OFFPRINTS FROM: TOXIC CONTAMINANTS AND ECOSYSTEM HEALTH: A GREAT LAKES FOCUS
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BENTHIC INVERTEBRATES AND CONTAMINANT LEVELS IN THE GREAT LAKES: EFFECTS, FATES, AND ROLE IN CYCLING

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1. INTRODUCTION

Most persistent toxic contaminants have a strong affinity for particulate material and, over time, become associated with the bottom sediments. As a result, benthic organisms play an integral role in the effects, fates, and cycling of contaminants throughout the rest of the aquatic system. In general, studies of interactions between the benthos and toxic contaminants can be divided into two categories. The first category examines the response of benthic organisms to changes in contaminant levels. Benthic response may be examined in the field by assessing changes in benthic community structure, or in the laboratory by using acute and chronic toxicity tests. The other category concerns the effect of benthic-mediated processes on contaminant cycling and distribution. Examples of such processes are bioaccumulation, bioturbation, and trophic transfer.

This review will focus on known interactions between benthic organisms and contaminants in the Great Lakes. Benthic organisms are here defined as those invertebrates retained by a 0.5- to 0.6-mm screen, or those commonly referred to as the "macrobenthos." Although other benthic groups, such as bacteria and meiobenthos, are also important in contaminant fates and effects, they will not be considered in this review. Since benthic groups occurring in the Great Lakes are not unique to this system, relevant studies from other freshwater environments will also be presented.

2. COMPONENTS OF GREAT LAKES BENTHOS

The typical benthic community in the Great Lakes, excluding the connecting channels and shallow bays and basins, might best be described in terms of broadly defined habitat zones based on gradients in water depth (see Cook and Johnson, 1974, for review). Two important depth-related factors affecting benthic distributions are sediment type and temperature regime. The benthos of the littoral zone (0–30 m) is characterized by low abundances, high diversity, and great variability in both time and space. This variability is a reflection of the physical instability of this zone, particularly at the shallow end of this depth range. Downwellings and oscillating thermoclines cause wide fluctuations in bottom temperatures, and waves and bottom currents cause resuspension of bottom substrates. Organisms with broad environmental tolerances or life cycles suited to these changing conditions do best in this zone; tubificids, naidids, and chironomids are often the dominant forms. As water depth increases within the littoral zone, the bottom environment becomes increasingly stable and the amphipod *Pontoporeia hoyi* and the oligochaete *Stylodrilus heringianus*, both obligate stenotherms, gradually increase in abundance.

In the sublittoral zone (30–70 m), temperatures fluctuate less and the sediments are less influenced by storms and currents. Suspended particles from the littoral begin to settle, providing increased food resources for the benthos. Fewer species are found in this zone than in the littoral, but total biomass reaches a maximum.

Tubificids and sphaeriid clams are abundant, and *P. hoyi* and *S. heringianus* are found at maximum densities.

The profundal (>70 m) is a stable region of uniformly low temperatures, low light regimes, and unconsolidated sediments. *P. hoyi* is clearly the most abundant form in this zone, although the semiplanktonic *Mysis relicta*, because of its large size, is also important when biomass is considered. *S. heringianus* is the dominant oligochaete. Both sphaeriid clams and chironomids are found in the profundal, but abundances are generally low. As depth increases in the profundal, all benthic forms except *Mysis* decrease in abundance. Maximum densities of *Mysis* occur at depths greater than 100 m (Carpenter et al., 1974).

In shallow bays and basins (Green Bay, Saginaw Bay, western Lake Erie), in the connecting channels (St. Mary's River, St. Clair River, Detroit River), and in Lake St. Clair, other benthic forms may also become abundant. The most significant of these are the mayfly larvae *Hexagenia* and unionid bivalves (mussels). *Hexagenia* was once very abundant in the bays and shallow basins. The decline of this species has been well documented and generally attributed to severe oxygen depletion in bottom waters (Britt, 1955; Howmiller and Beeton, 1971; Schneider et al., 1969; Carr and Hiltunen, 1965). In western Lake Erie, *Hexagenia* populations were reduced by 90% after a prolonged period of thermal stratification that resulted in decreased oxygen levels (Britt, 1955). *Hexagenia* is also sensitive to environmental pollutants (Fremling, 1970) and numbers presently remain low in these areas, although recent evidence indicates populations may be increasing in the Detroit River and parts of western Lake Erie (Thornley, 1985).

Mussels are found mostly in the Huron–Erie corridor (St. Clair River, Lake St. Clair, Detroit River) and in the western basin of Lake Erie. This group has generally been considered sensitive to pollutants and, at least in western Lake Erie, abundances have apparently declined over the past few decades (Mackie et al., 1980) and existing populations have showed signs of senescence (Roth and Mozley, 1973). Since mussels are dependent upon fish hosts for the development of their glochidia, shifts in fish populations over the years also may have contributed to these changes.

3. EFFECTS OF CONTAMINANTS ON BENTHIC COMMUNITIES

3.1. Field Assessments

Whether or not toxic contaminants have altered the structure of Great Lakes benthic communities is difficult to assess. Unless concentrations of a particular contaminant are high and the resulting impact on the benthos is quite dramatic, changes in community structure resulting from chemical contaminants cannot be readily distinguished from other perturbations such as organic pollution, cultural eutrophication, or natural and human-induced ecological changes. For example, with an increase in eutrophication, increased nutrient levels and organic inputs to the bottom cause a shift in the community to species best suited to exploiting these

additional food levels. These "opportunistic" species tend to have a larger reproductive capacity and an inherent ability to tolerate changing environmental conditions. These species may also increase in abundance because of a decrease in competition from forms more sensitive to changing conditions. Because the same particles that transport organic material utilized as food also transport toxic contaminants, the relative impact of contaminants cannot be readily distinguished when examining only benthic distributions in the field. Both trophic contaminants (organic carbon, total phosphorus) and chemical contaminants may interact in defining benthic community structure (Lang and Lang-Dobler, 1979).

Throughout the Great Lakes, changes in the benthic community in response to polluting influences are relatively predictable. Species assemblages in polluted areas are distinctive and quite different from assemblages in clean areas. The response of Great Lakes benthos to changes in general pollution levels have been documented in studies of trends through time (Howmiller and Beeton, 1971; Schneider et al., 1969; Carr and Hiltunen, 1965; Robertson and Alley, 1966) and in studies measuring attenuation in space relative to a point discharge or river input (Brinkhurst, 1967; Vander Wal, 1977; Johnson and Brinkhurst, 1971; Nalepa and Thomas, 1976). Typically, tubificid oligochaetes increase in abundance relative to other forms and may achieve great densities, while *S. heringianus, P. hoyi,* and *Hexagenia,* if present initially, become rare and may eventually disappear.

