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Biological Considerations for Open Water Disposal of Dredged Material in the Great Lakes

P.G. Sly

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Abstract

A review of Great Lakes biological data describing benthic communities, habitat and life cycles and a review of the biological pathways and behaviour of particulate associated contaminants provide the basis for stating biological concerns about open water disposal of dredged material. With the exception of clean materials used for shore protection or enhancement, disposal should take place in deep water where biological productivity is least. The area of dump sites should be kept to a minimum, to affect the least amount of benthic habitat. Materials with biologically available contaminants should not be allowed to disperse during settlement, and if dumped in areas of naturally low sediment accumulation, they should be covered by a seal of inert material. It is assumed that the materials are not lethally toxic, that dumping does not occur at/or during biological events (e.g. spawning, migration, hatch and swim-up), and that dumped materials are not subsequently redistributed by wave or ice action. It is unlikely that concern about toxic contaminants will diminish in the future. The presence of macro- and micro-nutrients in disposal materials may present opportunities for ecological enhancement and will require site-specific evaluation. Coarse materials may be used to enhance some forms of microhabitat.

This contribution does not discuss the appropriateness of guidelines and standards which are presently applied to contaminant disposal. Although directly related to the biological concerns, the concentrations and forms of contaminants which may be included in open water disposal are separate topics.

Résumé

Les descriptions des communautés, de l'habitat et des cycles biologiques du benthos des Grand lacs ainsi que le cheminement et le comportement biologiques des con-taminants véhiculés par les particules sont à l'origine des inquiétudes d'ordre biologique au sujet de l'élimination de matériaux de dragage en eau libre. Sauf dans le cas des matériaux propres utilisés pour la protection ou l'embel-lissement des rivages, l'élimination devrait se faire en eau profonde, là où la productivité biologique est faible. La super-ficie des lieux d'immersion devrait être minimale afin de perturber le moins d'habitat benthique possible. Il ne faudrait pas permettre que des matériaux contenant des contaminants biologiquement actifs se dispersent au moment du dépôt, et, si leur immersion a lieu dans des secteurs où la sédimentation est faible, ils devraient être couverts par une couche de matériaux inertes. On suppose que les maté-riaux ne sont pas mortellement toxiques, qu'ils ne sont pas éliminés pendant ou durant des événements biologiques (c'est-à-dire le frai, la migration, l'éclosion et le stade d'alevin nageur), qu'ils ne seront pas redistribués par les vagues ou les glaces. Il est peu probable que les inquiétudes au sujet des contaminants toxiques s'estompent. La présence d'éléments nutritifs majeurs et mineurs dans les matériaux éliminés peut être l'occasion d'une amélioration écologique, qui exigera des évaluations ponctuelles des stations. Des matériaux grossiers peuvent servir à bonifier certains micro-habitats.

Cette étude ne fait pas l'examen de l'utilité des lignes directrices et des normes que l'on applique actuellement dans le domaine de l'élimination des contaminants. Quoique liées directement aux inquiétudes d'ordre biologique, les concentrations et les formes des contaminants que l'on pourrait trouver dans le secteur de l'élimination en eau libre demeurent des sujets d'un autre ordre.

Biological Considerations for Open Water Disposal of Dredged Material in the Great Lakes

INTRODUCTION

Under existing Great Lakes regulations the term "open water disposal" is generally taken to imply disposal anywhere in lake waters (bay and harbour-mouth or lagoonal embayment) and may, under some conditions, include interconnecting channels between the lakes. However, because limnological conditions which influence biological communities differ so greatly between shallow and deep water environments, it is essential to distinguish between them. A distinction can be made on the basis of environmental hydraulics, such as the boundary conditions which are characterized by the ability of surface wave action to rework the most easily erodable bed material (unconsolidated medium/fine sand). Short-term variability, of course, means that the depth of wave action changes greatly, and although wave field data may be derived from wind direction, duration and velocity, variations in shoreline geometry and underwater morphology make such computations complex and uncertain. On the other hand, bottom sediments reflect an integration of hydraulic conditions (Sly *et al.*, 1983), and the distributions of sand-size materials can be easily determined.

Using this approach, the boundary between shallow and deep water conditions occurs at about 20 m depth throughout much of the Great Lakes area. In the larger lakes, however, such as Superior, Recent medium/fine sands can be found at depths substantially greater than 20 m, whereas in the smaller lakes and embayments the depth may be much less than this. Even within the same lake, the boundary depth varies significantly because of differences in wind velocity and duration over different directions, and changes in fetch (Sly, 1978).

In shallow water areas, where wave-induced shear stress exceeds the critical shear stress of bed materials, most dumped materials will be reworked and selectively redistributed; and the greater the wave energy, the more rapid the process. Thus, it may be assumed that materials of a dominantly sandy composition will have little long-term impact on sandy receiving sites in shallow waters, because of their rapid dispersal. The presence of cohesive materials, however, will result in persistence of dumped materials for a greater length of time, because of their high critical shear stress. Cobble-gravel and other large blocks of

material will also tend to be more stable in high wave energy environments and are used to form artificial reefs or shore protection structures.

