	0.2-60	Thomas & Mudroch, 1979	
	Whole lake	1975	3.20	2.0-5.5	Traversy et al., 1975	
	Western	1970	7.9±2.5	4.0-12.3	MOE, 1981	
	Western	1971	1.0±1.4	0.2-6.5	Thomas & Mudroch, 1979	
	Western	1973	<2.4±1.1		Mich. DNR in Konasewich et al., 1978	Michigan nearshore north
	Western	1973	<2.5±1.1		Mich. DNR in Konasewich et al. 1978	Michigan nearshore south
	Western	1974	<2.0		Burge, 1974 unpublished in Konasewich et al., 1978	Port Clinton
	Western	1974	1.2		Walters et al., 1974	
	Western	1975		3.0-4.0	Traversy et al., 1975	
	Western	1979	6.5±3.0	3.2-12.0	MOE, 1981	
	Central	1971	1.2±1.7	0.2-7.0	Thomas & Mudroch, 1979	
	Central	1974	1.6		Walters et al., 1974	
	Central	1974		7.0-25	Burge, 1974 in Konasewich et al., 1978	Ashtabula Harbor

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Arsenic	Central	1974		7.0-15	Burge & Elly, 1974 in Konasewich et al., 1978	Conneaut Harbor
	Central	1975	3.4	2.0-5.5	Traversy et al., 1975	
	Central	1975		11-16	EPA, 1975 in Konasewich et al., 1978	Ashtabula Harbor
	Central	1975		6-16	EPA, 1975 in Konasewich et al., 1978	Fairport Harbor
	Eastern	1971	0.4±0.6	0.2-3.0	Thomas & Mudroch, 1979	
	Eastern	1974	0.45		Walters et al., 1974	
	Eastern	1975	2.7	2.0-4.0	Traversy et al., 1975	
Cadmium	Whole lake	1971	2.4±1.5	0.1-10.8	Thomas & Mudroch, 1979	
	Western	1970	5.6±3.5	2.2-13.7	MOE, 1981	
	Western	1971	4.2±1.7	1.4-10.8	Thomas & Mudroch, 1979	
	Western	1973	4.6±1.4		Mich. DNR in Konasewich et al., 1978	north region

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Cadmium	Western	1973	1.6±0.6		Mich. DNR in Konasewich et al., 1978	south region
	Western	1974	3.5		Burge, 1974 in Konasewich et al., 1978	
	Western	1979	3.0±2.4	0.8-6.0	MOE, 1981	
	Central	1971	2.8±1.1	1.0-6.3	Thomas & Mudroch, 1979	
	Central	1974		4.0-12.0	Burge, 1974 in Konasewich et al., 1978	Ashtabula Harbor
	Central	1974		1.9-7.4	Burge, 1974 in Konasewich et al., 1978	Conneaut Harbor
	Central	1975	1.0		EPA, 1975 in Konasewich et al., 1978	Ashtabula Harbor
	Central	1975		1.0-2.5	EPA, 1975 in Konasewich et al., 1978	Fairport Harbor
	Eastern	1971	2.3±0.8	1.1-3.7	Thomas & Mudroch, 1979	
Chromium	Whole lake	1971	51±34	6-238	Thomas & Mudroch, 1979	

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Chromium	Whole lake	1972-3	29.5	3.4-156.8	Hutchinson & Fitchko, 1974	
	Western	1970	177±118	50-362	MOE, 1981	
	Western	1971	96±43	23-238	Thomas & Mudroch, 1979	
	Western	1972	37		Walters et al., 1974	
	Western	1973	49±14		Mich. DNR in Konasewich et al., 1978	nearshore north region
	Western	1973	14±3		Mich. DNR in Konasewich et al., 1978	nearshore south region
	Western	1974	6		Burge, 1974 in Konasewich et al., 1978	Port Clinton Harbor
	Western	1979	69 ± 49	15-150	MOE, 1981	
	Central	1972	60		Walters et al., 1974	
	Central	1971	58 ± 20	21-107	Thomas & Mudroch, 1979	
	Central	1974		42-2100	Burge, 1974 in Konasewich et al., 1978	Ashtabula Harbor
	Central	1974		23-49	Burge, 1974 in Konasewich et al., 1978	Conneaut Harbor

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Chromium	Central	1974		44-102	Burge, 1974 in Konasewich et al., 1978	Fairport Harbor
	Central	1975		22-82	EPA, 1975 in Konasewich et al., 1978	Ashtabula Harbor
	Central	1975		55-130	EPA, 1975 in Konasewich et al., 1978	Fairport Harbor
	Eastern	1971	48 ± 21	14-79	Thomas & Mudroch, 1979	
	Eastern	1972	12		Walters et al., 1974	
Copper	Whole lake	1971	39 ± 27	3-207	Thomas & Mudroch, 1979	
	Western	1970	79 ± 46	30-183	MOE, 1981	
	Western	1971	69 ± 25	26-162	Thomas & Mudroch, 1979	
	Western	1972	31		Walters et al., 1974	
	Western	1973	50 ± 10		Mich. DNR in Konasewich et al., 1978	nearshore north region
	Western	1973	19 ± 11		Mich. DNR in Konasewich et al., 1978	nearshore south region

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Copper	Western	1974		<3	Burge, 1974 in Konasewich et al., 1978	Port Clinton
	Western	1979	51 ± 30	10-97	MOE, 1981	
	Central	1971	52 ± 22	18-207	Thomas & Mudroch, 1979	
	Central	1972	40		Walters et al., 1974	
	Central	1974		15-54	Burge & Elly, 1974 in Konasewich et al., 1978	Conneaut Harbor
	Central	1974		16-30	Burge, 1974 in Konasewich et al., 1978	Ashtabula Harbor
	Central	1974?		30.4-44.8	MOE in Konasewich et al., 1978	Port Stanley
	Central	1974?		14.6-19.6	MOE in Konasewich et al., 1978	Port Burwell
	Central	1975		26-48	EPA in Konasewich et al., 1978	Ashtabula Harbor
	Central	1975		26-46	EPA in Konasewich et al., 1978	Fairport Harbor

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Copper	Eastern	1971	34 ± 13	5-57	Thomas & Mudroch, 1979	
	Eastern	1972	12		Walters et al., 1974	
	Eastern	1973		245-351	MOE, 1973 in Konasewich et al., 1978	Port Colborne, Inco effluent
	Eastern	1973		7.7-35.6	MOE, 1973 in Konasewich et al., 1978	Port Colborne, Nickel Beach
	Eastern	1973		35.8-44.5	MOE, 1973 in Konasewich et al., 1978	Port Colborne, Welland Canal
Mercury	Whole lake	1971	0.582 ± 0.555	0.008-2.929	Thomas & Jaquet, 1976	
	Whole lake	1971	0.578 ± 0.554	0.008-2.929	Thomas & Mudroch, 1979	
	Whole lake	1971	0.609 ± 0.703	0.013-7.488	Thomas, 1974	
	Western	1967	0.72		Skoch & Turk, 1972	island area
	Western	1968	0.89		Skoch & Turk, 1972	island area
	Western	1970	1.14 ± 1.28	0.05-3.6	MOE, 1981	
	Western	1970	1.27 ± 1.23	0.05-4.6	Kinkead & Hamdy, 1978	
	Western	1970	<0.05 (dry wt)	0.8-2.1(wet wt)	Fed. Water Qual. Admin., 1970 in Konasewich et al., 1978	

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Mercury	Western	1971		<1.0-4.2	Walters et al., 1972	
	Western	1971		0.1-4.8	Kovacik & Walters, 1973	
	Western	1971	1.217 ± 0.792	0.065-2.929	Thomas & Jaquet, 1976	
	Western	1971	1.622 ± 0.694	0.484-2.929	Thomas & Mudroch, 1979	
	Western	1972	0.44		Walters et al., 1974	
	Western	1974	0.2		Burge, 1974 in Konasewich et al., 1978	Port Clinton
	Western	1976	0.94 ± 0.54	0.02-2.3	Kinkead & Hamdy, 1978	
	Western	1979	0.60 ± 0.42	0.10-1.3	MOE, 1981	
	Central	1971	0.544 ± 0.191	0.056-1.03	Thomas & Jaquet, 1976	
	Central	1971	0.544 ± 0.191	0.156-1.03	Thomas & Mudroch, 1979	
	Central	1972	0.33		Walters et al., 1974	
	Central	1974		<0.2-0.7	Burge & Elly, 1974 in Konasewich et al., 1978	Conneaut Harbor
	Central	1974		<0.2-4.6	Burge, 1974 in Konasewich et al., 1978	Ashtabula Harbor

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Mercury	Central	1974		<0.4-0.6	Burge, 1974 in Konasewich et al., 1978	Fairport Harbor
	Central	1975	<0.1	<0.1	EPA, 1975 in Konasewich et al., 1978	Fairport Harbor
	Central	1975	<0.1		EPA, 1975 in Konasewich et al., 1978	Ashtabula Harbor
	Eastern	1971	0.483 ± 0.272	0.045-0.977	Thomas & Jaquet, 1976	
	Eastern	1971	0.464 ± 0.260	0.045-0.812	Thomas & Mudroch, 1979	
	Eastern	1972	0.07		Walters et al., 1974	
Lead	Whole lake	1971	87 ± 50	9-299	PLUARG, 1977 in Konasewich et al., 1978	
	Whole lake	1971	48 ± 30		PLUARG, 1977 in Konasewich et al., 1978	non-depositional zone
	Whole lake	1971	112 ± 44		PLUARG, 1977 in Konasewich et al., 1978	off shore stations
	Whole lake	1972-3	25.4	4.6-120	Hutchinson & Fitchko, 1974	
	Western	1970	86 ± 48	30-173	MOE, 1981	

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Lead	Western	1971	145 ± 52	69-299	PLUARG, 1977 in Konasewich et al., 1978	
	Western	1973	61.6 ± 14.7		Mich. DNR in Konasewich et al., 1978	nearshore north region
	Western	1973	17.2 ± 5.5		Mich. DNR in Konasewich et al., 1978	nearshore south region
	Western	1974		30-173	MOE in Konasewich et al., 1978	
	Western	1974	<10		Burge, 1974 in Konasewich et al., 1978	Port Clinton Harbor
	Western	1974		8.8-12.2	MOE in Konasewich et al., 1978	Leamington
	Western	1979	56 ± 36	6-110	MOE, 1981	
	Central	1971	111 ± 34	43-194	PLUARG, 1977 in Konasewich et al., 1978	
	Central	1974		10-20	Burge, 1974 in Konasewich et al., 1978	Ashtabula Harbor