A typical example of pollution-induced changes in the benthos in both time and space has been documented in Green Bay (Howmiller and Beeton, 1970, 1971). Green Bay is a relatively shallow arm of Lake Michigan that receives large amounts of both domestic and industrial wastes (primarily from pulp and papermills). Wastes enter the lower bay via the Fox River and as a result, the environmental quality of the lower bay is severely degraded (Beeton, 1978). The upper bay mixes with cleaner Lake Michigan water and is less influenced by the river. Based on benthic community changes, the quality of the entire bay declined between 1952 and 1969. During this period, the proportion of tubificid oligochaetes (mostly *Limnodrilus hoffmeisteri*) in the total number of benthic organisms increased from 66% to 85% in the lower bay and from 23% to 64% in the middle bay. The sensitive *S. heringianus* was abundant only in the upper bay, while amphipods and sphaeriids declined throughout the area and *Hexagenia* disappeared. In 1969, the area nearest the mouth of the river was devoid of all fauna.

Such evaluations of community changes are based on ecological observations and attributed to "pollution effects" without implying a specific cause and effect relationship. In areas where oxygen concentrations reach low levels [which is the case in many parts of Green Bay (Howmiller and Beeton, 1971)], the cause of community change is rather obvious. However, with toxic contaminants, the specific causative agent or agents are not readily apparent unless concentrations are high enough to entirely eliminate species from a given area. In the Great Lakes, this has been documented in a few instances and has occurred primarily in more shallow areas where the impacts are direct and most immediate. In Lake Superior, *P. hoyi* was not found in an area where concentrations of copper in the sediment exceeded 395 mg/kg (Kraft, 1979) and, in the same general area, *Hexagenia* was not found where copper concentrations exceeded 595 mg/kg (Malueg et al., 1984).

In the St. Mary's River, *Hexagenia* was absent or occurred in reduced numbers in sediments with visible signs of oil (Hiltunen and Schloesser, 1983). Also, in Hamilton Bay, Lake Ontario, oligochaetes and all other benthic forms were absent from sediments with an iron content over 25% (Johnson and Matheson, 1968). Tubificids (mostly *L. hoffmeisteri* and *Tubifex tubifex*) typically increased and *P. hoyi* decreased near a pulpmill effluent in Lake Superior (Vander Wal, 1977). These changes were related to increased organic carbon content of the sediments in areas near the pulpmill discharge. In the area nearest the discharge, tubificid abundances were reduced despite continued high levels of organic carbon. Apparently, some toxic chemical or chemical mixture associated with the high carbon levels was detrimental to the organisms.

The presence or absence of a particularly sensitive species may indeed be useful when assessing the impact of a toxic contaminant in the field. Yet, more often, the effects are not readily discernible from the many other environmental factors that can affect species distributions. For example, the Niagara River has been identified as the primary source of contaminants into Lake Ontario; high contaminant concentrations are found in the sediments east of the river mouth along the southern shoreline (Thomas, 1983). In this area, *P. hoyi* is rare or absent at depths where it is normally found (Nalepa and Thomas, 1976). High contaminant levels may indeed be adversely affecting *P. hoyi,* but low Eh values also characterize the sediments of this area (Thomas et al., 1972), and water temperatures here are higher than in other shoreline areas of Lake Ontario because of frequent downwellings (Lee, 1972). Both of these physical features may also be limiting *P. hoyi* populations.

Various statistical techniques, such as discriminant analysis, principal component analysis, and cluster analyses, may be helpful in defining the relevant physicochemical variables affecting benthic populations (Green and Vascotto, 1978). Thus far, however, these techniques have not been widely used in the Great Lakes. Although useful, the results of such techniques should be interpreted with caution when assessing the impact of toxic contaminants. In the Duluth basin of western Lake Superior, low benthic abundances were attributed to either high metal levels in the sediments or the heavy silt loads in that area (Cook, 1975). A subsequent study used canonical correlation analyses to determine the important factors affecting benthic distributions (Dermott, 1978). Both sphaeriids and *S. heringianus* were negatively associated with levels of Zn in the sediments. Yet, although an overall negative relationship was found, site-specific discrepancies were apparent. Sphaeriids were more abundant in Thunder Bay than in the Duluth basin, even though Zn levels were significantly greater in the former area. Many factors can affect the toxic impact of heavy metals. For instance, the organic fraction of sediments binds heavy metals and reduces their toxicity (Lewis et al., 1973).

3.2. Laboratory Assessments and Sublethal Effects

Because of the difficulties associated with field assessments, direct cause and effect relationships are commonly examined using acute toxicity bioassays under controlled laboratory conditions. While bioassays have been useful in establishing

water quality criteria and standards for effluents, the results are difficult to apply to field situations because of the multiplicative effects of chemical mixtures, variations in bioavailability, species interactions, long-term low level exposures, and so on. These tests, however, have provided some insights into the complexity of assessing the impact of chemical contaminants on the benthos. The average LC_{50} value of 15 different chemicals to 12 different benthic species was determined and the species were ranked according to mean tolerance levels of all the compounds (Slooff, 1983). The rankings did not always follow traditional lines; that is, species considered generally tolerant to pollution (in the ecological sense of being found in polluted areas) were not necessarily ranked highest in terms of tolerance to the various chemical compounds. In other studies, the "pollution-tolerant" tubificids *L. hoffmeisteri* and *T. tubifex* were less tolerant of pentachlorophenol, black liquor, mercury, and cadmium than the "pollution-sensitive" *S. heringianus* (Chapman et al., 1982). Interestingly, the same study confirmed the relative tolerances of these species in the traditional sense of organic pollution and eutrophication; *L. hoffmeisteri* and *T. tubifex* were far more tolerant of sewage sludge and low oxygen levels than *S. heringianus*.

Several studies have examined the acute toxicity of Great Lakes harbor sediments and the adverse effects of these sediments on sensitive benthic species were readily apparent. *P. hoyi* and chironomids, when given a choice, preferred clean, unpolluted sediments to sediments from various polluted harbors (Gannon and Beeton, 1969). Also, in 48-hour toxicity tests, mortality for *P. hoyi* in harbor sediments was 70%, and mortality for chironomids was even higher. Prater and Anderson (1977) performed 96-hour toxicity tests with sediments from several areas in Duluth and Superior harbors. The sediments were classified as nonpolluted, moderately polluted, and highly polluted based on mortality of several aquatic organisms including *Hexagenia*. Although highest mortalities (10–50%) occurred in sediments with the greatest concentrations of inorganic and organic contaminants, mortality could not be related to any single compound. The assumption was that either synergistic or antagonistic effects had occurred, or chemical contaminants not measured led to the observed mortalities. Differences in bioavailability from the various sediments may also have contributed to the results. In similar studies on sediments from Lake Michigan harbors, mortality of *Hexagenia* was positively correlated with sediment concentrations of lead, iron, copper and chromium (Laskowski-Hoke and Prater, 1984) and to elutriate concentrations of chromium, cynanide, and zinc (Laskowski-Hoke and Prater, 1981).