Deep water areas, which lie below the influences of wave action may be characterized by other, though less consistent, environmental conditions. Many deep water areas lie towards the base of the summer thermocline and beneath the depth of potential winter ice-scour. Deep water areas may be subject to periodic tilting of the thermocline and to bottom currents, but otherwise they are characterized by conditions in which there is a minimum of physical disturbance.

Most particulate contaminant loads are associated with the silt- and clay-size fractions and, since transport of these materials usually results in their eventual accumulation in deep water, emphasis in the following discussions draws particular attention to the biota of deep water areas. It is assumed, because of the potential for rapid reworking of contaminant materials and their effect upon the food web of the highly productive nearshore, that biologically available contaminants will not be disposed of in this zone. Also, disposal should be timed and located so that it will not affect important biological events, such as migration, spawning, hatch, and swim-up.

In the Great Lakes, most of the lake bed below 20 m is smooth and rather featureless (Sly, 1975) and the sediments are composed of fine sandy and silty-clay muds which become progressively finer in deeper water. Accumulation rates generally decrease away from shore and, in the Great Lakes, vary between a high of more than 2-3 mm/year and a minimum of less than 0.2 mm (Kemp and Harper, 1976; Sly, 1983a). Deep water disposal sites may be chosen over the outer shelf, slope or profundal areas of the lakes where subtle variations in habitat can be marked by significant changes in the composition of biological communities. Whereas the physical and chemical characteristics of the offshore environment are now reasonably well understood and can be quantified, the associations within biological communities are only incompletely understood and little can be quantified with certainty.

Using the lake bed for open water disposal has the potential to cause a number of different types of impact

Table 1. Sediment and abundance (n/m^2) data for major groups and species of benthic fauna in Lake Ontario (modified after Nalepa and Thomas, 1976; courtesy of J. Great Lakes Res.)

	Number of stations	Depth (m)	Particle size (phi)	Particle size (μm)	Total numbers of organisms	*Stylodrilus heringianus*	*Limnodrilus hoffmeisteri*	*Tubifex tubifex*	*Potamothrix vejdovski*	Other oligochaetes	Total Chironomidae	Total Pelecypoda	Total Gastropoda	Total Amphipoda
x̄	22	23	3.0	125	11 940*	2 660	2 800	780	1 310	710	140	970	150	2 160
S.D.		7.1	1.2		13 650*	3 190	6 610	2 150	3 430	1 110	290	1 290	390	2 740
x̄	13	65	4.7	40	7 000	650	600	3 430	0	50	20	110	0	2 140
S.D.		19	1.8		12 220	600	1 290	11 410	10	80	40	170	0	1 660
x̄	20	148	6.2	14	1 200	253	50	50	0	0	10	20	0	800
S.D.		35.8	1.5		810	180	60	110	10	10	10	60	0	630

* Revisions by T.F. Nalepa, November 1983.

which can be either detrimental or beneficial to the environment. Direct effects include smothering and bedform changes, and biota and contaminant introductions. The extent of the impacts depends upon the mobility of natural populations in the "target areas", life cycles of the community, biological pathways and the duration of exposure to the dumped material. In the following discussions on the impact of open water disposal, biological community data are drawn largely from Lake Ontario examples and additional disposal impact data have been drawn mostly from Lake Erie.

BENTHIC COMMUNITIES

Nalepa and Thomas (1976) sampled the benthos of Lake Ontario using a Ponar Grab and distinguished three broad sub-divisions in the lake, based upon a total of 55 sample stations. The shallow water benthos extended from a minimum sampling depth of 7 m to a maximum of about 35 m; an intermediate depth benthos occurred between 40 m and 90 m and the deep water benthos was restricted to depths in excess of 95 m. These depth range values are not rigid and, as noted by the authors, the shallow water benthos extended to greater depth along the north shore of the lake where the lake bed is subject to strong activity from current circulation and wave motion. The depth ranges of Nalepa and Thomas (1976) compare well with those reported by Kinney (1972) and with those of Sly (1983b; unpublished data). Table 1 summarizes the major groups and species which comprise the benthic fauna of Lake Ontario, and densities of organisms (n/m^2) are provided in the form of rounded mean and standard deviation values (S.D.), based on Nalepa and Thomas (1976). Precisely the same community structures, of course, do not exist throughout the Great Lakes, but the Lake Ontario assemblage may be taken as broadly characteristic of the lakes as a whole (Dermott, 1978; Freitag et al., 1976; Mozley and Alley, 1973; Powers and Robertson, 1965; Schuytema and Powers, 1966; Veal and Osmond, 1968).