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Lead	Central	1974		10-50	Burge, 1974 in Konasewich et al., 1978	Conneaut Harbor
	Central	1974		20-50	Burge, 1974 in Konasewich et al., 1978	Fairport Harbor
	Central	1974		34.1-82.6	MOE in Konasewich et al., 1978	Port Stanley
	Central	1974		18.8-29.9	MOE in Konasewich et al., 1978	Port Burwell
	Central	1975		12-36	EPA, 1975 in Konasewich et al., 1978	Ashtabula Harbor
	Central	1975		13-57	EPA, 1975 in Konasewich et al., 1978	Fairport Harbor
	Eastern	1971	81 ± 35	18-128	PLUARG, 1977 in Konasewich et al., 1978	
	Eastern	1973		305-398	MOE, 1973 in Konasewich et al., 1978	Port Colborne

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Lead	Eastern	1973		18.9-57.7	MOE, 1973 in Konasewich et al., 1978	Nickel Beach
	Eastern	1973		59.4-73.4	MOE, 1973 in Konasewich et al., 1978	Welland Canal
Nickel	Whole lake	1971	45 ± 21	9-121	Thomas & Mudroch, 1979	
	Western	1971	71 ± 20	31-121	Thomas & Mudroch, 1979	
	Western	1972	65		Walters et al., 1974	
	Western	1973	49 ± 14		Mich. DNR in Konasewich et al., 1978	nearshore north region
	Western	1973	22 ± 9		Mich. DNR in Konasewich et al., 1978	nearshore south region
	Western	1974	40		Burge, 1974 in Konasewich et al., 1978	Port Clinton
	Central	1971	54 ± 11	24-80	Thomas & Mudroch, 1979	
	Central	1972	56		Walters et al., 1974	
	Central	1974		110-130	Burge, 1974 in Konasewich et al., 1978	Ashtabula Harbor

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Nickel	Central	1974		110-270	Burge & Elly, 1974 in Konasewich et al., 1978	Conneaut Harbor
	Central	1975		19-38	EPA, 1975 in Konasewich et al., 1978	Ashtabula Harbor
	Central	1975		16-40	EPA, 1975 in Konasewich et al., 1978	Fairport Harbor
	Eastern	1971	45 ± 17	16-68	Thomas & Mudroch, 1979	
	Eastern	1972	32		Walters et al., 1974	
	Eastern	1973		4800-5630	MOE, 1973 in Konasewich et al., 1978	Port Colborne - Algoma Inco effluent
	Eastern	1973		115-796	MOE, 1973 in Konasewich et al., 1978	Nickel Beach
	Eastern	1973		97.2-257	MOE, 1973 in Konasewich et al., 1978	Welland Canal
Zinc	Whole lake	1971	166 ± 99	16-536	Thomas & Mudroch, 1979	
	Whole lake	1974		12-252.5	Hutchinson & Fitchko, 1974	
	Western	1970	224 ± 16	54-530	MOE, 1981	

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Zinc	Western	1971	246 ± 82	95-536	Thomas & Mudroch, 1979	
	Western	1972	18		Walters et al., 1974	
	Western	1973	249 ± 41		Mich. DNR in Konasewich et al., 1978	nearshore north region
	Western	1973	69 ± 13		Mich. DNR in Konasewich et al., 1978	nearshore south region
	Western	1974	10		Burge, 1974 in Konasewich et al., 1978	Port Clinton Harbor
	Western	1979	200 ± 131	45-440	MOE, 1981	
	Central	1971	224 ± 71	79-370	Thomas & Mudroch, 1979	
	Central	1972	26		Walters et al., 1974	
	Central	1974		130-150	Burge, 1974 in Konasewich et al., 1978	Ashtabula Harbor
	Central	1974		100-140	Burge, 1974 in Konasewich et al., 1978	Conneaut Harbor
	Central	1974		100-400	Burge, 1974 in Konasewich et al., 1978	Fairport Harbor

Appendix 2. Continued

Metal	Basin	Year	Mean Concentration (ppm)	Range (ppm)	Reference	Remarks
Zinc	Central	1975		106-156	EPA, 1975 in Konasewich et al., 1978	Ashtabula Harbor
	Central	1975		80-161	EPA, 1975 in Konasewich et al., 1978	Fairport Harbor
	Eastern	1971	178 ± 98	33-332	Thomas & Mudroch, 1979	
	Eastern	1972	12		Walters et al., 1974	
PCB's	Whole lake	1971	95 ± 114	4-800	Frank et al., 1977; Thomas & Mudroch, 1979	
	Whole lake	1971	64 ± 105	8-800	Frank et al., 1977	non-depositional zone
	Whole lake	1971	115 ± 114	4-660	Frank et al., 1977	depositional zone
	Whole lake	1978		74-252	PLUARG, 1978 in Kaiser, 1978	
	Western	1971	252 ± 156	4-660	Frank et al., 1977; Thomas & Mudroch, 1979	
	Central	1971	74 ± 56	12-330	Frank et al., 1977; Thomas & Mudroch, 1979	
	Eastern	1971	86 ± 85	12-320	Frank et al., 1977; Thomas & Mudroch, 1979	

Appendix 3

Bioaccumulation of contaminants

in the biota of Lake Erie

Concentrations in ppm unless

otherwise indicated

Appendix 3a. - heavy metals

3b. - PCB's

Appendix 3a.

Organism		Metals							Remarks	Reference
		As	Cd	Cr	Cu	Pb	Hg	Ni	Zn		
Micrococcus luteus	BCF whole cell					941.7				bacteria from Fort Collins Park (Co.)	Tornabene & Edwards, 1972
	[] in cells					4880				(whole cells) (dry wt.)	
	[] in controls					2149				incubated for 2 days	
	[] in media					2.9					
Azobacter sp.	BCF					2.27×10^6				bacteria from Fort Collins Park	Tornabene & Edwards, 1972
	[] in cells					3.1×10^5				whole cells-dry wt.	
	[] in control					1.48×10^4				incubated for 7 days	
	[] in media					0.13					
bacterial community	BCF ranges		673-1260							bacteria collected from River Ourthe (Liege, Belgium)	Remacle, 1981
	[] in cells		673-1260							ranges due to ranges in initial bacteria dilution	
	[] in media		1							(values for free bacteria)	
										BCF for adhering bacteria = 6100	
Pseudomonas fluorescens	BCF						896.6			Ottawa River water	Ramamoorthy et al., 1977
	[] in cells						1300			72 hr. exposure	
	[] in water (initial)						1.45			58% loss of Hg from system over 72 hr	
natural bacterial population	BCF			2	1.75		0.7		1.5	Savannah River	Guthrie et al., 1977
	[] in cells			0.4	0.7		0.02		0.6	14 day exposure	
	[] in water			0.2	0.4		0.03		0.4	(wet wt.)	
	[] in cells (Cu added)			0.3	0.3		0.02		0.3	Cu added 2.0 parts/10^6Cu added	
	[] in cells (Hg added)			0.5	0.9		0.04		0.5	Hg added 0.04 parts/10^6 Hg added incr. [] of Cu, Cr & Hg	
natural bacterial population	BCF			2.6	0.2		2		1.2	L. Houston (ash basin effluent)	Guthrie et al., 1977
	[] in cells			0.65	1.2		0.02		1.4	wet wt. - 14 day exposure	
	[] in water			0.25	5.4		0.01		1.17		
	[]in cells (Cu added)			0.75	3.2		0.01		4.0	Cu added-2 parts/10^6 Cu added - incr. [] of Cr, Cu, & Zn	
	[] in cells (Hg added)			1.0	10.0		0.03		4.0	Hg added-0.04parts/10^6 Hg added - incr.[] of all metals	

Appendix 3a. continued

Organism		Metals As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Remarks (Physical Parameters)	Reference
Pseudomanas sp.	BCF		1000-3000							bacteria isolated from Ottawa River (Ohio) -resistant to 50 ppb of Cd (dry wt. basis) 4 day exposure pH 6-7; 23 & 26°C at pH 7 & 4°C BCF = 500 (est. from graph) at pH 8.5 & 23°C BCF = 250 (est. from graph)	Titus & Pfister, 1982
	[] in cells		1000-3000								
	[] in water		1								
phytoplankton	BCF	1500		252	≈1200		5900		1690	L. Michigan field data	Copeland & Ayers, 1972 in Konasewich et al., 1978
(mixed pop.)	[] in plants	1.5		0.43	6.0		0.16		27.0		
mixed algae & macrophytes	BCF	70	14	35	18		13		12	Savannah River project 1974 (before dredging) field data	Cherry & Guthrie, 1977
	[] in plants	4.2	1.5	5.7	7.2		0.4		5.0		
	[] in water	0.06	0.12	0.16	0.39		0.03		0.39		
	[] in sediments	20	1.7	38	81		0.8		6.4		
mixed algae & macrophytes	BCF	75	10	14	35		20		15	1975 (after dredging) Savannah River project field data	Cherry & Guthrie, 1977
	[] in plants	5.3	0.9	2.9	14		0.4		51		
	[] in water	0.07	0.09	0.20	0.40		0.02		3.30		
	[] in sediments	27	3.8	34	32		0.8		7.6		
Fragilaria crotonensis	BCF		3500-8000							24 hr. laboratory experiment max. BCF of 8000 at 4.5 ug/l min. BCF of 3500 at 0.2 ug/l at ambient levels	Conway & Williams, 1979
	initial uptake		0.2-18ngCd/mm^3								
	max. [] in cells		0.3-64 ng Cd/mm^3								
	ambient []		0.05-8.5ug/l								
Fragilaria crotonensis	BCF live cells		8000							7 hr. laboratory experiment	Conway & Williams, 1979
	BCF cold-killed		9000								
	BCF Hg-killed		2200								
	ambient []		4-5ug/l								

Appendix 3a. continued

Organism		Metals								Remarks (Physical Parameters)	Reference
		As	Cd	Cr	Cu	Pb	Hg	Ni	Zn		
Phytoplankton (+ Rotifers)	BCF [] in plants [] in water					$6x10^4$-$6.95x10^5$ 12-278 0.2-2.5ppb				3 English lakes - field data ranges of values	Denny & Welsh, 1979
Asterionella formosa	BCF initial uptake		20,300-24,400 2.5-17.0ng/mm^3							24 hr. laboratory experiment max BCF of 24,400 at 4.1ug/l (ambient level) min BCF of 20,300 at 8.8ug/l (ambient level)	Conway & Williams, 1979
	max. [] in cells		43-179 ng/mm^3								
	ambient []		1.9-8.8ug/l								
Asterionella formosa	BCF live cells BCF cold-killed BCF Hg-killed ambient []		24,400 11,000 2100 4-5ug/l							72 hr. experiment	Conway & Williams, 1979
Chlamydomonas sp.	BCF BCF		5480			18600 360	6780			11 day old-culture } 3 hr 44 day old-culture } laboratory exposure	Hassett et al., 1980
Scenedesmus obliquus	BCF BCF		2700 105			not sig	6540 319			11 day old-culture } 3 hr 44 day old-culture } laboratory exposure	Hassett et al., 1980
Chlorella pyrenoidosa	BCF		16,700-32,000			not sig	24,000-47,300			laboratory experiment 11 day old cultures 3 hrs exposure, varying pH BCF max. occur at pH 7 or 8 BCF min. occur at pH 5 or 6	Hassett et al., 1980
Scenedesmus obliquus	BCF [] in water		2700-3600 .366ppb			not sig 0.025ppb	3540-6540 0.041ppb				Hassett et al., 1980