In some cases, acute toxicity tests have been useful in interpreting benthic distributions in the field. In an Indiana lake, tubificids were found in sediments with concentrations of cadmium, zinc, and chromium (970, 1400, and 2100 μg/g, respectively) that eliminated most other forms (Wentsel et al., 1977a), while in an Idaho stream, oligochaete distributions were not affected by zinc concentrations of 1000–7000 μg/g (Funk et al., 1973). Oligochaete densities in both studies were highest in areas with the highest metal levels, and it was suggested that the elimination of more metal-sensitive predators-competitors such as chironomids may account for this distribution. Subsequent laboratory experiments have shown that

chironomids are very sensitive to certain heavy metals (Wentsel et al., 1977b, c). This may explain tubificid distributions in the Detroit River, where some of the highest densities of tubificids were found at stations with the highest heavy metal concentrations (particularly Zn) (Thornley and Hamdy, 1984).

Only one study in the Great Lakes has used acute toxicity assays to interpret benthic community response to a particular contaminant. In the Keewenaw Waterway, Lake Superior, benthic distributions were examined in relation to copper-contaminated sediments (Malueg et al., 1984). Chironomids accounted for 96% of all organisms found in an area with high copper concentrations ($\bar{x}$ = 589 μg/g), while they comprised only 25% in area with lower copper levels ($\bar{x}$ = 33 μg/g). *Hexagenia* was not found in the former area, but was the dominant organism in the latter. Acute toxicity tests on *Hexagenia* found no significant difference in mortality using sediments from the two areas. In similar tests using the more sensitive *Daphnia magna* as the test organism, mortality was greater in the sediments with higher copper levels. It was concluded that, although not acutely toxic, high copper levels had a subtle, long-term negative impact on *Hexagenia* populations.

As illustrated in the above study, sublethal levels of contaminants probably have more of an impact on benthic populations than acutely lethal levels simply because such levels occur more frequently and over a much broader area. Laboratory studies have shown that low concentrations of some contaminants have subtle effects on organism behavior and physiology. For instance, heavy metals affected the respiration rates of tubificids (Brkovic-Popovic and Popovic, 1977); phenols, chlorophenol, and styrene affected the swimming behavior of *P. hoyi* (Lindstrom and Lindstrom, 1980); and sediments with heavy metals were actively avoided by both tubificids (McMurtry, 1984) and chironomids (Wentsel et al., 1977b). Often, sublethal effects may not be recognized without employing special techniques. Histopathological studies revealed that low cadmium levels caused rupturing of the body wall and degeneration of amoebocytes in polychaetes (Thompson et al., 1982).

The effects of low, sublethal levels of contaminants are likely to be greatest in depositional areas where contaminants tend to accumulate. However, as noted earlier, sublethal effects are difficult to assess since organic inputs in these areas also are high. Although there have been no studies designed to evaluate the effects of low-level, long-term exposures of contaminants on Great Lakes benthic communities, there is some evidence that such effects may exist. For example, PCB concentrations of over 50 ng/g caused *S. heringianus* to terminate feeding after 58 days (White and Klahr, 1982). This concentration is below the average PCB levels of 85.0 and 115.0 ng/g found in the depositional basins of Lake Ontario and Lake Erie, and below the maximum levels of 75.0 ng/g found in southern Lake Michigan. Also, zinc concentrations of between 58.5 and 123.5 μg/g significantly lowered swimming activity in *Pontoporeia* (Magnuson et al., 1976). Most depositional areas of Lake Michigan have zinc concentrations of over 60 μg/g and often higher than 200 μg/g (Cahill, 1981). If and how such sublethal effects become evident at the population level remains unclear. For instance, despite these seemingly high PCB and zinc levels in southern Lake Michigan, populations of both *S.*

heringianus and *Pontoporeia* have increased or remained stable over the years (Nalepa, 1987).

Morphological deformities in organisms occurring in polluted areas may be indicative of sublethal effects. Deformed setae have been noted in oligochaetes from areas with high mercury levels (Milbrink, 1983) and in areas with high concentrations of zinc and copper (Chapman and Brinkhurst, 1984). Deformities have also been noted in the mouthparts of chironomids occurring in areas contaminated with certain heavy metals (Wiederholm, 1984). In the Great Lakes, chironomid deformities have been reported in western Lake Erie near the mouths of the Detroit and Maumee Rivers (Brinkhurst et al., 1968), in the Bay of Quinte (Warwick, 1980), and in Georgian Bay (Hare and Carter, 1976).

4. BIOTURBATION AND CONTAMINANT CYCLING

Through their constant burrowing and feeding activities, benthic organisms physically rework and mix the upper layers of sediments. These activities, termed bioturbation, tend to increase the rate of material flux between the sediments and overlying waters and also homogenize many years (often decades) of accumulated sediments. The exchange process is mediated by diffusion or by physical transport. Numerous studies have demonstrated the impact of bioturbation on the direct diffusion of various nutrients from sediments to overlying waters (for reviews, see Petr, 1977; Fisher, 1982). However, considering the sorptive nature of most toxic contaminants, the importance of bioturbation in contaminant cycling lies mainly in the physical transport and resuspension of sediment particles. These processes reintroduce contaminants into the water column.

Each benthic species characteristically alters the physical and chemical nature of the sediments, depending on its particular feeding mode and life habit. Oligochaetes are probably the most important agents of physical transport. These organisms feed at depth and defecate at the sediment surface. Sediments are thus mixed down to the feeding depth, the exact location of which will vary depending on abiotic and biotic factors. In laboratory studies, the feeding depth of *S. heringianus* in Lake Michigan sediments was 4 cm (Krezoski et al., 1984b) and that of *T. tubifex* in Lake Erie sediments was 8 cm (McCall and Fisher, 1980). Field studies of the vertical distribution of cesium-137 and lead-210 have confirmed a zone of constant activity consistent with the vertical distribution of oligochaetes (Robbins, 1982). In addition, several studies in western Lake Erie have shown a mixed zone of heavy metals (Hg, Pb, Cr, and Zn) from 6–20 cm deep (Kemp et al., 1976; Wolery and Walters, 1974; Walters et al., 1974); this uniform distribution was attributed to the reworking activities of the benthos (Wolery and Walters, 1974).