Shallow water benthic communities are characterized by high diversity and large numbers of organisms. Intermediate depth samples are notable for a lack of certain oligochaete species and also for a lack of gastropods. With few exceptions, the numbers of individual organisms are substantially less than in shallow water. Most deep water sites are devoid of both pelecypods and gastropods; they have few chironomids and there is a shift in dominance from the oligochaete Tubifex tubifex to Stylodrilus heringianus. Mean sample densities of the total numbers of organisms decrease from $11940/m^2$ in shallow water to $7000/m^2$ at intermediate depth and to $1200/m^2$ in deep water communities. Amphipods are represented by two species; Gammarus fasciatus, which is a shallow water form

(Johnson and Brinkhurst, 1971a), and Pontoporeia affinis, which occurs throughout the lake basin and at all depths. Pontoporeia affinis is a dominant member of the benthic community and represents 22%, 61% and 58% of all organisms from shallow, intermediate and deep water sites (Nalepa and Thomas, 1976). Based on the author's records, isopods occur only in shallow water and no recoveries were made from depths in excess of 35 m.

Great lakes zooplankton include at least 30 species of crustaceans (Patalas, 1969; Watson, 1974) of which the opossum shrimp, Mysis relicta (Beeton, 1960; Carpenter et al., 1974), is the largest; this species, which is also considered to be part of the macrobenthos, is a migratory form that spends most of the daylight hours at, or slightly above, the lake bed and the dark hours at shallow depth (but usually not less than 20 m) in the water column. During the daytime, maximum densities occur at water depths of 125 m to 200 m; in Lake Ontario, the peak abundance (mean of August sampling at 30 sites, Carpenter et al., 1974) was $113/m^2$, and concentrations were restricted to the central parts of the lake. The much smaller copepod Limnocalanus macrurus is also migratory and characteristic of cold deep water habitat in the Great Lakes (Watson, 1974); early stages of its life cycle are spent in the bottom sediments (Carter, 1969). Similarly, part of the life cycle of Senecella calanoides is spent in the bottom sediments, but this copepod differs from L. macrurus in its continued preference for a cold mid-water habitat (depths of 60 m or greater) where it exhibits only a weak migratory behaviour (Carter, 1969). Shallow nearshore waters are characterized by the presence of large cladoceran fauna (Watson, 1974).

In the Great Lakes, sculpins are considered to be dominant members of the benthic fish community, and several other species such as burbot, whitefish (lake and round), lake trout, alewife, rainbow smelt, lake herring and bloater have an indirect association with the benthic habitat, largely as a result of their feeding habits. The deepwater sculpin (Myoxocephalus quadricornis) occurs at depths between 45 m and 180 m with a maximum abundance between 75 m and 105 m in the Upper Lakes (Scott and Crossman, 1973), but it is now believed to be extinct in Lake Ontario (Christie, 1972). The slimy sculpin (Cottus cognatus) is widespread in all of the Great Lakes and is typically found at depths between 6 m and 90 m with greatest abundance between 35 m and 75 m. The mottled sculpin (C. bairdi) is more typical of shallow water conditions where it prefers patches of cobble debris in areas of sandy silt. The spoonhead sculpin (C. ricei) occurs at depths intermediate between the slimy and deepwater sculpins; little is known about this species, which has not been recorded from Lake Ontario (Scott and Crossman, 1973).

FOOD WEB AND SUBSTRATES

Many members of the benthic community are detrivores and there is strong evidence to show that bacteria provide a very important pathway by which persistent organic materials, such as cellulose, lignin and chitin, can be made available to this fauna. Particle size, organic material and availability of oxygen are important controls on the type of fauna, but the presence of aerobic bacteria appears to be of particular significance (Bell and Dutka, 1972; Johnson and Matheson, 1968; Marzolf, 1965a).

The dominant species of oligochaete worms are burrowing detrivores. *Stylodrilus heringianus*, the most widely distributed form, is generally considered to be an indicator of oligotrophic conditions; *Tubifex tubifex* is both an oligotrophic and a pollution-tolerant form; *Limnodrilus hoffmeisteri* prefers organically enriched sediments and *Potamothrix vejdroski* has a strong preference for coarse sediments (muddy cobble gravel lag deposits) with a high organic content (Nalepa and Thomas, 1976). Although many of the worms may be regarded as unspecialized feeders, the work of Brinkhurst and Chua (1969) suggests that various species have different means of utilizing nutritional resources in sediments. In Toronto Harbour muds, for example, *Peloscolex multisetosus* was found capable of utilizing amino acids from solution, in the absence of gut bacteria, whereas neither *T. tubifex* nor *L. hoffmeisteri* could do so. Oligochaete worms typically ingest "aged" materials at a depth of 2-5 cm below the sediment surface (Gardner *et al.*, 1983) and population densities appear to show a positive correlation with the thickness of detrital layers (Alley and Anderson, 1968), but this may not be characteristic of all forms.