Appendix 3a. continued

Organism		Metals As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Remarks (Physical Parameters)	Reference
Cladophora glomerata	BCF		4.9×10^4		2.2×10^3	1.6×10^4			2.9×10^3	Deadman Bay (L. Ont.) field	Keeney et al.,
	[] in plants		3.9		7.2	9.5			23.7	data (near Kingston)	1976
Cladophora glomerata	BCF		1.8×10^4		1.9×10^3	2.0×10^4			1.0×10^3	Main Duck Island area (L. Ont.)	Keeney et al.,
	[] in plants		1.4		6.4	12.2			8.2	field data (unpopulated)	1976
Lemna minor	BCF		735.3			677.4				control pond - Illinois River	Mathis et al.,
	[] in plants		1.25			2.1				values except water estimated	1979
	[] in water		0.0017			0.0031				from graph	
Lemna minor	BCF		414.3			629.6				experimental pond-Illinois River	Mathis et al.,
	[] in plants		0.58			1.7				values except water estimated	1979
	[] in water		0.0014			0.0027				from graph	
Ceratophyllum demersum	BCF		182.2			85.3				Wintergreen Lake, Mi.,	Mathis &
	[]in plants		0.164			1.45				[]in sediments (dry wt.)	Kevern,
	[]in water		0.9 ppb			0.017				mean levels	1975
	[]in sediments		1.85			31.9					
Lemna minor	BCF		≈ 79-240							estimated from graphs	Mangi et al., 1978
Zooplankton	BCF	1000		206	≈1000		3330		1440	L. Michigan field data	Copeland & Ayers, 1972 in Konasewich et al., 1978
	[]in plankton	1		0.35	5		0.09		23		
Zooplankton	BCF		441.1			464.7				Wintergreen Lake - field data	Mathis & Kevern,
	[]in plankton		0.397			7.9	ND			(mean levels)	1975
	[]in water		0.9ppb			0.017	ND			consisted of Chaoborus sp. & Daphnia sp.	
	[] in sediments		1.85			31.9	0.095			[]in sediments (dry wt.)	

Appendix 3a. continued

Organism		Metals As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Remarks (Physical Parameters)	Reference
Zooplankton	BCF					$1.0x10^4$-$6.3x10^4$				3 English lakes - field data (ranges of values)	Denny & Welsh, 1979
	[]in animals					2.0-19.0					
	[]in water					0.2-.50ppb					
Daphnia magna	BCF						$1.7x10^4$-$2.0x10^4$			laboratory experiment-21 day exposure; inorganic mercuric chloride	Biesinger et al., 1982
	[]in animals						8.59-23.28				
	[]in control						1.26				
	[]in water						0.36-2.7ppb				
Daphnia magna	BCF						$3.95x10^5$-$7.0x10^5$			laboratory experiment-21 day exposure; organic methyl mecuric chloride	Biesinger et al., 1982
	[]in animals						16.42-183.75				
	[]in control						0.92				
	[]in water						0.04-0.26ppb				
Physa sp.	BCF		3529			7000				control power plant pond (Illinois R.)	Mathis et al., 1979
	[]in animals		6.0			21.7				[] in animals estimated from graph	
	[]in water		0.0017			0.0031					
Physa sp.	BCF		4521			12333				experimental power plant pond (Illinois R.)	Mathis et al., 1979
	[]in animals		6.33			33.3				[]in animals estimated from graph	
	[]in water		0.0014			0.0027					
Musculium transversum	BCF		1764			10870				control power plant pond (Illinois R.)	Mathis et al., 1979
	[]in animals		3.0			33.7				[]in animals estimated from graph	
	[]in water		0.0017			0.0031					
Musculium transversum	BCF		6928			10962				experimental power plant pond (Illinois R.)	Mathis et al., 1979
	[]in animals		9.7			29.6				[]in animals estimated from graph	
	[]in water		0.0014			0.0027					
Fusconia flava	BCF		1150	366	1700	1850		1050	2129	Illinois River mean of 17 samples	Mathis & Cummings, 1973
	[]in animals		0.69	7.7	1.7	3.7		2.1	66		
	[]in water		0.6 ppb	0.021	0.001	0.002		0.002	0.031		

Appendix 3a. continued

Organism		Metals							Remarks (Physical Parameters)	Reference
		As	Cd	Cr	Cu	Pb	Hg	Ni	Zn		
Amblema plicata	BCF		633	209	1200	1350		550	3064	Illinois River	Mathis &
	[]in animals		0.38	4.4	1.2	2.7		1.1	95	mean of 25 samples	Cummings, 1973
	[]in water		0.6ppb	0.021	0.001	0.002		0.002	0.031		
Quadrula quadrula	BCF		933	233	1700	1100		450	1548	Illinois River	Mathis &
	[]in animals		0.56	4.7	1.7	2.2		0.9	48	mean of 20 samples	Cummings, 1973
	[]in water		0.6ppb	0.021	0.001	0.002		0.002	0.031		
Physa gyrina	accum. rate		0.550ppm/hr							24 hr. laboratory exposure	Wier & Walter,
	[]in water		1.30							experiment; accu. rate for snails of average mass.; uptake for 0.05 gm less than average = 0.787 ppm/hr, more than average = 0.366 ppm/hr	1976
Asellus meridianus (whole animal)	BCF				1460	24333.4				8 day laboratory experiment values mean	Brown, 1977
	initial []in animals				486.7	333.3				of 3 pops. from sites on the	
	final []in animals				1216.7	12500				Rivers Hayle&Cannel BCF = (final[]-initial[])/	
	[]in water				0.5	0.5				([]in water)., appears that heptopancreas & intestine accumulates much of the Cu & Pb	
Tubificids (Limnodrilus hoffmeisteri & Tubifex tubifex)	BCF		1833	476	23000	8500		5500	1322	Illinois River - field data	Mathis &
	[]in animals		1.1	10	23	17		11	41		Cummings, 1973
	[]in water		0.6ppb	0.021	0.001	0.002		0.002	0.031		
	[]in sediments		2.0	17.0	19.0	28		27.0	81.0		
Tubificids (Limnodrilus hoffmeisteri)	BCF		2000			2806				Control power plant pond (Illinois River); [] in animals estimated from graph	Mathis et al., 1979
	[]in animals		3.4			8.7					
	[]in water		0.0017			0.0031					
Tubificids (Limnodrilus hoffmeisteri)	BCF		4928			925				experimental power plant pond (Illinois R.) []in animals estiamted from graph	Mathis et al., 1979
	[]in animals		6.9			2.5					
	[in water		0.0014			0.0027					

Appendix 3a. continued

Organism		Metals As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Remarks (Physical Parameters)	Reference
Amblema perplicata	BCF		13.3-851.3					1120.1-1922.4		7 day exposure (caged clams)	Adams et al.,
	[]in whole body		1.43-7.78					241-388		field data; Williamson Ditch &	1981
	[]in controls		1.31					127		Trimble Creek, Ind.	
	[]in water		3.15-9.03ppb					59.3-216.9ppb		contaminated by electroplating discharges	
	[]in sediments		9.53-35.4					185-272		BCF = ([whole body]-[control])/ambient[];	
										-clams not allowed purging time	
Tubifex sp.	[]in controls		0.1	<2	12-18	2-4			220-249	Ashtabula River sediments test	Slowey & Neff,
	[]in purged		0.1	<1	13	4			207		1977
	[]in unpurged		1.1	38	44	25			220	4 day exposure	
	[]in water			<5ppb	8ppb	1ppb			85ppb	to sediment	
	elutriate []			<5ppb	6ppb	1.5ppb			50ppb	[]in purged - gut purged for 1 day	
	[]in sediments			175	37	42			315	prior to analysis	
Chironomids	BCF			2960	546				15847	Skeleton Creek, Oklahoma	Namminga &
	[]in animals			2.96	1.91	1.32			57.05	chironomid data	Wilhm, 1977
	[]in water			1.0ppb	3.5ppb				3.6ppb	was given for only 1 of 5 sites	
	[]in sediments			7.6	1.8	3.3			16.0	(in August)	
benthos	BCF (water)	2000		382	≈1400		5185		875	L. Michigan - field data	Copeland &
	BCF (sediment)			4.2							Ayers, 1972 in
	[]in animals	2.0		0.65	7		0.14		14		Konasewich
											et al., 1978
Glossiphonia heteroclita	BCF		882.4			1354.8					
	[]in animals		1.5			4.2				control power plant pond	Mathis et al.,
	[]in water		0.0017			0.0031				(field data); []in animals	1979
	[]in sediments		0.83			11.7				estimated from graph (Illinois R.)	
Glossiphonia heteroclita	BCF		2264.3			2481.5					
	[]in animals		3.17			6.7				experimental power plant pond	Mathis et al.,
	[]in water		0.0014			0.0027				(field data) (Illinois R.)	1979
	[]in sediments		0.75			11.3				[]in animals estimated from graph	
benthos	BCF	35	33.3	60.6	79.5		167		38.5	Savannah River	Cherry &
	[]in animals	2.1	4.0	9.7	31		0.5		15	1974 (before dredging)	Guthrie,
	[]in water	0.06	0.12	0.16	0.39		0.03		0.39	benthos incl. chironomids, crayfish,	1977
	[]in sediments	20	1.7	38	81		0.8		6.4	odonates and coleopterans	