The net effect of sediment reworking is to keep settled contaminants from being completely buried via sedimentation. Thus, even if a contaminant is no longer introduced from external loadings, it will not be readily lost from the system. The

reworking rate of oligochaetes (calculated from field densities and laboratory derived egestion rates) has been estimated at 4 times the sedimentation rate in Lake Huron (Krezoski et al., 1978) and over 10 times the sedimentation rate in parts of Lake Erie (Fisher et al., 1980). In addition, since oligochaetes selectively feed on the organic fraction of the sediments (Brinkhurst and Austin, 1979) and persistent contaminants are closely associated with this fraction, the net effect of these feeding activities is to transport contaminants preferentially back to the sediment surface. Release of contaminants from the sediments to the overlying waters was enhanced four- to sixfold in the presence of oligochaetes (Karickhoff and Morris, 1985).

The burrowing activities of oligochaetes increases sediment porosity and decreases tortuosity (Krezoski, 1981; McCall and Fisher, 1980). This is also true of other benthic forms. *P. hoyi* is a surface detritivore that randomly plows through the upper 1–2 cm of sediment. These activities mix the sediments in an "eddy diffusive" manner (Robbins et al., 1979). Similarly, sphaeriid clams live in the upper sediments where they filter feed on the surface or burrow to a depth of several millimeters. Many chironomids and *Hexagenia* live in distinct burrows and filter feed by drawing water in and through these constructions. *Hexagenia* burrows may extend down to 15 cm. Mussels burrow in the upper 2–3 cm and, because of their large size, may alter sediment structure as deep as 10 cm. In western Lake Erie, mussels are estimated to move 2–6% of the upper 10 cm of sediment every week; they also increase water content in the top 4 cm by 10–20% (McCall et al., 1979). This burrowing activity decreases sediment compaction which, in turn, enhances the probability of particle resuspension by physical processes (waves and currents). In addition, both *P. hoyi* and *M. relicta* make excursions into the water column and may resuspend particles directly. In the Baltic, a highly turbid zone extending 1–2 m above the bottom has been observed in areas with high densities of *Pontoporeia* (Ankar, 1977). The importance of sediment resuspension in contaminant cycling has recently been noted (Eadie et al., 1983a; Rosa, 1985). In Lake Michigan, about 14% of particle-associated PCBs in the water column during the winter could be accounted for by resuspension (Eadie et al., 1983a).

By physically mixing the sediments and thereby increasing the exchange of interstitial and overlying waters, the benthos have a profound impact on sediment redox potential and pH. These two parameters, in turn, influence the complexation of heavy metals, altering both bioavailability and toxicity. Free Cu and Zn ions are more toxic than their organically complexed forms, while complexation of Hg, Pb, and Cd enhances their uptake and toxicity. In a study to determine the effect of tubificids and mussels on mercury uptake by fish, mercury was placed at various depths in experimental sediment columns (Jernelov, 1970). In controls (without benthos) the methyl mercury content of fish decreased as the depth of the buried contaminated layer increased. In columns with benthos, the methyl mercury content of fish was higher than that of the controls and remained high until a critical depth was reached. The critical depth was 3 cm for tubificids and 9 cm for mussels. Bioturbation by these organisms likely enhanced mercury methylation by increasing sediment oxidation and microbial processes.

5. BIOACCUMULATION OF ORGANIC CONTAMINANTS

An important role of the benthos in contaminant cycling is their tendency to accumulate certain compounds from their immediate environment. Thus, contaminants become concentrated and available to higher trophic levels. Sources of contaminants to the benthos are the overlying waters, the porewaters, and sediment particles. The predominant route for bioaccumulation is considered to be uptake by passive diffusion from the aqueous phase, generally across the integument or respiratory surface (Neely et al., 1974). Absorption through the gut wall will occur when contaminants in the sediments or porewaters are ingested. With the sediments as a source, the contaminants are desorbed from particles as they pass through the digestive tract.

Bioaccumulation of contaminants in aquatic organisms is usually reported as the bioconcentration factor (BCF). This factor is the ratio, at steady state, between the concentration in the organism divided by the concentration in the water or sediment. Since uptake is by passive diffusion, the BCF of a particular organic contaminant is related to its octanol:water partition coefficient (K_{ow}) or its ability to partition into the organism. For instance, experiments to determine the uptake of various compounds from Lake Ontario sediments by tubificids showed that the BCF was low for compounds with a K_{ow} of $< 10^5$, increased to a peak for compounds with a K_{ow} of 10^5–10^6, and then decreased for compounds with a K_{ow} of $> 10^6$ (Fig. 4.1) (Oliver, 1984). It was hypothesized that, for compounds with low K_{ow} values, elimination rates by the worms are high and equilibrium is reached very quickly; BCFs are thus mainly governed by the lipophilicity (K_{ow}) of the compound. Compounds with intermediate K_{ow} values are eliminated more slowly, and will accumulate in the organisms over time. Compounds with high K_{ow} values are more strongly bound to the sediments and would therefore be less available to the worms. Also, the large molecular size of compounds with a high K_{ow} would

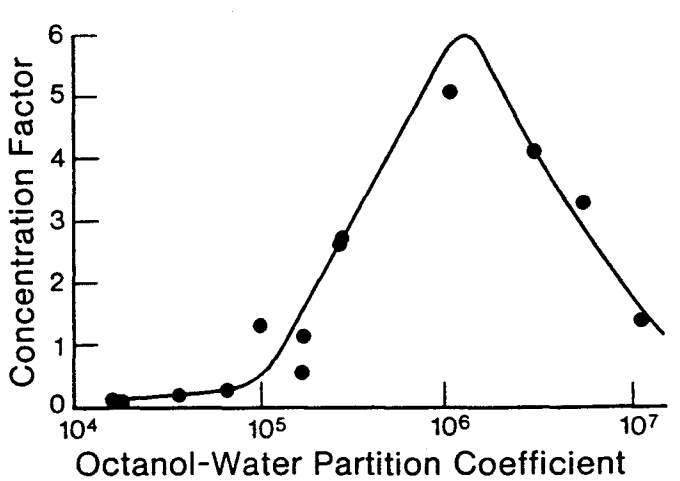

Figure 4.1. Worm concentration factor for various chemicals plotted against the chemical's octanol-water partition coefficient (curve fitted to points by eye). From Oliver (1984); with permission.

make chemical transport across worm membranes very difficult. A similar relationship between the K_{ow} of individual PAH compounds and the BCF was noted in *P. hoyi* (Eadie et al., 1983b).