The Chironomidae are represented by both detrivores and predator species. Most detrivore species feed on suspended or very recently deposited material close to the sediment/water interface (Gardner *et al.*, 1983), and their mean ratio for excretory products is similar to those of the phytoplankton biomass reported by Goldman *et al.* (1979). *Chironomus* sp. is a detrivore, typical of areas of low to moderate sediment accumulation; *Heterotrissocladius* sp. can tolerate slightly greater rates of accumulation; and *Micropsectra* sp. is typical of areas of high sedimentation rates. *Procladius* sp., however, is predatory (Warwick, 1978) and typically feeds on other chironomid species that move at, or become forced to, the sediment surface because of excessively high rates of accumulation.

Pontoporeia affinis is the dominant amphipod and has a preference for fine sand - coarse silt size sediments (Henson, 1970), but it can be found in most sediments finer than 1.0-phi size (0.5 mm) as noted by Marzolf

(1965a,b). It is a shallow-burrowing detrivore typical of well-oxygenated interface conditions. It shows a preference for bacteria-rich sediments, largely independent of their content of organic material (Marzolf, 1965a,b; Mozley and Howmiller, 1977). This suggests that *Pontoporeia affinis* is utilizing only that portion of available organic debris which is being actively degraded by bacteria, and because of the oxic conditions required by this amphipod, the bacterial populations are characterized by high aerobe: anaerobe ratios.

Mysis relicta was originally thought to be both a grazing herbivore, feeding on the deep summer phytoplankton maximum, and a carnivore, feeding on smaller zooplankton at night (Bowers and Grossnickle, 1978; Lasenby and Langford, 1973). It was also thought to be a detrivore during periods of reduced plankton availability. Most recently, however, Parker (1980) has demonstrated that *Mysis relicta* is capable of deriving a significant portion of its dietary intake from predation on *Pontoporeia affinis*, and because of the size of mysid populations this implies that they can act as an additional major pathway for the transfer of benthic organic carbon to fish. During the early stages of their life cycle, other smaller zooplankton (e.g. *Limnocalanus macrurus* and *Senecella calanoides*) are also associated with a benthic habitat, but little is known about their feeding habits. Most adult forms are considered to be filter feeders (Kibby and Rigler, 1973; Rigler, 1972), although recent observations (Bowers and Warren, 1977) have demonstrated that *L. macrurus* can be carnivorous. It is therefore possible that other pathways exist between the benthic organic carbon pool and fish, in addition to the direct transfer from *Pontoporeia affinis* and the indirect transfer through *Mysis relicta*.

Freshwater pelecypods are filter feeders, and for non-sessile forms the characteristics of the supporting substrate (low clay content) are particularly important (Johnson and Matheson, 1968). Gastropods are mostly detrital feeders, making use of plant debris at the sediment surface.

Fish diets are normally determined from analyses of stomach contents. The following summary is based largely on the statements of Scott and Crossman (1973).

Mottled sculpin	Large range of benthic invertebrates
Slimy sculpin	(including mayfly, caddisfly, stonefly and dragonfly larvae, dipterous larvae, oligochaetes and amphipods) and organic debris
Deepwater sculpin	Amphipods, mysids, copepods (especially *Limnocalanus*) and chironomid larvae

Burbot	Eggs, young ciscoes, perch, alewife, smelt, sculpins, and mysids
Whitefish and round whitefish	Chironomid larvae and imagoes, amphipods, isopods, gastropods, pelecypods, ostracods and planktonic crustaceans
Lake trout	Young whitefish, ciscoes, smelt, perch, alewife, sculpins, mysids, amphipods, other crustaceans and freshwater sponges
Alewife	Mostly zooplankton feeders (copepods, cladocerans and ostracods) but also mysids and amphipods
Rainbow smelt	Omnivorous feeders, including young burbot, whitefish and lake trout, sculpins, mysids, amphipods, chironomid larvae and oligochaetes
Lake herring cisco and bloater	Mysids, amphipods and plankton

Figure 1 provides a summary of the food-web pathways towards some top predators; the food-web structure of the intermediate depth zone shares pathways of both shallow and deep water environments. Although little data exist to show that oligochaetes are part of the diet of sculpins, and other intermediate benthic predators, they have been reported in smelt (Hurley, in preparation; Scott and Crossman, 1973) and sculpin (Godkin, in preparation). There is no reason to believe that this food source remains completely unutilized and it is assumed that they are eaten by fish when available at the sediment surface (their breakdown after ingestion is fairly rapid and setae are hard to identify).

Although data are not available to accurately quantify vectoring of benthic and pelagic systems in Lake Ontario (Christie, 1972), benthic productivity is generally lower than pelagic productivity and has a longer turn-over time for nutrients. Based on the faunal distribution data (Nalepa and Thomas, 1976) and upon what is known of the food web in Lake Ontario (Christie and Thomas, 1981), it is evident that biological productivity remains high at depths much below the effects of surface wave activity. The lowest levels of productivity in Lake Ontario occur at depths of more than 100 m (covering an area of about 45% of the total lake).