Appendix 3a. continued

Organism		Metals As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Remarks (Physical Parameters)	Reference
benthos	BCF	857.1	27.8	17.5	167.5		10		7.6	Savannah River	Cherry &
	[]in animals	60	2.5	3.5	67		0.2		25	1975 (after dredging)	Guthrie, 1977
	[]in water	0.07	0.09	0.20	0.40		0.02		3.3	benthos incl. chironomids, crayfish,	
	[]in sediments	27	3.8	34	32		0.8		7.6	odonates, & coleopterans.	
Stenomena sp.	[]in animals		5.55		17.55	30.04			252.92	Fox River, Wisconsin & Illinois	Anderson, 1977a
Hexagenia sp.	[]in animals		<0.5		11.29	29.12			177.83	field data, (dry wt.)	
Chironomidae	[]in animals		2.17		13.05	29.74			144.21		
Sigara sp.	[]in animals		<0.5		19.47	19.51			172.78		
Orconectes sp.	[]in animals		1.60		86.61	25.68			107.12	missing []in H_2O	
Asellus sp.	[]in animals		2.62		99.19	22.05			124.94	no BCF can be calculated at this time	
Gammarus sp.	[]in animals		<0.5		70.74	<4.00			101.19		
Physa sp.	[]in animals		2.97		22.01	21.64			69.93		
Campeloma sp.	[]in animals		1.76		18.37	21.79			99.58		
Gonabasis sp.	[]in animals		2.19		13.40	19.73			22.69		
Pleurocera sp.	[]in animals		2.31		10.70	24.08			19.19		
Sphaerium sp.	[]in animals		1.99		10.06	32.18			61.07		
Lampsilis sp.(body parts)	[]in animals		2.23		12.67	21.93			353.04		
Anodonata sp.(body parts)	[]in animals		1.78		6.88	13.73			232.10		
Anodonta sp. (shell)	[]in animals		1.35		9.30	10.19			3.70		
Erpobdella sp.	[]in animals		3.80		16.83	39.78			136.23		
Placobdella sp.	[]in animals		<0.5		7.59	<4.00			148.37		
Helisoma trivolvis	BCF						≈2000			laboratory exp't.-40wk exposure	Titus et al.,
	[]in animals						≈120			all values est. from graphs	1980
	[]in water						≈0.06			BCF did not plateau	
Campeloma decisa	BCF						≈800			laboratory exp't.-40wk exposure	Titus et al.,
	[]in animals						≈50			all values est. from graphs	1980
	[]in water						≈0.06			BCF did not plateau	
Ephemerella grandis	BCF				903-1817.9	8080.3-14912.9			135.3-1130	flow through 14 day exposure	Nehring, 1976
	[]in animals				1240-9125	5702-104700			1794-2381	(or until death)	
	[]in controls				94.7	126.6			1116	[]in control (for Zn & Cu) natural	
	[]in water				0.63-10	0.69-9.24			0.6-9.2	background levels, (for Pb) -	
										due to holding tank contamination by	
										lead base paint chips.	

Appendix 3a. continued

Organism		Metals As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Remarks (Physical Parameters)	Reference
Daphnia galeata mendotae	BCF		1.76×10^4							L. Michigan water	Marshall, 1978
	(S.E.)		±0.04							22 wk. exposure	
	[]in animal		17.6							[]in animal in dry wt.	
	(S.E.)		±0.36								
	[]in water		1.0ppb								
Daphnia galeata mendotae	BCF		0.65×10^4							L. Michigan water	Marshall, 1978
	(S.E.)		±0.03							22 week exposure	
	[]in animal		51.7							[]in animal in dry wt.	
	(S.E.)		±2.67								
	[]in water		8ppb								

Appendix 3a. continued

Organism		Metals								Remarks (Physical Parameters)	Reference
		As	Cd	Cr	Cu	Pb	Hg	Ni	Zn		
Yellow perch	BCF		44.4			22.2				Wintergreen Lake; field data (mean levels) filleted samples	Mathis & Kevern, 1975
	[]in fish		0.04			0.378	0.124				
	[]in water		0.9ppb			0.017	N.D				
	[]in sediments		1.85			31.9	0.095				
fish	BCF	2000		382	≈1400		5185		875	L. Michigan - field data edible portions only	Copeland & Ayers, 1972 in Konasewich, et al., 1979
	[] in fish			0.01			0.15		9.0		
Morone chrysops	BCF		40	2.9	190	225		40	145.2	Illinois River - field data mean values muscle tissue	Mathis & Cummings, 1973
	[]in fish		0.024	0.06	0.19	0.45		0.08	4.5		
	[]in water		0.6ppb	0.021	0.001	0.002		0.002	0.031		
	[]in sediments		2.0	17.0	19.0	28.0		27.0	81.0		

Appendix 3b.

ORGANISM		ORGANICS PCB's ISOMER		REMARKS	REFERENCES
Fragilaria crotonensis	BCF live cells	2,4,5,2',4',5' hexachlorobiphenyl	1.17×10^5	laboratory experiment time - live cells 19 hrs - dead cells } 19 hrs & 43 hrs averaged - frustules }	Lederman & Rhee, 1982
	BCF dead		3.25×10^5		
	BCF frustules		0.38×10^5		
	pop. density		320,00/ml		
Fragilaria crotonensis	BCF live cells		3.13×10^5	laboratory experiment time - live cells 19 hrs dead cells } 19 hrs & 43 hrs. averaged frustules }	Lederman & Rhee, 1982
	BCF dead		6.98×10^5		
	BCF frustules		0.18×10^5		
	pop. density		16,000/ml		
	[] in water		3 ppb		
Chlorella pyrenoidosa	BCF	tetrachloro	3200	laboratory experment 1 hr. exposure	Urey et al., 1976
	[]in cells		32		
	[]in water		10ppb		
Chlorella pyrenoidosa	BCF	hexachloro	7000	laboratory experiment - 1 hr exposure	Urey et al., 1976
	[]in cells		70		
	[]in water		10ppb		
Chlorella pyrenoidosa	BCF	octachloro	1600	laboratory experiment - 1 hr exposure	Urey et al., 1976
	[]in cells		16		
	[]in water		10ppb		
Chlorella pyrenoidosa	BCF		5200	laboratory experiment - 1 hr. exposure	Urey et al., 1976
	[] in cells		52		
	[]in water		10ppb		
Chlorella pyrenoidosa	BCF	tetrachloro	6200	laboratory experiment uptake by dead cells> uptake by live cells	Urey et al., 1976
	[]in dead cells		62		
	[]in water		10ppb		

Appendix 3b. continued

ORGANISM	ORGANICS			REMARKS	REFERENCES
		PCB's ISOMER			
Chlorella pyrenoidosa	BCF	hexachloro	15600	laboratory experiment	Urey et al., 1976
	[]in dead cells		156	uptake by dead cells 2 times greater	
	[]in water		10ppb	than live cells	
Net plankton	BCF	Aroclor 1254	12,996.4	L. Ontario field data	Haile et al., 1975 in
	[] in plankton	equivalent	0.72	[]in plankton (64µmmesh) - wet wt.	Thomann, 1979
	[]in water		0.0554ppb	[]in H_2O (<64µm)from	
Daphnia magna	BCF	Aroclor 1254	47272.7	laboratory experiment - 4 day	Sanders & Chandler, 1972
	[]in animals		52	exposure	
	[]in water		1.1ppb		

Appendix 3b. continued

ORGANISM		ORGANICS PCB's ISOMER		REMARKS	REFERENCES
Physa sp.	BCF	lindane & AROCLOR 5460	452	model ecosystem	Sanborn, 1974 in Thomas 1975
	BCF	2,5,2'- trichlorobiphenyl	5795		
	BCF	2,5,2',5'-tetra- chlorobiphenyl	39,439		
	BCF	2,4,5,2',5'- pentachlorobiphenyl	59,629		
Chaoborus punctipennis	BCF [ ]in animals [ ]in water	AROCLOR 1254	23076.9 30 1.3ppb	laboratory experiment - 4 day exposure; organisms collected in field (Missouri)	Sanders & Chandler, 1972
Orconectes nais	BCF [ ]in animals [ ]in water	AROCLOR 1254	166.7 0.2 1.2ppb	laboratory experiment - 4 day exposure; organisms collected in field (Missouri); max. observed BCF≈5200 on day 21 (no plateau observed)	Sanders & Chandler, 1972
Gammarus pseudolimnaeus	BCF [] in animals [] in water	AROCLOR 1254	24375 39 ppm 1.6 ppb	laboratory experiment 4 day exposure animals collected in Missouri from graph BCF max. ≈ 28500 between day 7 & 14 (plateau)	Sanders & Chandler, 1972

Appendix 3b. continued

ORGANISM		ORGANICS PCB's ISOMER		REMARKS	REFERENCES
Alewife, smelt, slimy sculpin	BCF []in fish []in water	AROCLOR 1254 equivalent	53483.8 3.24 0.0554ppb	L. Ontario - field data []in water (< 64µm) []in fish (wet wt.)	Haile et al., 1975 in Thomann, 1979.
Lake Trout	BCF []in fish []in water	PCB's	3.41×10^6 28.0 8.2×10^{-6}	L. Michigan field data	cited in Metcalf, 1977

Appendix 4

Concentration of Contaminants

found in the fish of lake Erie

in parts per million (ppm)

Fish	Basin	Year	Level	Range	Remarks	References
ARSENIC						
Barbote	Eastern	1975	0.03		wet wt.	Traversy et al., 1975
Carp	Central	1975	0.12		wet wt.	Traversy et al., 1975
Channel catfish	Western	1973	0.02 0.03		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
	Central	1975	0.09		wet wt.	Traversy et al., 1975
	Central	1976	0.03			Brezina & Arnold, 1977 in Konasewich et al., 1978
Smelt	Central	1975	0.04		wet wt.	Traversy et al., 1975
	Central	1976		0.13 - 0.16	Wheatley Dock	Ont. M.N.R. (unpubl.) in Konasewich et al., 1978
	Eastern	1975	0.12		wet wt.	Traversy et al., 1975
Walleye	Western	1973	0.02		off Swan Creek	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
White bass	Eastern	1975	0.08		wet wt.	Traversy et al., 1975
White sucker	Central	1976	0.01			Brezina & Arnold, 1977 in Konasewich et al., 1978
Yellow perch	Western	1973	< 0.01 0.10		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
	Western	1975	0.09		wet wt.	Traversy et al., 1975
Yellow perch	Central	1975		0.03 - 0.10	wet wt.	Traversy et al., 1975
	Central	1976	0.01			Brezina & Arnold, 1977 in Konasewich et al., 1978
	Eastern	1975		0.05 - 1.2	wet wt.	Traversy et al., 1975