Relatively high concentrations of organic contaminants have been found in Great Lakes benthos; bioaccumulation is evident in most cases although some variation is apparent (Table 4.1). For example, levels of PCBs in oligochaetes from the Detroit River were lower than levels found in the sediments (Smith et al., 1985), but oligochaetes from Lake Ontario had PCB levels that were 2–8 times the levels found in the sediments (Fox et al., 1983). Laboratory experiments have confirmed a BCF of 5 for PCB uptake by oligochaetes from Lake Ontario sediments (Oliver, 1984); similarly, a BCF of 4 was reported for PCB uptake by oligochaetes from Raisin River sediments (Mac et al., 1984).

Obviously, numerous variables may contribute to contaminant uptake by the benthos as related to both the physical nature of the environment and attributes of the organisms themselves. Probably the most important physical factor is the organic content of the sediments and/or the amount of dissolved organics found in the water. Through complexation or adsorption onto organic matter, the amount of contaminant available for biological uptake is dramatically reduced (Lynch and Johnson, 1982; Landrum et al., 1985). Laboratory experiments have shown that benthic organisms in highly organic sediments will have consistently lower BCFs than organisms in sediments with little organic content (Rubinstein et al., 1983; Muir et al., 1985).

Bioaccumulation in a given species is a function of its uptake rate, depuration rate and rate of biotransformation. For the Great Lakes, uptake rate experiments have focused primarily on the uptake of polynuclear aromatic hydrocarbons (PAH) from water by *P. hoyi, S. heringianus,* and *M. relicta* (Table 4.2). Uptake rate constants are highest in *P. hoyi,* and lowest in *M. relicta.* Since the uptake rate from water is directly related to the respiration rate (Murphy and Murphy, 1971; Frank et al., 1986), larger organisms with lower specific metabolic rates such as *Mysis* may be expected to have lower uptake rates. Also, external adsorption through the integument would be less significant in larger organisms since the surface area to volume ratio is lower than in smaller organisms.

The kinetic parameter that has perhaps the most control over the body burden is the conditional depuration rate constant. Depuration rates are estimated by placing exposed organisms into uncontaminated systems and measuring reduction in body burden. Depuration rates for *P. hoyi, S. heringianus* in sediments, and *M. relicta* in water are given in Table 4.3. The rates for *M. relicta* would likely be higher in the presence of uncontaminated food sources (Leversee et al., 1982; Landrum, 1982; Landrum and Scavia, 1983), but they do provide a basis for species comparisons. The conditional depuration rate constant was lowest for *P. hoyi* and highest for *S. heringianus.* This would seem to indicate that lipid levels play a major role in contaminant elimination; lipid levels are highest in *P. hoyi,* and lowest in *S. heringianus* (Gardner et al., 1985). Overall, depuration rates are lower in these three species than rates found in similarly sized organisms in warmer waters (Landrum and Scavia, 1983; Frez and Landrum, 1986; Bahner et al., 1977).

Table 4.1 Levels of Organic Contaminants Found in Great Lakes Benthos[a]

	Organic Contaminant (ng/g dry wt)								
Organism	PCB	DDE	DDD	HEOD	Chlordane	HCB	OCS	Location	Reference
Oligochaetes	440 (0.7)	5.9 (4.9)	1.5 (0.7)		4.5 (1.6)	3.6 (1.2)		One site, Detroit River	Smith et al. (1985)
Oligochaetes	930–3650 (2.4–7.8)					105–735 (0.7–2.3)		4 sites, western Lake Ontario	Fox et al. (1983)
Pontoporeia hoyi	4215–11,000 (9.0–19.0)					230–895 (1.6–6.0)		4 sites, western Lake Ontario	Fox et al. (1983)
Benthos (mostly *P. hoyi*)	341 (4.0)	34 (4.3)	1.8 (1.2)	2.9 (3.2)				Lake Ontario near Rochester	Haile et al. (1975)
Pontoporeia hoyi	1305	154	96	141	252			1 site, southeastern Lake Michigan	Evans et al. (1982)
Pontoporeia hoyi	560	58	62	35				1 site, eastern Lake Erie	Whittle and Fitzsimons (1983)
Pontoporeia hoyi	1378	292		226				3 sites, western Lake Ontario	Whittle and Fitzsimons (1983)
Pontoporeia hoyi	1849	730		376				1 site, eastern Lake Ontario	Whittle and Fitzsimons (1983)
Unionid bivalves (mussels)	133						59	Huron–Erie Corridor	Pugsley et al. (1985)

[a]Number in parentheses is the bioconcentration factor (concentration in organism/concentration in sediment).

Table 4.2 Average Uptake Rate Constants of PAH for Three Great Lakes Benthic Invertebrates[a]

Organism	Compound	Uptake Rate Constant ($mL\ g^{-1}\ h^{-1}$)	Reference[b]
Stylodrilus heringianus	Anthracene	86 (10)	1
	Phenanthrene	91 (13)	1
	Pyrene	112 (1)	1
	Benzo[*a*]pyrene	85 (16)	1
Pontoporeia hoyi	Biphenyl	85 (23)	2
	Anthracene	126 (36)	3,4
	Phenanthrene	123 (37)	4
	Pyrene	208 (52)	2
	Benzo[*a*]pyrene	137 (7)	4
	Benzo[*a*]anthracene	148 (50)	2
Mysis relicta	Phenanthrene	32 (12)	4
	Anthracene	63 (18)	4
	Benzo[*a*]pyrene	112 (35)	4

[a]Constants are given as milliliters of water cleared of compound per gram wet weight of organism per hour. Number in parentheses is standard deviation.
[b]1, Frank et al. (1986); 2, Landrum (unpublished data); 3, Landrum (1982); 4, Frez and Landrum (1986).

Biotransformation has an impact on both the toxicity and overall fate of the contaminant. Although biotransformation has been noted in chironomids (Leversee et al., 1982), attempts to measure biotransformation of PAH in either *P. hoyi* or *S. heringianus* have not yielded any measurable rate (Landrum, 1982; Frank et al., 1986). Thus, rates of biotransformation by these organisms are very slow or

Table 4.3 Depuration Rate Constant of PAH for Three Great Lakes Benthic Invertebrates

Organism	Compound	Depuration Rate Constant[a] (h^{-1})	Reference[b]
Pontoporeia hoyi	Benzo[*a*]pyrene	0.0016 (0.0012)	1,2
	Anthracene	0.004 (0.003)	2
	Phenanthrene	0.004 (0.001)	2
Mysis relicta	Benzo[*a*]pyrene	0.013 (0.002)	2
	Anthracene	0.015 (0.005)	2
	Phenanthrene	0.012 (0.003)	2
Stylodrilus heringianus	Benzo[*a*]pyrene	0.012 (0.002)	3
	Anthracene	0.017 (0.004)	3
	Phenanthrene	0.018 (0.003)	3
	Pyrene	0.017 (0.001)	3

[a]Number in parentheses is standard deviation.
[b]1, Landrum et al. (1985); 2, Frez and Landrum (1986); 3, Frank et al. (1986).

nonexistent, which may also contribute to their relatively low depuration rate constants.