LIFE CYCLES AND DISPERSALS

Oligochaetes are permanent members of the benthic community. Because of their reproductive process they are capable of rapidly expanding existing colonies but with a rather low rate of dispersal. Although the shelled molluscan fauna are also permanent members of the benthos, free-swinging and/or parasitic stages provide a means of spreading their populations in a way that would be otherwise impossible, even for the non-sessile forms. The life span of the larger shelled species covers several years. Chironomids, however, inhabit the bottom sediments only during the larval stage of their life cycle; for them, redistribution is accomplished largely as a result of dispersal during the adult stage.

Pontoporeia affinis appears to go through a migratory phase (Marzolf, 1965b) which may be associated with reproductive processes, and it is assumed that this species is capable of fairly rapid but passive dispersal to areas of suitable habitat. *Mysis relicta* (Reynolds and DeGraeve, 1972) has a 1- to 2-year life span, and because of the effects of both diel migration (Beeton, 1960; Carpenter *et al.*, 1974) and large-scale water motion, this species also appears capable of rapid dispersal (via passive transport).

Sculpins have a life span of 5-6 years and mature at about age 2. Most lake sculpins migrate to shallow water for spawning and slimy sculpins, which return to the nearshore habitat, are usually territorial. It is not known if this behaviour is also shared by the mottled and deepwater sculpins in the offshore environment, but it is probable. The migratory habits of spawning sculpins ensure a reasonably high rate of dispersal.

PHYSICAL ASPECTS OF DISPOSAL

The physical effects of material disposal in lake waters give rise to three points of biological concern: the effects of particulate suspension in water, the smothering effect on the bed, and the effect of changing the particle-size composition and bedform of the receiving substrate.

Observations on disposal plumes in the Great Lakes have been limited to water depths of 20 m or less, but it is evident that background concentrations of suspended particulates are typically re-established within a few hours (Chemex, 1973); it is also evident, in terms of periods of naturally high suspended sediment concentrations in the water column (storms and spring runoff), that disposal plumes are quantitatively insignificant. Therefore, although it is recognized that high concentrations of particulate matter can be deleterious to plankton and fish (Report of

TOP PREDATORS

Rock bass

Largemouth bass

SHALLOW WATER

SMALL SIZE DETRITAL
FEEDERS

INTERMEDIATE PREDATORS

Oligochaetes

Chironomid larvae

Amphipods

Sculpins

large insect larvae

crayfish

TOP PREDATORS

HARD-SHELLED FILTER AND

DETRITAL FEEDERS

Yellow perch

Whitefish

Pelecypods

Gastropods

INTERMEDIATE DEPTH (part shallow, part deep)

INTERMEDIATE
PELAGIC PREDATORS

Alewife, Smelt,

Herring cisco,

Bloater, Burbot

Lake trout

TOP PREDATORS

Burbot

Lake trout

Amphipods

Chironomid larvae

Oligochaetes

INTERMEDIATE
BENTHIC PREDATORS

Mysids,

Sculpins, Alewife,

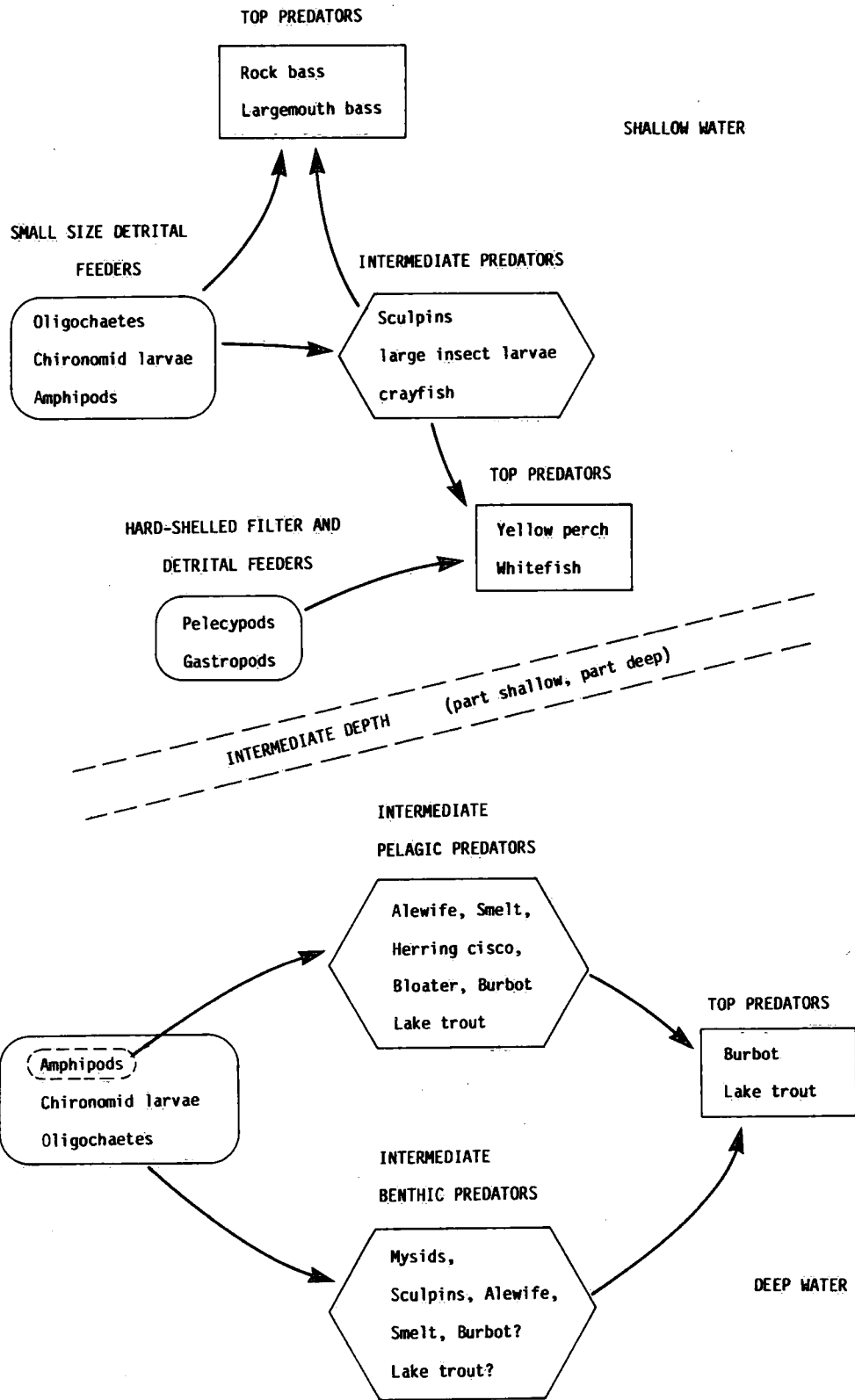Smelt, Burbot?