Fish	Basin	Year	Level	Range	Remarks	References
CADMIUM						
Brown bullhead & channel catfish	Central	1976	0.11		Presque Ile Peninsula	Brezina & Arnold, 1977 in Konasewich et al., 1978
Channel catfish	Western	1973	0.78 0.02		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
Smelt	Central	1976	< 0.1		Wheatley Dock	Ont. M.N.R. (unpubl.) in Konasewich et al., 1978
Walleye	Western	1973	0.10 0.14		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
White crappie	Central	1976	0.07			Brezina & Arnold, 1977 in Konasewich et al., 1978
White sucker	Central	1976	0.05			Brezina & Arnold, 1977 in Konasewich et al., 1978
Yellow perch	Western	1973	0.02 0.02		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
	Central	1976	0.01			Brezina & Arnold, 1977 in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
CHROMIUM						
Channel catfish	Western	1973	0.18 0.39		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl) in Konasewich et al; 1978
	Central	1976	0.13			Brezina & Arnold, 1977 in Konasewich et al., 1978
Coho salmon	Whole lake	1978	0.30 ± 0.07			Great Lakes Water Qualilty, 1979, App. B
Walleye	Western	1973	0.20 0.12		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
Yellow perch	Western	1973	0.18 0.29		off Swan Creek Whiting Power Plant	Mich D.N.R. (unpubl.) in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
COPPER						
Channel catfish	Western	1973	0.61 0.40		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
Channel catfish & brown bullhead	Central	1976	0.51		Presque Ile Peninsula	Brezina & Arnold, 1977 in Konasewich et al., 1978
Coho salmon	Whole lake	1978	1.56 ± 2.05		wet wt. whole fish	Great Lakes Water Quality, 1979, App. B
Rainbow smelt	Whole lake	1978	0.36 ± 0.17		wet wt. whole fish (5 fish in composite)	Great Lakes Water Quality, 1979, App. B
	Central	1976		0.68 - 0.80	Wheatley Dock	Ont. M.N.R. 1976 (unpubl.) in Konasewich et al., 1978
Walleye	Western	1973	0.44 0.32		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
White crappie	Central	1976	0.24		Presque Ile Peninsula	Brezina & Arnold, 1977 in Konasewich et al., 1978
White sucker	Central	1976	0.82		Outer Harbour, Erie, Pa.	Brezina & Arnold, 1977 in Konasewich et al., 1978
Yellow perch	Whole lake	1978	0.70 ± 0.53		wet wt. whole fish	Great Lakes Water Quality, 1979, App. B
	Western	1973	0.30 0.41		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
	Central	1976	0.18		Outer Harbour, Erie, Pa.	Brezina & Arnold, 1977 in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
LEAD						
Channel catfish	Western	1973	0.34 0.25		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
	Central	1976	0.38		Presque Ile Peninsula	Brezina & Arnold, 1977 in Konaswich et al., 1978
Northern pike	Eastern	1978	0.10	<0.10 - 0.10	Long Pt. Bay	Whittle, 1980 (unpubl.) in Great Lakes Science Advisory Bd., 1980
Smelt	Western	1978	0.13 + 0.01	<0.10 - 0.21		Whittle, 1980 (unpubl.) in Great Lakes Science Advisory Bd., 1980
	Central	1976	0.40		Wheatley Dock	Ont. M.N.R. (unpubl.) in Konosewich et al., 1978.
	Central	1978	0.12 ± 0.01 0.13 ± 0.01	<0.10 - 0.16 <0.10 - 0.19	Erieau Wheatley	Whittle, 1980 (unpubl.) in Great Lakes Science Advisory Bd., 1980
	Eastern	1978	0.11 ± 0.01	<0.10 - 0.12	Long Pt. Bay	Whittle, 1980 (unpubl.) in Great Lakes Science Advsiory Bd., 1980
Walleye	Western	1973	0.52 0.30		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
	Western	1978	0.13 ± 0.03	<0.10 - 0.16		Whittle, 1980 (unpubl.) in Great Lakes Science Advsiory Bd., 1980
	Central	1978	0.18 ± 0.01	<0.10 - 0.18	Erieau	Whittle, 1980 (unpubl.) in Great Lakes Science Advisory Bd., 1980
White sucker	Central	1970	0.09		Outer Harbour, Erie, Pa.	Brezina & Arnold, 1977 in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
LEAD						
Yellow perch	Western	1973	0.30 0.20		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
Yellow perch	Western	1978	0.15 ± 0.01	<0.10 - 0.28		Whittle, 1980 (unpubl.) in Great Lakes Science Advisory Bd., 1980
	Central	1976	0.14		Outer Harbor, Erie, Pa.	Brezina & Arnold, 1977 in Konasewich et al; 1978
	Central	1978	0.20 ± 0.02 0.16 ± 0.01	<0.10 - 0.40 <0.10 - 0.28	Erieau Wheatley	Whittle, 1980 (unpubl.) in Great Lakes Science Advisory Bd., 1980
	Eastern	1978	0.16 ± 0.01	<0.10 - 0.38	Long Pt. Bay	Whittle, 1980 (unpubl.) in Great Lakes Science Advisory Bd., 1980

Fish	Basin	Year	Level	Range	Remarks	References
MERCURY						
Alewife	Whole lake	1976	0.08	0.07 - 0.10		Sherbin, 1979
Brown bullhead	Whole lake	1967-1968	0.19 ± 0.03	0.12 - 0.26	fillet	Thommes et al., 1972 in Konasewich et al., 1978
Channel catfish & brown bullhead	Central	1976	0.08			Brezina & Arnold, 1977 in Konasewich et al., 1978
Carp	Whole lake	1967-1968	0.22 ± 0.08	0.07 - 0.30	fillet	Thommes et al., 1972 in Konasewich et al., 1978
	Whole lake	1976	0.23	0.13 - 0.42		Sherbin, 1979
	Western	1970	0.28 0.08		Bono fillet Sandusky fillet	Fed. Water Quality Admin., 1970 in Konasewich et al., 1978
	Western	1970	0.23		edible tissue (25 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Central	1970	0.35		edible tissue (17 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Eastern	1970	0.36		edible tissue (14 fish in composite	Willford, 1971 in Konasewich et al., 1978
Channel catfish	Western	1970	0.32 1.8 1.3		Sandusky; fillet Bono; fillet Monroe; fillet	Fed. Water Quality Admin., 1970 in Konasewich et al., 1978
	Western	1970	0.36		edible tissue (25 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Western	1973	0.49 0.41		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
	Central	1970	0.42		edible portion (20 fish in composite)	Willford, 1971 in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
MERCURY						
Coho salmon	Whole lake	1969	0.36		whole fish	Fed. Water Quality Admin., 1970 in Konasewich et al., 1978
	Whole lake	1976	0.20	0.11 - 0.35		Sherbin, 1979
	Whole lake	1978	0.15 ± 0.06		ea. sample - 5 fish (whole fish)	Great Lakes Water Quality, 1979, App. B
	Western	1970	0.24 0.96 0.96		Sandusky; fillet Bono; fillet Monroe; fillet	Fed. Water Quality, Admin., 1970 in Konasewich et al., 1978
	Western	1970	0.69		edible tissue (20 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Central	1970	0.58		edible tissue (10 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Eastern	1970	0.51		edible tissue (13 fish in composite)	Willford, 1971 in Konasewich et al., 1978
Drum	Whole lake	1967-1968	0.35 ± 0.04	0.19 - 0.40	(fillet)	Thommes et al., 1972 in Konasewich et al., 1978
	Western	1970	0.67		edible portion (25 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Central	1970	0.62		edible portion (20 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Eastern	1970	0.30		edible portion (25 fish in composite)	Willford, 1971 in Konasewich et al., 1978
Gizzard shad	Whole lake	1967-1968	0.14 ± 0.04	0.05 - 0.25	fillet	Thommes et al., 1972 in Konasewich et al., 1978
	Western	1970	0.24		Sandusky fillet	Fed. Water Quality Admin., 1970 in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
MERCURY						
Gizzard shad	Western	1970	0.22		edible portion (25 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Central	1970	0.21		edible portion (15 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Eastern	1970	0.26		edible portion (18 fish in composite)	Willford, 1971 in Konasewich et al., 1978
Goldfish	Whole lake	1967-1968	0.13 ± 0.06	0.01 - 0.20		Thommes et al., 1972 in Konasewich et al., 1978
Rainbow smelt	Whole lake	1978	0.05 ± 0.02		ea. sample = 5 fish (whole fish)	Great Lakes Water Quality, 1979, App. B
	Central	1976		0.03 - 0.15	Wheatley Dock	Ont M.N.R. (unpubl.) in Konasewich et al., 1978
	Eastern	1970	0.30		Whole fish (10 fish in composite)	Willford, 1971 in Konasewich et al., 1978
Sheepshead	Western	1970	0.24		Sandusky; fillet	Fed. Water Quality Admin., 1970 in Konasewich et al., 1978
Smallmouth bass	Central	1970	0.55		edible tissue (14 fish in composite)	Willford, 1971 in Konasewich et al., 1978
Steelhead	Western	1970	<0.15		Monroe; fillet	Fed. Water Quality Admin., in Konasewich et al., 1978
Walleye	Whole lake	1967-1968	0.84		fillet	Thommes et al., 1972 in Konasewich et al., 1978
	Western	1970	3.6 2.6 3.57		Monroe; fillet Sandusky; fillet Raison Pt.; fillet	Fed. Water Quality Admin., 1970 in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
MERCURY						
Walleye	Western	1970	0.79		edible tissue (25 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Western	1971	0.55	0.4 - 0.94	fish tissue	Kinkead & Hamdy, 1978
	Western	1972	0.58	0.14 - 1.35	fish tissue	Kinkead & Hamdy, 1978
	Western	1973	0.79 0.35		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konaswich et al., 1978
	Western	1974	0.52	0.2 - 1.06	fish tissue	Kinkead & Hamdy, 1978
	Western	1975	0.68	0.15 - 1.98	fish tissue	Kinkead & Hamdy, 1978
	Western	1976	0.31	0.09 - 1.25	fish tissue	Kinkead & Hamdy, 1978
	Central	1970	0.65		edible tissue (25 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Eastern	1970	0.33		edible tissue (25 fish in composite	Willford, 1971 in Konasewich et al., 1978A
White Bass	Whole lake	1967-1968	0.40 ± 0.03	0.38 - 0.43	fillet	Thommes et al., 1972 in Konasewich et al., 1978
	Whole lake	1972	0.39	0.15 - 1.34		Sherbin, 1979
	Whole lake	1975	0.63	0.17 - 1.09		Sherbin, 1979
	Whole lake	1977	0.46	0.14 - 0.82		Sherbin, 1979
	Western	1970	0.80 0.80 0.53		Bono; fillet Sandusky; fillet Raison Pt.; fillet	Fed. Water Quality Admin., 1970 in Konasewich et al., 1978
	Western	1970	0.60		edible tissue (25 fish in composite)	Willford, 1971 in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
MERCURY						
White bass	Western	1971	1.19	0.49 - 2.12	fish tissue	Kinkead & Hamdy, 1978
	Western	1972	0.53	0.08 - 1.96	fish tissue	Kinkead & Hamdy, 1978
	Western	1975	0.77	0.12 - 1.57	fish tissue	Kinkead & Hamdy, 1978
	Western	1976	0.31	0.26 - 0.37	fish tissue	Kinkead & Hamdy, 1978
	Western	1977	0.21	0.06 - 1.06	fish tissue	Kinkead & Hamdy, 1978
	Central	1970	0.72		edible tissue (25 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Eastern	1970	0.43		edible tissue (25 fish in composite)	Willford, 1971 in Konasewich et al., 1978
White crappie	Central	1976	0.12		Presque Ile Peninsula	Brezina & Arnold, 1977 in Konasewich et al., 1978
White sucker	Western	1970	0.55		edible tissue (24 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Central	1970	0.56		edible tissue (8 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Central	1976	0.09		Outer Harbor, Erie, Pa.	Brezina & Arnold, 1977 in Konasewich et al., 1978
	Eastern	1970	0.35		edible tissue (25 fish in composite)	Willford, 1971 in Konasewich et al., 1978
Yellow perch	Whole lake	1967-1968	0.42 ± 0.05	0.29 - 0.61	fillet	Thommes et al., 1972 in Konasewich et al., 1978
	Whole lake	1978	0.09 ± 0.05		ea. sample = 5 fish (whole fish)	Great Lakes Water Quality, 1979, App. B