As previously noted, contaminants can be taken up directly from the water or from the sediments via the food pathway. For PAH uptake, models derived from field data indicate that the relative contribution of each source is dependent upon the relative concentration of contaminant, the physicochemical properties of the specific compound, and the life habit of the organism. When sediment PAH concentrations are low, both *P. hoyi* and oligochaetes obtain a substantial fraction of their PAH from water, but when sediment concentrations are high, much of their PAH is obtained from that source (Eadie et al., 1983b). Soluble PAH compounds such as phenanthrene, fluoranthene, pyrene, and chrysene are taken up to a greater extent from the water, while insoluble compounds such as benzo[*a*]pyrene (BaP) are taken up from the sediments. Laboratory-derived kinetics models of BaP predict that sediments contribute only 10–50% of the body burden in *P. hoyi* (Landrum et al., 1985), but 40–90% in *S. heringianus* (Frank et al., 1986). The reason, of course, is that *P. hoyi* is more readily found at the sediment surface and has more contact with the overlying waters than *S. heringianus*, which occurs deeper in the sediments. In terms of sediment contribution, the proportion of a contaminant passing through the gut that is actually incorporated into the body depends upon the feeding rate of the organism and the assimilation efficiency of the particular compound. Tubificids assimilated 20–30% of sediment-associated hexachlorobiphenyl (Krezoski et al., 1984a); this is considerably lower than the 90% assumed in some models (Thomann, 1981; Thomann and Connolly, 1984). Overall, the contribution of the food pathway to the total body burden of contaminants is an area that has not been systematically studied (see Biddinger and Gloss, 1984, for review).

6. BIOACCUMULATION OF INORGANIC CONTAMINANTS

The mechanism for the accumulation of heavy metals is markedly different from that of organic compounds. Most metals require an active or facilitated process for assimilation, while organic compounds accumulate by passive diffusion; the exception is organometalics such as methyl mercury which also accumulate via a passive mode. As a result, the accumulation of metals occurs mostly from the sediments through the food pathway. In addition, although free metals are readily available for uptake, they complex very strongly with the organic fraction in both sediments and water, significantly reducing their bioavailability. This complexation is much stronger for metals than for organic compounds.

Because many factors affect the complexation and thus the bioavailability of heavy metals (geochemical phases of the metal, organic content of sediment, Eh, pH), bioconcentration factors in the benthos tend to be highly variable and strongly influenced by local conditions. For instance, concentrations of nine heavy metals (Cu, Zn, Pb, Fe, Mn, Ni, Co, Hg) were examined in both tubificids and sediments

at two stations in the Frazer River system (Chapman et al., 1979a). BCFs were less than 0.6 for all metals except mercury, which had a BCF of 1.7 and 4.0 at the two stations. In contrast, another study in the same river system with the same benthic group found that, of the five metals examined, BCFs were greater than 1.0 for Cu, Zn, Pb, and Mn and less than 1.0 for only Fe (Bindra and Hall, 1978).

Few studies in the Great Lakes have examined metal levels in both the benthos and sediments. Engler (1979) noted that oligochaetes did not bioaccumulate Cd, Cu, Mn, Zn, or Fe from Lake Erie sediments while in Lake St. Clair, average levels of Cr and Pb in mussels were 28.6 and 1.4 times average levels found in the sediments (Pugsley et al., 1984). These latter bioconcentration factors were based on sediment samples taken down to 10 cm, which may not accurately reflect actual exposure concentrations. Bioconcentration factors for mussels in the Illinois River were less than 1.0 for 8 different heavy metals, including Cr and Pb (Mathis and Cummings, 1973).

7. TROPHIC TRANSFER

Benthic organisms are an important food item in the diet of most species of Great Lakes fish. The smaller forage species, such as alewife (*Alosa pseudoharengus*), smelt (*Osmerus mordax*), slimy sculpin (*Cottus cognatus*), deepwater sculpin (*Myoxocephalus thompsoni*), trout-perch (*Percopsis omiscomaycus*), and bloater (*Corgonus hoyi*), feed on both *P. hoyi* and *M. relicta* (Janssen and Brandt, 1980; Wells and Beeton, 1963; Heberger and House, 1974; Anderson and Smith, 1971; Wells, 1980). These fish, in turn, serve as food for the large piscivores. These valuable gamefish may also feed directly on benthic species during the early stages of their life cycles. Although laboratory experiments have indicated that fish may obtain most of their body burden of organic contaminants from their food (Macek and Korn, 1970; Weininger, 1978), little is known of the actual proportion attributable to benthic organisms. In comparing PCB concentrations of various size organisms ranging from zooplankton to lake trout (*Salvelinus namaycush*), *P. hoyi*, slimy sculpin, and lake trout had higher PCB concentrations than might be expected from a strict size-dependent relationship (Fig. 4.2) (Borgmann and Whittle, 1983). Lake trout feed on slimy sculpin, which, in turn, feeds heavily on *P. hoyi*. From a series of simulation models, Breck (1984) estimated that 40% of the body burden of PCBs in adult alewife came from *P. hoyi* and that 25% of the PCBs entering the entire alewife population (juvenile and adult) in Lake Michigan could be attributed to uptake from benthic organisms (*P. hoyi* and *M. relicta*).

Various species of ducks from the Great Lakes are also known to feed on the benthos including scaup (*Aythya affinis, Aythya marila*), goldeneyes (*Bucephala clangula*), and oldsquaw (*Clangula hyemalis*) (Rofritz, 1977; Hickey et al., 1966; Drobney et al., 1982). Although contaminant levels are high in waterfowl (Smith et al., 1985), sources of contaminants are difficult to establish because of their seasonal migrations and their great diversity of food items.