Lake trout?

DEEP WATER

Figure 1.   Summary of food-web pathways to some top predators (note: non-benthic food-web associations of intermediate and top predators have been omitted). Amphipods are the major food source for intermediate predators in deep water.

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International Working Group on the Abatement and Control of Pollution from Dredging Activities, 1975), the small area (usually a few thousand m²) relative to open lake area (Fig. 2), and the transience of the event, clearly indicate minimal impact on the biota. This assessment, however, may not remain valid in confined nearshore areas, if disposal is by means of continuous or near-continuous discharge, or if disposal is coincident with a specific and sensitive biological event.
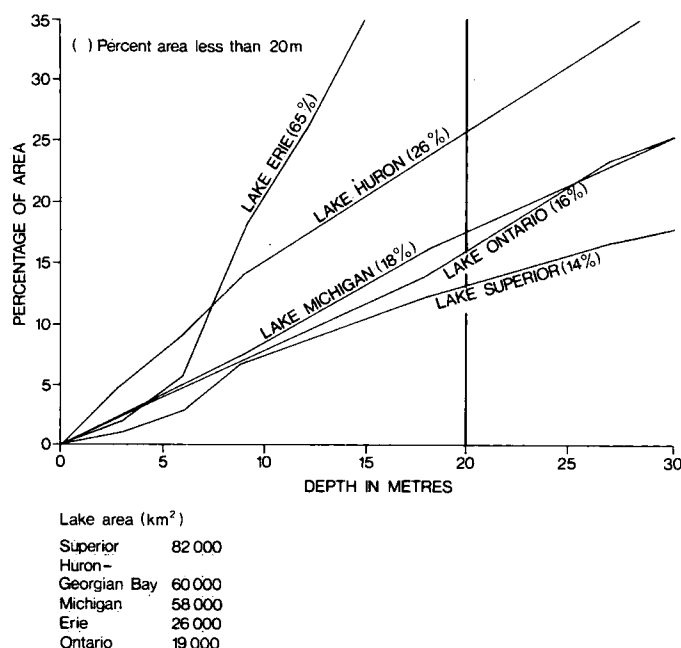

Figure 2. Lake Ontario area-depth curves (modified after U.S. Fish and Wildlife Service Report, 1969).

Since natural bioturbation is most active in the topmost 2-5 cm of sediment in the Great Lakes (Robbins *et al.*, 1979) and upward mobility of the benthic fauna is often insufficient to penetrate more than a few centimetres of additional material, dumped sediment has the effect of smothering most of the pre-existing biota at disposal sites. This has been demonstrated by studies in western Lake Erie by Flint (1979) and Sweeney *et al.* (1975). A change in biota resulting from the effects of sediment dumping (depth about 20 m) was clearly discernible for a least 5 years after disposal took place (Sweeney *et al.*, 1975). In the study by Flint (1979) the effects of disposal of coarse riverine sediments seemed to be less long-lasting than the disposal of silty harbour mud, but populations of isopods and some nematodes, pelecypods and gastropods were depressed. The effect of smothering, therefore, can cause a long-term change in the benthic environment, but relative to the total depth-area of the Great Lakes

(Fig. 2), dump sites of a few km² pose little significant change to the total open lake environment. The change on a local basis is direct, but it is also possible that a disposal site may have an effect upon the surrounding habitat through some form of indirect modification. The effects of smothering indicate the need for this aspect of biological assessment to be strongly site-specific.