Fish	Basin	Year	Level	Range	Remarks	References
MERCURY						
Yellow perch	Western	1970	0.44 0.32 1.7		Bono; fillet Sandusky; fillet Monroe; fillet	Fed. Water Quality Admin., 1970 in Konasewich et al., 1978
	Western	1970	0.61		edible tissue (25 fish in composite)	Willford, 1971 in Konasewich et al., 1978
	Western	1973	0.55 0.57		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
	Central	1969	0.25		Wheatley Dock (whole fish)	Fed. Water Quality Admin., 1970 in Konasewich et al., 1978
	Central	1970	0.49		edible tissue (25 fish in composite	Willford, 1971 in Konasewich et al., 1978
	Central	1976	0.28		Outer Harbour, Erie, Pa.	Brezina & Arnold, 1977 in Konasewich et al., 1978
	Eastern	1970	0.29		edible tissues (25 fish in composite)	Willford, 1971 in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
NICKEL						
Channel catfish	Western	1973	0.12 0.23		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
Channel catfish & brown bullhead	Central	1976	0.41		Presque Ile Peninsula	Brezina & Arnold, 1977 in Konasewich et al., 1978
Walleye	Western	1973	0.13 0.20		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
White Crappie	Central	1976	0.68		Presque Ile Peninsula	Brezina & Arnold, 1977 in Konasewich et al., 1978
White sucker	Central	1976	1.91		Outer Harbour, Erie, Pa.	Brezina & Arnold, 1977 in Konasewich et al., 1978
Yellow perch	Western	1973	0.20 0.20		off Swan Creek Whiting Power Plant	Mich. D.N.R. (unpub.) in Konasewich et al., 1978
	Central	1976	7.53		Outer Harbor, Erie, Pa.	Brezina & Arnold, 1977 in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
ZINC						
Channel catfish	Western	1973	6.05		off Swan Creek	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
			5.35		Whiting Power Plant	
	Central	1976	5.50			Brezina & Arnold, 1977 in Konasewich et al., 1978
Rainbow smelt	Central	1976		20 - 24		Ont. M.N.R. (unpubl.) in Konasewich et al., 1978
Walleye	Western	1973	4.34		off Swan Creek	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
			4.36		Whiting Power Plant	
White sucker	Central	1976	4.34			Brezina & Arnold, 1977 in Konasewich et al., 1978
Yellow perch	Western	1973	4.59		off Swan Creek	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
			4.27		Whiting Power Plant	
	Central	1976	4.8			Brezina & Arnold, 1977 in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
PCB's						
Alewife	Western	1975	0.5	0.4 - 0.5	headless and eviscerated (21 fish in composite)	Frank et al., 1978
	Central	1975	0.4	0.3 - 0.4	headless and eviscerated (22 fish in composite)	Frank et al., 1978
	Eastern	1971	3.0	1.9 - 3.7	headless and eviscerated	Frank et al., 1978
Bass	Eastern	1978	0.57	0.37 - 0.86	muscle; Port Dover	Rees, et al., 1979
Black crappie	Eastern	1968	<0.1	<0.1 - 0.1	headless and eviscerated	Frank et al., 1978
	Eastern	1976	N.D.		muscle	Crawford & Brunato, 1978
Bluegill	Central	1978	0.06	0.02 - 0.23	muscle; Rondeau Bay	Rees, et al., 1979
	Eastern	1968	<0.1		headless and eviscerated	Frank et al., 1978
	Eastern	1976	0.003	N.D. - 0.02	muscle	Crawford & Brunato, 1978
Brown bullhead	Eastern	1968	<0.1		headless and eviscerated	Frank et al., 1978
	Eastern	1976	N.D.		muscle	Crawford & Brunato, 1978
Burbot	Central	1971	0.30 0.45			MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Eastern	1976	N.D.		muscle	Crawford & Brunato, 1978
Carp	Western	1974	3.7 3.9 ± 1.6		Monroe <5 lbs. >5 lbs.	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
	Western	1975	0.13 ± 0.11 0.24		<5 lbs. >5 lbs.	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
	Western	1977	0.21 0.32		whole body	Herdendorf et al., 1978 in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
PCB's						
Carp	Eastern	1970-1971	2.0	0.3 - 5.3		MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Eastern	1976	0.003	N.D. - 0.01	muscle	Crawford & Brunato, 1978
Channel catfish	Western	1968	0.2	<0.1 - 0.2	headless and eviscerated	Frank et al., 1978
	Western	1971	5.0	4.2 - 5.7	headless and eviscerated	Frank et al., 1978
	Western	1971	5.0			MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Western	1974	2.97 3.0		length >17" length 17 - 20"	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
	Western	1975	0.16 ± 0.13 0.30		length >17" length 17 - 20"	Mich. D.N.R. (unpubl.) in Konasewich et al., 1978
	Western	1977	3.59 3.14		dressed fish carcass	Herdendorf et al., 1976 in Konasewich et al., 1978
	Central	1976	0.93		fillet analysed	Penn. D.N.R., 1977 in Konasewich et al., 1978
	Eastern	1970-1971	4.4	1.4 - 7.8		MOE, 1976 (unpubl.) in Konasewich et al., 1978
Chinook salmon	Central	1977	0.135		muscle	Crawford & Brunato, 1978
Coho salmon	Whole lake	1978	0.91 ± 0.66		wet wt. whole fish	Great Lakes Water Quality, 1979, App. B
	Western	1975	1.4	0.6 - 2.7	headless and eviscerated	Frank et al., 1978
	Central	1968	0.3	0.2 - 0.4	headless and eviscerated	Frank et al., 1978
	Central	1970	4.0	1.0 - 14.0	headless and eviscerated	Frank et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
PCB's						
Coho salmon	Central	1971	1.7	1.5 - 2.0	headless and eviscerated	Frank et al., 1978
	Central	1975	0.7	0.4 - 2.0	headless and eviscerated	Frank et al., 1978
	Central	1977	0.52 1.13	0.23 - 0.98 0.785 - 1.55	muscle	Crawford & Brunato, 1978
	Central	1977	0.53	0.17 - 1.21	muscle; Port Stanley	Crawford & Brunato, 1978
	Eastern	1976	0.3	0.1 - 0.5	headless and eviscerated	Frank et al., 1978
	Eastern	1978	1.89		muscle Inner Long Pt. Bay	Reese, et al., 1979
Drum	Western	1968	<0.1		headless and eviscerated	Frank et al., 1978
	Western	1971	1.4	0.7 - 3.5	headless and eviscerated (9 fish in composite)	Frank et al., 1978
	Western	1975	0.6	0.2 - 1.8	headless and eviscerated (23 fish in composite)	Frank et al., 1978
	Central	1971	3.7	2.2 - 4.7	headless and eviscerated	Frank et al., 1978
	Central	1975	0.7	0.4 - 1.4	headless and eviscerated	Frank et al., 1978
	Eastern	1971	1.3	0.6 - 1.8	headless and eviscerated	Frank et al., 1978
	Eastern	1975	0.4	0.2 - 0.6	headless and eviscerated	Frank et al., 1978
Emerald shiner	Western	1975	0.6	0.5 - 0.7	headless and eviscerated (60 fish in composite)	Frank et al., 1978
	Central	1975	0.4	0.3 - 0.6	headless and eviscerated (12 fish in composite)	Frank et al., 1978
	Eastern	1975	0.4	0.3 - 0.6	headless and eviscerated (12 fish in composite)	Frank et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
PCB's						
Emerald shiner	Eastern	1976	0.69	N.D. - 2.5	muscle	Crawford & Brunato, 1978
Gizzard shad	Western	1968	0.3	<0.1 - 0.6	headless and eviscerated	Frank et al., 1978
	Western	1971	2.6	2.1 - 3.5	headless and eviscerated (6 fish in composite)	Frank et al., 1978
	Western	1975	0.7	0.6 - 0.9	headless and eviscerated (27 fish in composite)	Frank et al., 1978
	Central	1971	3.4	2.4 - 4.7	headless and eviscerated (9 fish in composite)	Frank et al., 1978
	Central	1975	0.5	0.4 - 0.6	headless and eviscerated (7 fish in composite)	Frank et al., 1978
Largemouth bass	Central	1977	0.64	0.36 - 0.93	muscle	Crawford & Brunato, 1978
	Central	1978	0.12	0.01 - 0.45	muscle; Rondeau Bay	Rees, et al., 1978
	Eastern	1975	0.1	0.1 - 0.3	headless and eviscerated	Frank et al., 1978
	Eastern	1976	0.003	N.D. - 0.01	muscle	Crawford & Brunato, 1978
Pumpkinseed	Eastern	1968	<0.1	<0.1 - 0.1	headless and eviscerated	Frank et al., 1978
Rainbow smelt	Whole lake	1978	0.23 ± 0.10		wet wt. whole fish (5 fish in composite)	Great Lakes Water Quality, 1979, App. B.
	Western	1973	0.5		headless and eviscerated (10 fish in composite)	Frank et al., 1978
	Western	1975	0.4	0.2 - 0.6	headless and eviscerated (60 fish in composite)	Frank et al., 1978
	Western	1976	0.06			Ont. M. Agr. Food (unpubl.) in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
PCB's						
Rainbow smelt	Western	1978	0.65	0.36 - 1.16	whole fish	Rees, et al., 1979
	Central	1968	0.2	0.2 - 0.3	headless and eviscerated (13 fish in composite)	Frank et al., 1978
	Central	1971	1.3			MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Central	1973	0.5			MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Central	1975	0.1		headless and eviscerated (70 fish in composite)	Frank et al., 1978
	Central	1976	0.59			Ont. M. Agr. Food., 1976 in Konasewich et al., 1978
	Central	1978	0.63	0.32 - 0.85	Whole fish	Rees, et al., 1979
	Eastern	1971	1.3	1.2 - 1.4	headless and eviscerated (7 fish in composite)	Frank et al., 1978
	Eastern	1975	0.3	0.1 - 0.6	headless and eviscerated (23 fish in composite)	Frank et al., 1978
	Eastern	1976	0.32	0.25 - 0.95	Long Pt. Bay	Ont. M. Agr. Food in Konasewich et al., 1978
	Eastern	1976	0.3	< 0.1 - 1.4	headless and eviscerated	Frank et al., 1978
	Eastern	1978	0.60	0.4 - 0.76	whole fish; Port Dover	Rees, et al., 1979
	Eastern	1978	0.44	0.33 - 0.56	muscle; Inner Long Pt. Bay	Rees, et al., 1979
Rainbow trout	Western	1975	0.70			Mich. D.N.R., 1976 (unpubl.) in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
PCB's						
Rainbow trout	Central	1974	0.3	<0.1 - 0.8	headless and eviscerated	Frank et al., 1978
	Central	1977	1.18	0.6 - 2.4	muscle	Crawford & Brunato, 1978
	Eastern	1978	0.77	0.34 - 1.29	muscle; Haldimand-Norfolk area	Rees, et al., 1979
Rock bass	Eastern	1968	0.2	<0.1 - 0.5	headless and eviscerated	Frank et al., 1978
	Eastern	1971	0.3	0.2 - 0.6	headless and eviscerated	Frank et al., 1978
Sauger	Central	1977	0.07	0.05 - 0.1	muscle	Crawford & Brunato, 1978
Smallmouth bass	Central	1977	1.0	0.25 - 2.69	muscle	Crawford & Brunato, 1978
	Eastern	1968	0.3	0.2 - 0.84	headless and eviscerated	Frank et al., 1978
	Eastern	1971	5.8	2.3 - 9.3	headless and eviscerated	Frank et al., 1978
	Eastern	1972	0.7	0.4 - 1.2	headless and eviscerated	Frank et al., 1978
	Eastern	1975	0.3	0.2 - 0.4	headless and eviscerated	Frank et al., 1978
Spot-tail shiner	Western	1975	0.06	0.04 - 0.07	headless and eviscerated (60 fish in composite)	Frank et al., 1978
	Western	1975	0.85 ± 0.40		Point Pelée (wet wt.)	Suns & Rees, 1978
	Eastern	1975	0.08 ± 0.03 0.06 ± 0.03		Port Colborn (wet wt.) Port Rowan (wet wt.)	Suns & Rees, 1978
Walleye	Whole lake	1976	0.11		edible portion	Penn. D.N.R., 1977 in Konasewich et al., 1978
	Western	1968	0.2	<0.1 - 0.3	headless and eviscerated	Frank et al., 1978
	Western	1971	1.0	0.5 - 1.6	headless and eviscerated	Frank et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
PCB's						
Walleye	Western	1971	1.0			MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Western	1974	0.22 ± 0.4			Mich. D.N.R., 1976 (unpubl.) in Konasewich et al., 1978
	Western	1975	0.34 ± 0.42			Mich. D.N.R., 1976 (unpubl.) in Konasewich et al., 1978
	Western	1975	1.3	0.3 - 5.1	headless and eviscerated (14 fish in composite)	Frank et al., 1978
	Western	1976	4.6			Ont. M. Agr. Food, 1976 in Konasewich et al., 1978
	Central	1977	0.13	0.02 - 0.45	muscle	Crawford & Brunato, 1978
	Eastern	1971	0.6		headless and eviscerated	Frank et al., 1978
White bass	Whole lake	1976	0.32		edible portion	Penn. D.N.R. in Konasewich et al., 1978
	Western	1968	0.1	<0.1 - 0.3	headless and eviscerated	Frank et al., 1978
	Western	1971	2.2			MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Western	1971	2.2	1.1 - 4.8	headless and eviscerated	Frank et al., 1978A
	Western	1972	5.6			MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Western	1974	2.18 ± 1.22			Mich. D.N.R., 1976 (unpubl.) in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
PCB's						
White bass	Western	1975	0.55		Monroe; 10"	Mich. D.N.R., 1976 (unpubl.) in Konasewich et al., 1978
			1.78 ± 1.10		Monroe; 10"	
	Western	1976	0.26			Ont. M. Agr. Food., 1976 in Konasewich et al., 1978
	Western	1978	0.59	0.17 - 1.54	muscle; Pelée Island	Rees, et al., 1979
	Central	1971	1.6	0.9 - 2.2	headless and eviscerated	Frank et al., 1978
	Central	1971	1.6			MOE, 1976 (unpubl.) in Konasewich et al., 1978
			1.5			
	Central	1972	0.96			MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Central	1976	0.1			Ont. M. Agr. Food in Konasewich et al., 1978
	Central	1978	0.79	0.39 - 1.39	muscle	Rees, et al., 1979
	Eastern	1968	< 0.1		headless and eviscerated	Frank et al., 1978
	Eastern	1970-1971	2.1	1.4 - 4.3		MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Eastern	1971	0.8	0.5 - 1.4	headless and eviscerated	Frank et al., 1978
	Eastern	1972	5.6			MOE, 1976 (unpubl.) in Konasewich et al., 1978
			1.0	0.6 - 1.7	Long Pt. Bay	
	Eastern	1972	1.0	0.5 - 5.4	headless and eviscerated	Frank et al., 1978
White crappie	Eastern	1976	N.D.		muscle	Crawford & Brunato, 1978
White sucker	Central	1969	2.5			MOE, 1976 (unpubl.) in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
PCB's						
White sucker	Central	1976	0.05		fillet	Penn. D.N.R. 1977 in Konasewich et al., 1978
	Eastern	1976	N.D.		muscle	Crawford & Brunato, 1978
Yellow perch	Whole lake	1976	0.18		edible portion	Penn. D.N.R., 1977 in Konasewich et al., 1978
	Whole lake	1978	0.22 ± 0.17		wet wt. (whole fish)	Great Lakes Water Quality, 1979, App. B
	Western	1968	<0.1		headless and eviscerated	Frank et al., 1978
	Western	1971	1.0	0.2 - 2.6	headless and eviscerated	Frank et al., 1978
	Western	1971	0.96			MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Western	1974	0.03			Mich. D.N.R., 1976 (unpubl.) in Konasewich et al., 1978
	Western	1975	0.6	0.4 - 0.9	headless and eviscerated (59 fish in composite)	Frank et al., 1978
	Western	1976	0.58 0.6			Ont. M. Agr. Food, 1976 in Konasewich et al., 1978
	Western	1977	0.44 0.35 0.25 1.11		whole body whole body fillet carcass	Herdendorf et al., 1978 in Konasewich et al., 1978
	Central	1969	2.3			MOE, 1976 (unpubl.) in Konasewich et al., 1978