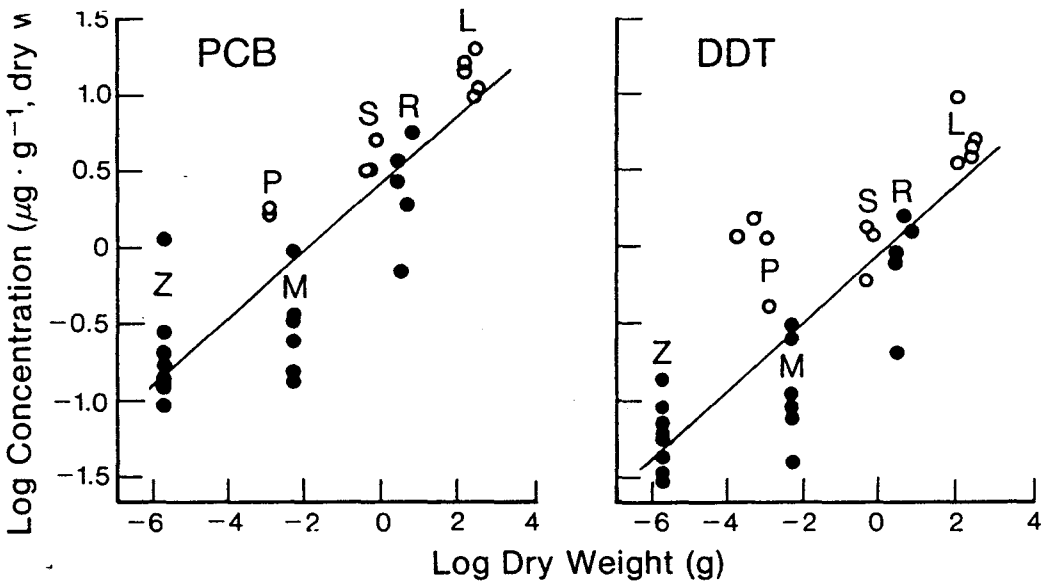

Figure 4.2. PCB and DDT concentrations as a function of body size in aquatic organisms from the eastern portion of Lake Ontario. Z, zooplankton; P, *Pontoporeia hoyi;* M, *Mysis relicta;* S, slimy sculpin; R, rainbow smelt; L, lake trout. From Borgmann and Whittle (1983); with permission.

8. BENTHIC ORGANISMS AS BIOMONITORING TOOLS

The use of benthic organisms to monitor both heavy metals and organic compounds in the environment has several advantages over simply measuring concentrations in the water or sediment: (1) Concentrations in the benthos provide an indication of the bioavailability of the particular contaminant. The major concern in the whole toxic contaminant issue is not whether contaminants are present, but whether they are available for bioaccumulation. (2) Many chemical and physical features of both water and sediments will affect the bioavailability of a given contaminant. Benthic organisms integrate contaminants from various sources forming a basis for assessing bioavailability. (3) Since most benthic communities are relatively stable in both time and space, contaminant levels in the benthos might be used in comparing concentrations at different sites, and also in comparing concentrations at the same site at different times.

Several benthic groups are potentially useful as biomonitors of contaminants in the Great Lakes. The most frequently analyzed groups have been oligochaetes and *P. hoyi* (Table 4.1). The advantages of using oligochaetes as a monitoring tool for heavy metals have recently been reviewed (Chapman et al., 1979b). Oligochaetes are the most widespread of the Great Lakes benthic groups and, because of the high tolerance levels of many oligochaete species, are often the only benthic group available for contaminant assessment in polluted areas. *P. hoyi* is the dominant species in the profundal waters of the Great Lakes where contaminants accumulate, is an important component in the diet of many fish, and tends to bioaccumulate organic contaminants to a greater extent than oligochaetes. In terms of assessing

trends in bioavailable contaminants, life habits of the individual taxonomic groups as well as varying tendencies to bioaccumulate contaminants should be considered. For example, concentrations of two contaminants (hexachlorobenzene and PCB) in amphipods (mostly *P. hoyi*) and oligochaetes were compared to corresponding concentrations in surficial (0–2 cm) Lake Ontario sediments (Fox et al., 1983). Concentrations in oligochaetes were directly related to concentrations in the sediments, but no correlation was observed for amphipods (Fig. 4.3). Oligochaetes integrate contaminants in the upper sediments and thus body burdens will reflect contaminant levels down to the depth of the feeding zone. *P. hoyi*, on the other hand, feeds mostly on particulates recently settled from the water column. These surface particles have contaminant concentrations several times greater than concentrations in sediments integrated over several centimeters (Eadie et al., 1983a). Also, *P. hoyi* tends to bioaccumulate contaminants to a greater extent than oligochaetes; thus, body burdens in *P. hoyi* will not reflect sediment concentrations as closely as body burdens in oligochaetes.

Unionid bivalves are potentially an ideal biomonitoring organism. Mussels filter large quantities of water and thus are exposed to large amounts of contaminants from both the water and from suspended particulates. They are relatively long-lived (10–20 years), which allows the sampling of more than one year class, and are large enough to provide adequate tissue for analysis with a minimum amount of sampling effort. In the Great Lakes, mussels were successfully used to delineate sources of an organic contaminant (octachlorostyrene) in the Huron–Erie corridor (Pugsley et al., 1985). The disadvantage of using mussels as in situ bioindicators is their limited distribution in the Great Lakes. As noted, mussels are found in

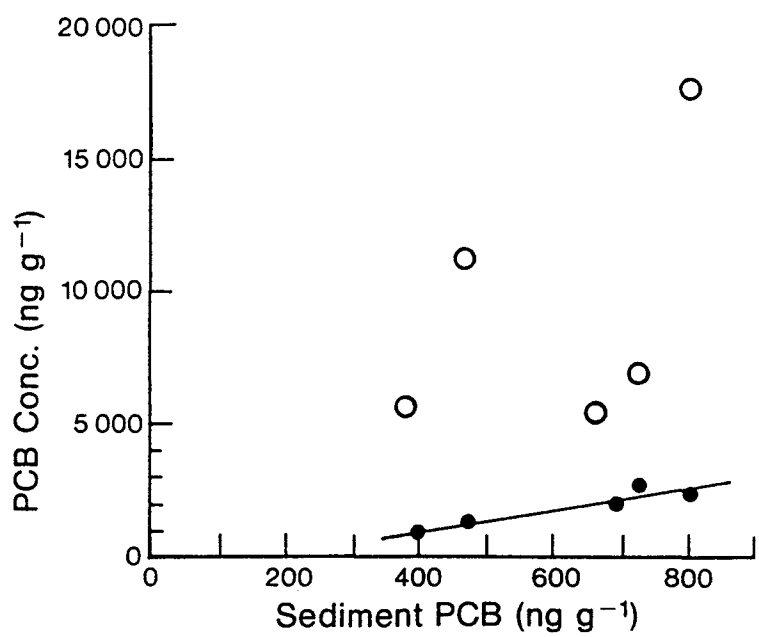

Figure 4.3. Concentrations (dry weight) of PCB in oligochaetes and amphipods as a function of the corresponding surficial sediment concentrations at five Lake Ontario sampling sites. ●, oligochaetes; $r = 0.96$. ○, amphipods, $r = 0.47$. Modified from Fox et al. (1983).

abundance only in the Huron–Erie corridor and in western Lake Erie. A possible means of circumventing this problem is to place uncontaminated mussels in cages that are anchored near suspected contaminant sources (Kauss and Hamdy, 1985). Mussels bioaccumulate contaminants very rapidly and maximum concentrations may be attained in 8–16 days (Kauss et al., 1983).