The study by Flint (1979) demonstrated that changes in the sedimentary environment, resulting from disposal, favoured opportunistic species; and this has also been observed in the marine environment (Grassle and Grassle, 1974). However, no studies have specifically addressed the effects on biota of local changes in particle-size composition caused by open water disposal. Where the source of materials is of similar silty particle-size composition to the receiving site (or slightly finer), it is likely that the transposed biota may persist. If the source material is much coarser than at the receiving site, it is unlikely for source communities to persist, unchanged. To assess likely changes comparison may be sought elsewhere in the lakes for generally comparable relict deposits, where coarse materials are being slowly covered by the accumulation of fine muds; Figure 3 shows one such habitat in Georgian Bay, which appears to be particularly attractive to sculpins (Sandilands and Sly, 1977). Biota associated with artificial reefs and spawning substrates are generally representative of high-energy lacustrine environments, and are not comparable to communities of modified coarse substrate in intermediate or profundal depths of open lake waters.
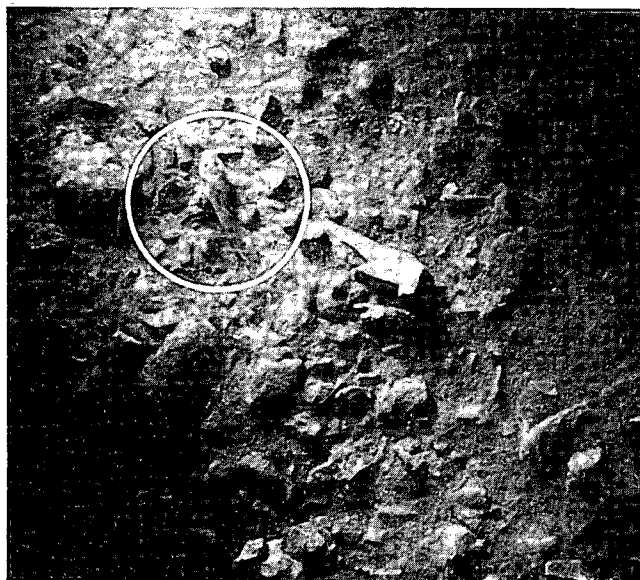

Figure 3. Sculpin, on coarse substrate being modified by accumulation of Recent mud (after Sandilands and Sly, 1977).

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A

WATER SURFACE

INTERFERENCE

NOTE: WATER COLUMN ESSENTIALLY CLEAR

LAKE BED

SCALE LINES SEPARATION 15m

EVIDENCE OF OLD DREDGE
DUMPING AND DISPOSAL

B

WATER SURFACE

NOTE: EXTREME NOISE IN WATER COLUMN,
NUMEROUS BIOLOGICAL REFLECTORS.