Fish	Basin	Year	Level	Range	Remarks	References
PCB's						
Yellow perch	Central	1971	0.34 0.65			MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Central	1971	0.3	0.2 - 0.6	headless and eviscerated	Frank et al., 1978
	Central	1972	0.25			MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Central	1975	0.2	<0.1 - 0.8	headless and eviscerated (70 fish in composite)	Frank et al., 1978
	Central	1976	0.29 0.23			Ont. M. Agr. Food, 1976 in Konasewich et al., 1978
	Eastern	1968	<0.1	<0.1 - 0.1	headless and eviscerated	Frank et al., 1978
	Eastern	1971	0.6	0.3 - 1.0	headless and eviscerated	Frank et al., 1978
	Eastern	1970-1971	0.8	0.2 - 2.4		MOE, 1976 (unpubl.) in Konasewich et al., 1978
	Eastern	1972	0.25	0.18 - 0.33		MOE, 1976 (unpubl.) in Konsewich et al., 1978
	Eastern	1972	0.3	0.1 - 0.4	headless and eviscerated	Frank et al., 1978
	Eastern	1975	0.1	0.1 - 0.8	headless and eviscerated (26 fish in composite)	Frank et al., 1978
	Eastern	1976	0.2	<0.1 - 0.8	headless and eviscerated	Frank et al., 1978
	Eastern	1976	TR.	N.D. - 0.001	muscle	Crawford & Brunato, 1978
	Eastern	1978	0.36 0.13 0.02	0.055 - 0.52 N.D. - 0.45 N.D. - 0.05	muscle; Port Dover muscle; Long Pt. Bay whole fish; Long Pt. Bay	Rees, et al., 1979

Appendix 5

Results of toxicity experiments
found in the literature

Appendix 5b

Organism	Type type	PCB's isomer	PCB's level	oil and grease type	oil and grease level	Remarks	References
Chlamydomonas sp.	significant inhibition of growth	1248	111 ppb			algae isolated from L. Michigan 25 day exposure	Christensen & Zieski, 1980
Chlorella vulgaris	decreased growth rate			furnace oil Assam crude UAE crude Bombay high crude	2 ppb 5 ppb 5 ppb 30 ppb	furnace oil most toxic 12 day exposure % reduction varies	Gaur & Kumar, 1981
Selenastrum capricornutum	decreased growth rate			furance oil ASSAM crude UAE crude Bombay high crude	2.0 ppb 5 ppb 5 ppb 5 ppb	furnace oil most toxic 12 day exposure % reduction varies	Gaur & Kumar, 1981
Scenedesmus quadricauda	reduced growth reduced growth			diesel oil lubricating oil	<0.1% by volume 10% by volume	12 day exposure 18°C Euglena not affected by either oil at levels tested	Dennington et al., 1975
Euglena gracilis	48 hr ID-50 48 hr ID-50	AROCLOR 1221 4.4 ppm AROCLOR 1232 55 ppm				25°C	Ewald et al., 1976
Daphnia pulex	48 hr LC50			Naphthalene Phenanthrene	3.4 ppm 1.14 ppm	O_2 consumption and Filtering rate also considered	Geiger & Buikema, 1981

Appendix 5a. continued

Organism	Type	As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Remarks	Reference
Dugesia sp.	96 hr LC50		4.9±0.25 ppm							as Cd Cl_2	Fennikoh et al.,
Hyalella sp.	96 hr LC50		85±10 ppb							not fed 48 hr prior to exp't T^oC=23^oC; hardness=20 mg/l as $CaCO_3$	1978
fish - several sp.	growth retarded or death		4-12 ppb							embryos & larvae larvae more sensitive than embryos 30-60 day exposure	Eaton et al., 1978
Lepomis gibbosus	24 hr LC50		2.8 ppm	19.1 ppm	3.5 ppm		0.41 ppm	16.4 ppm	25.1ppm	T^oC = 28^oC	Rehwoldt et. al.,
	48 hr LC50		2.2 ppm	17.8 ppm	2.9 ppm		0.39 ppm	12.1 ppm	21.0ppm	hardness = 55 mg/l	1972
	96 hr LC50		1.5 ppm	17.0 ppm	2.7 ppm		0.30 ppm	8.0 ppm	20.1ppm	D.O. 6.9 mg/l	