The mayfly nymph *Hexagenia* sp. may also prove to be a useful biomonitoring organism in certain areas of the Great Lakes. It is a rather large organism (up to 20 mg dry wt), is an important item in the diet of many fish, and ingests large amounts of sediment (Dermott, 1981). In the Great Lakes, contaminant levels in this genus have not been closely examined. In the upper Mississippi River, body burdens in *Hexagenia* were used as an indicator of the persistence and widespread distribution of PCB in that system (Clements and Kawatski, 1984).

Biological variables will also affect contaminant levels of benthic organisms. Contaminant uptake/retention by a particular organism will be affected by its lipid content, reproductive condition, age, and weight. For instance, seasonal changes in the body burden of BaP in *P. hoyi* varied by a factor of at least two without any change in contaminant load (Landrum et al., 1985). These changes were related to seasonal variations in lipid levels. As noted by Phillips (1978), the introduction of such biological variability necessitates the maintenance of rigid sampling and interpretative procedures.

9. SUMMARY AND RESEARCH NEEDS

Basically three approaches are available for studying the effects of pollutants on biota (Malins et al., 1984):

1. Effects can be inferred from field observations.
2. Effects can be derived from acute and chronic laboratory tests under controlled conditions.
3. Semicontrolled experiments can be conducted in the field in pens or other enclosures, or in model ecosystems under "natural" conditions.

In the Great Lakes, the first approach has been the most widely used method in assessing pollutant effects on the benthos. This approach has been successful in delineating specific zones of impact and in assessing the severity of the response. However, benthic populations are subject to wide fluctuations in both time and space and reflect the combined influences of all environmental variables, whether related to contaminant levels or not. This is especially true in nearshore and harbor areas where the impacts of contaminants are most severe. With this approach, unless a specific contaminant is present in unusually high concentrations, a definite cause-and-effect relationship cannot be established. Multivariate statistical techniques may be useful in discerning the most relevant factors affecting distributions (Dermott, 1978) but, unfortunately, few studies in the Great Lakes have simulta-

neously examined benthic distributions and levels of organic and inorganic pollutants.

The laboratory approach is useful in making relative toxicity comparisons of individual compounds or polluted sediments, and in evaluating tolerance levels of different benthic species. Only through the controlled conditions of the laboratory can the effects and fates of specific contaminants be separated from other environmental influences. However, the main limitation of laboratory studies is that they do not realistically simulate natural conditions. The complex physical, chemical, and biological factors that may affect a compound's toxicity and bioavailability in the field simply cannot be duplicated in the laboratory. Perhaps a more realistic assessment is to combine the two approaches, that is, to use laboratory toxicity tests in conjunction with detailed field observations. To date, only Malueg et al. (1984) have used both approaches to assess contaminant impact on Great Lakes benthic populations. Yet, this combined approach was successful only because one contaminant (copper) was at issue. Also of concern is the impact of interactions among low, sublethal levels of contaminants in the profundal areas of the Great Lakes. Such effects might best be examined by exposing natural populations (in native sediments) to low contaminant levels over long periods of time. Mesocosms, in conjunction with laboratory studies, may prove useful in this regard (Donaghay, 1984; McIntyre, 1977).

The ultimate goal of contaminant assessment is to develop predictive capabilities using effects models. Such models are still in the initial stages, yet inherent in

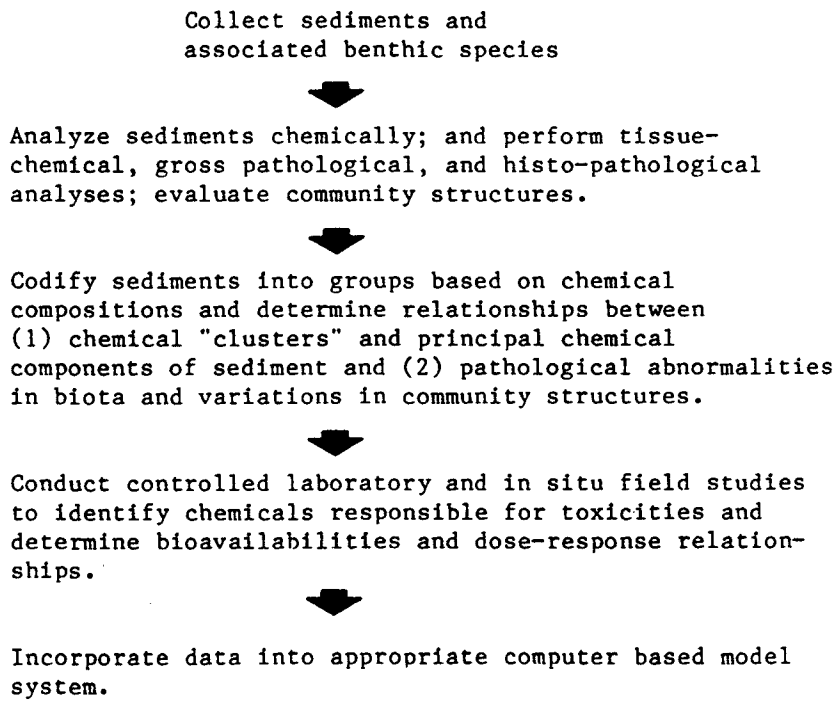

Figure 4.4. Protocol for relating sediment contaminants to biological effects. Modified from Malins et al. (1984).

this approach is a clear understanding of the fate of the contaminant in the organism, and also the environmental and physiological variables that may affect uptake and elimination. Kinetics and fates research should permit the estimation of an exposure scenario for the benthos and, by combining such a scenario with information on community structure and acute and chronic effects, a predictive model of contaminmant effects may be developed. A protocol for combining field studies, laboratory experiments, and model development to examine contaminant effects on the biota was given by Malins et al. (1984) and is illustrated in Fig. 4.4. The concepts presented might certainly be used as a initial guide in planning future efforts to assess contaminant effects on the Great Lakes benthos.

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