Appendix 5a. continued

Organism	Type	As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Remarks	Reference
Ephemerella grandis	14 day LC50				0.18-0.2 ppm	3.5 ppm			<9.2 ppm	DO_2 = 7-12 mg/l conductivity 130-340 umhos/cm alk. = 30-70 mg/l ($CaCo_3$) flow through experiment	Nehring, 1976
Nais sp.	LC50		4.6-1.7 ppm	12.1-9.3 ppm	2.3-0.9 ppm		1.9-1.0 ppm	16.2-14.1 ppm	21.2 -18.4 ppm	(24-96 hr)	Rehwoldt *et al.*, 1973
Gammarus sp.	LC50		0.14-0.07 ppm	6.4-3.2 ppm	1.2-0.91 ppm		0.09-0.01 ppm	15.2-13.0 ppm	10.2 -8.1 ppm	Temp. = 17°C	
Chironomus sp.	LC50		5.1-1.2 ppm	16.5-11.0 ppm	0.65-0.03 ppm		0.06-0.02 ppm	10.2-8.6 ppm	21.5- 18.2 ppm	Hardness = 50 mg/l	
Amnicola sp.	LC50(adults)		10.1-8.4 ppm	10.2-8.4 ppm	1.5-0.9 ppm		1.1-0.08 ppm	21.2-14.3 ppm	16.8- -14.0 ppm	pH = 7.6	
	LC50		5.1-3.8 ppm	15.2-12.4 ppm	4.5-9.3 ppm		6.3-2.1 ppm	26.0-11.4 ppm	28.1- 20.3 ppm	D.O. = 6.2 mg/l	
Dugesia dorotocephala	5 or 10 day LC50						<0.5 ppm			mercury as methyl mercuric chloride animals collected near Fort Collins, Co. T°C = 22°C	Best *et al.*, 1981
Chironomus tendipes	48 hr LC50		as Cd Cl_2 25 ppm			as $PbNO_3$ 50 ppm	as $HgCl_2$ 64 ppm		as $ZnSO_4$ 62.5 ppm	laboratory exp't pH = 7; D.O. = 5 mg/l Alk. = 216 mg/l Hg est. at [ ] used; 100% mortality in 24 hrs.	Rao & Saxena, 1981

Appendix 5a. continued

Organism	Type	As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Remarks	Reference
Daphnia pulex	increase in O_2 demand		50 ppb							experimental pond used at U.of Kansas cultured animals	Kettle *et al.*, 1980
Physa gyrina	24 hr LC50		7.6±0.91 ppm							adults	Wier & Walter, 1976
	48 hr LC50		4.25±0.19 ppm								
	96 hr LC50		1.37±0.21 ppm								
	228 hr LC50		0.83±0.19 ppm								
Physa gyrina	48 hr LC50		0.69±0.25 ppm							immature snails	Wier & Walter, 1976
	96 hr LC50		0.41±0.15 ppm								
Physa integra	7 day LC50		114 ppb							hardness 45 mg/l T°C 10-15°C	Spehar *et al.*, 1978
	28 day LC50		10.4 ppb								
Ephemerella sp.	28 day LC50		<3.0 ppb								Spehar *et al.*, 1978
Gammarus pseudolimnaeus	4 day LC50					124 ppb					Spehar *et al.*, 1978
	28 day LC50					28.4 ppb					
Tubificids (mixed pop.)	24 hr LC50					27.5 ppm				pH 8.5	Whitley, 1967
	24 hr LC50					49 ppm				pH 6.5	
	24 hr LC50								46 ppm	pH 7.5	

Appendix 5a. continued

Organism	Type	As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Remarks	Reference
Daphnia galeata mendotae	50% reductions in carrying capacity		7.7 ppb							L. Michigan water 22 week exposure	Marshall, 1978
	1% reduction in carrying capacity		0.15 ppb								
	LC50		5 ppb								
Daphnia magna	72 hr LC50				86.5 ppb					newborn daphnids	Winner & Farrell,
Daphnia pulex	72 hr LC50				86 ppb					Alk.=100-119 mg/l	1976;
Daphnia parvula	72 hr LC50				72 ppb					DO=8.7-11.4 mg/l	Winner, 1976
Daphnia ambigua	72 hr LC50				67.7 ppb					pH = 8.2-9.5 T°C = 20°C $CuSO_4.5H_2O$ used in algae	
Daphnia magna	72 hr LC50				13.5±2.72 ppm					unaerated, filtered Standard (ph 6.8-7.9)	Winner, 1976
	72 hr LC50				25.2±0.42 ppm					unaerated, unfiltered Standard	
	72 hr LC50				31.6 ppm					aerated, unfiltered Standard	
	72 hr LC50				78.2±16.6 ppm					Medium pH=8.2-9.5, Alk. =110-120;	
	72 hr LC50				85.1±3.63 ppm					Pond pH=8.2-9.5, Alk. = 100-118	
Daphnia ambigua	72 hr LC50			1.7±1.2 ppm	12.4±1.46 ppm					unaerated filtered Standard	Winner, 1976
Daphnia magna	72 hr LC50			5.2±2.8 ppm	13.5±2.72 ppm					pH=6.7-7.9	
Daphnia magna	72 hr LC50			42.1±8.49 ppm						in pond water	Winner, 1976
Daphnia ambigua	72 hr LC50			7.7±1.34 ppm							
Daphnia galeata	72 hr LC50			65.6±1.75 ppm							
Daphnia pulicara	72 hr LC50			110.8±30.1ppm							

Appendix 5a. continued

Organism	Type	As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Remarks	Reference
Daphnia pulex	48 hr LC50		0.14 ppm				3.0 ppb			metals-Cd nitrate	Canton & Adema,
Daphnia magna	48 hr LC50		0.047 ppm				≈ 2.2 ppb			Hg(II) chloride	1978
Daphnia cucullata	48 hr LC50		0.2 ppm				3.2 ppb				
Daphnia hyalina	48 hr LC50		55 ppb	22 ppb	5ppb	0.6 ppm	5.5 ppb	1.9 ppm	40 ppb	plankton collected	Baudouin &
	95% C.I.		9.5-3.2	32-15	6-4	0.89-0.41	9.8-3.1	2.48-1.45	51-31	(Italy)	Scoppa, 1974
Cyclops	48 hr LC50		3.8 ppm	10 ppm	2.5 ppm	5.5 ppm	2.2 ppm	15 ppm	5.5 ppm	metals as $CdSO_4.8H_2O$;	
abyssorum	95% C.I.		6.3-2.3	12.0-8.0	3.0-2.1	7.7-4.0	3.3-1.5	25.5-8.8	6.8-4.5	K_2CrO_4; $CuCl_2.2H_2O$;	
Eudiaptomus	48 hr LC50		0.55 ppm	10.1 ppm	0.5 ppm	4.0 ppm	0.85 ppm	3.6 ppm	0.5 ppm	$Pb(CH_3OO)_2.2H_2O$; HgCl;	
padanus	95% C.I.		0.77-0.39	11.9-8.6	0.65-0.38	6.4-2.5	1.02-0.71	4.6-2.8	0.72-0.35	$NiCl_2$; $ZnSO_4.7H_2O$, pH = 7.2; T°C = 10°C	
Daphnia galeata mendotae	48 hr LC50		40 ppb							T°C = 17-21°C	Marshall, 1979
	96 hr LC50		30 ppb							L. Michigan water used (few *in situ* experiments)	
Daphnia galeata mendotae	96 hr LC50 (10°C)		74 ppb			714 ppb				hardness 130-140 mg/l as $CaCO_3$	Wilson, 1980
Daphnia magna	96 hr LC50 (10°C)		270 ppb			998 ppb				ph = 8-8.2	
Cyclops bicuspidatus thomasi	96 hr LC50 (20°C)		441 ppb for males			1.135 ppm				Cd as $Cd(NO_3).4H_2O$	
	96 LC50		319 ppb for females							Pb as $Pb(NO_3)_2$	
Diaptomus sicilis	96 hr LC50 males (5°)					275 ppb					
	96 hr LC50 5°C females		86 ppb			460 ppb					
Cyclops sp.	96 hr LC50		340±320 ppb							as $CdCl_2$ T°C = 23°C not fed 48 hrs prior to exp't hardness = 20 mg/l as $CaCO_3$	Fennikoh *et al.*, 1978

Appendix 5a. continued

Organism	Type	As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Remarks	Reference
Scenedesmus quadricauda	EC-50				0.1 ppb				519.3 ppm	laboratory culture $ZnSO_4$ $CuSO_4$ EDTA in culture media.	Petersen, 1982
Chlorella sp.	EC-50								2.4± 0.02 ppm	laboratory cultures 96 hr. exp't	Rachlin & Farran 1974
Scenedesmus quadrciauda	significant growth reduction							0.1 ppm		laboratory tests Ni as $NiNO_3$	Spencer & Greene, 1981
Ankistrodesmus falcatus								0.1 ppm		14 day biomass $T^oC=20^oC$;	
Anabaena flos-aquae								0.6 ppm		16 hr light pH = 7.2	
Selenastrum capricornutum	initial growth rate reduction		50 ppb		50 ppb				30 ppb	pH 7.1-7.2 laboratory exp't	Bartlett et al., 1974
	complete inhibition		80 ppb		90 ppb				120 ppb	-cultures used	
	algicidal		650 ppb		300 ppb				700 ppb		
Selenastrum capricornutum	24 hr lag growth		50 ppb		50+60 ppb				30-40 ppb		Bartlett et al., 1974
	48 hr lag growth				70 ppb				50+60 ppb		
	72 hr lag growth		60 ppb								
Chlamydomonas sp.	inhibits growth	74.9 ppb								algae isolated from L. Michigan 25 day exposure	Christensen & Zielski, 1980
Daphnia pulex	72 hr LC50		62(54-72)ppb							fed Chlorella and yeast daily	Bertram & Hart, 1979
	96 hr LC50		47±5 ppb								

Appendix 5a.

Organism	Type	As	Cd	Cr	Cu	Pb	Hg	Ni	Zn	Remarks	Reference
mixed bacteria culture	resazurin IC 50	25 ppm								Na arsenite, Na arsenate, Na cacodylate } 2 hr. exp't	Anderson et al., 1980 in Liu, 1981
	resazurin IC 50	100 ppm									
	resazurin IC 50	2000 ppm									
bacteria	resazurin IC 06						0.1 ppm			8 day test	Liu, 1981
	resazurin IC 67						1.0 ppm				
	resazurin IC 90						5.0 ppm				
mixed natural zooplankton pop.	pop. decrease		5.0 ppb							L Michigan samples	Marshall & Mellinger, 1980
	general EC 50		3.5 ppb							field exp't 21 day exposure	
Asterionella formosa	18% reduction in growth		1.9 ppb							$CdCl_2$ used	Conway & Williams, 1979
	25% reduction in growth		4.1 ppb								
	46% reduction in growth		8.8 ppb								
Aquatic plants (several sp.)	growth inhibition			10.0 ppm						algal sp. collected mostly from Susquehanna R. (Penn.-N.Y.) - amount of growth inhibition varied from plant to plant	Mangi et al., 1978