LAKE BED

SCALE LINES SEPARATION 15m

INTERFERENCE

EVIDENCE OF FRESH DREDGE
MATERIAL DISPOSAL
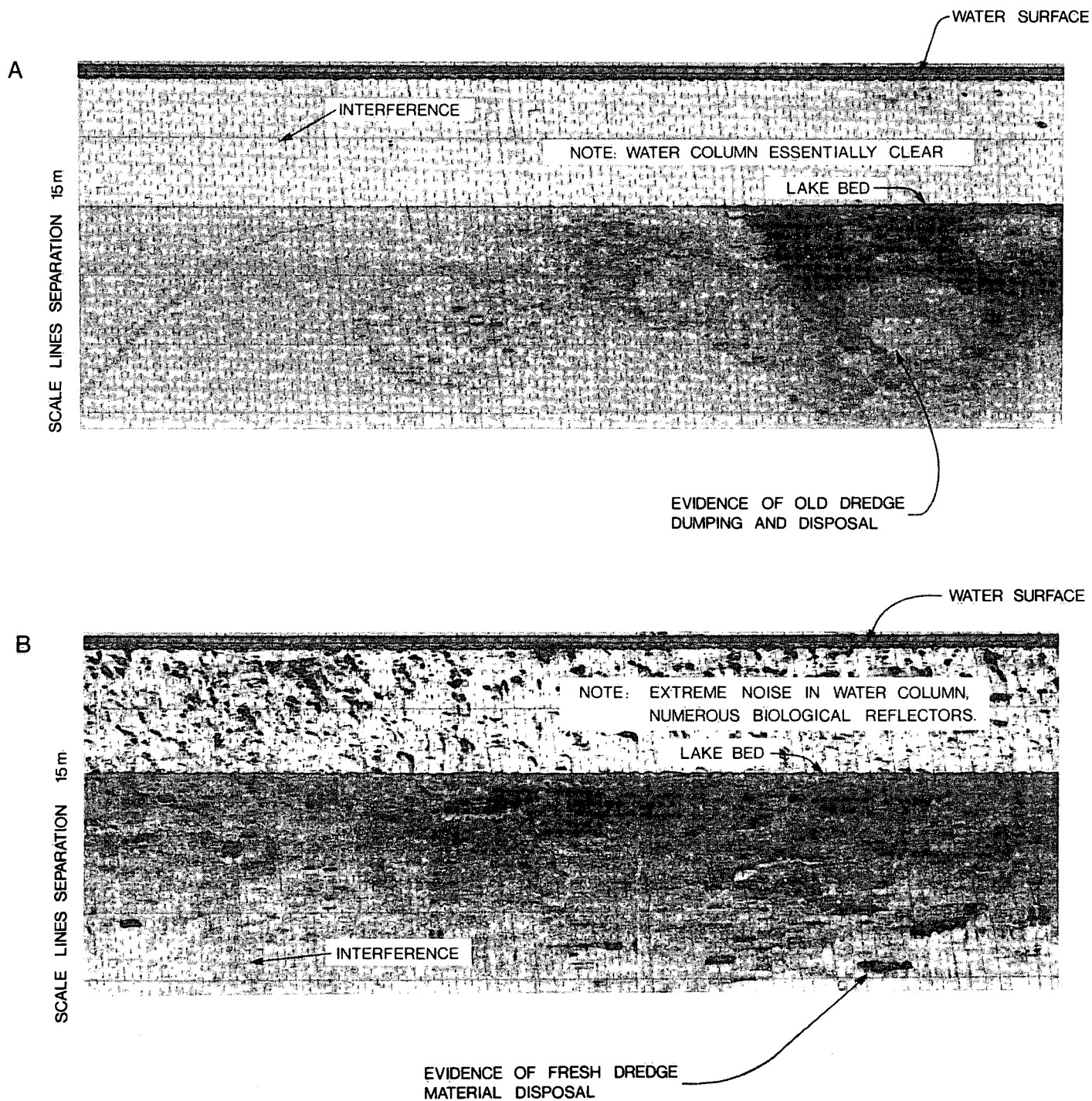
Figure 4. A comparison between adjacent areas at a dredged material dump site, Thunder Bay, Lake Superior, May 1975 (after Report of the International Working Group on the Abatement and Control of Pollution from Dredging Activities, 1975).

Comparative starboard hand side-scan sonar records of similar bottom sediment areas at a dump site. (A) Lake bed in area where dumping has been undertaken periodically, but not during 1974. (B) Lake bed in area where dumping has been undertaken within the last year.

SPECIES INTRODUCTIONS

As well as the suppression or elimination of pre-existing benthic fauna by disposal materials, the study by Flint (1979) demonstrated that a number of additional species of oligochaetes were introduced to the disposal sites, including members of the genus *Limnodrilus* and *Tubifex tubifex*. These species, however, did not disperse to colonize the areas adjacent to the disposal site. *Limnodrilus hoffmeisteri* was present prior to dumping at the disposal site, but its numbers increased considerably after disposal. This was thought to be due to both the introduction of mature forms which were opportunistic in their reproduction and to the introduction of numerous immature forms of the genus *Limnodrilus*.

The disposal of dredged sediments from harbour areas may also introduce significant populations of bacteria to lake bed dump sites. Based largely on experimental evidence, Scarce *et al.* (1964) concluded that streptococcus bacteria could persist for a number of days under conditions of improved water quality and that coliform bacteria might actually show an initial increase in numbers before approaching a die-away rate.

At temperatures of 5°C or less, when protozoan predation would be minimal, the bacteria would probably survive much longer than at optimum predator temperatures of 10°C-28°C. The fact that disposal of harbour muds did not reduce local populations of *Gammarus fasciatus* (Flint, 1979) strongly implies that amphipods were immediately able to utilize, directly or indirectly, the content of the dumped material. Under favourable conditions, therefore, recently dumped material may attract higher predators which, in turn, utilize the concentration of feeding amphipods; this may be the explanation for the aggregation of fish at the Thunder Bay dump site shown in Figure 4. The process of dredging and dumping has the effect of indiscriminately transposing large numbers of organisms from one environment to another; many die because of the change but, because of the transfer, there is also a potential for spreading the distribution of parasitic forms. The significance of this, however, remains uncertain.

SHORT-TERM CONTAMINANT RELEASE

During dredging and disposal, sediments become disturbed and are often partially disaggregated. Thus, when dumped through the water column, some of the pore water is released during the period of material fall, and the release process may continue for some days while sediment load adjustment and reconsolidation take place on the lake bed. Observations by Chemex (1975), in Lake Erie, tend to support this and demonstrate that concentrations of total and soluble phosphate increased in the water column after dumping (total phosphate decreased from 170 $\mu g/l^{-1}$ immediately following dumping to less than 100 $\mu g/l^{-1}$ over a period of about a week, but soluble phosphate increased from about 20 $\mu g/l^{-1}$ to more than 50 $\mu g/l^{-1}$ in this time). Both Fe and Mn are frequently released from sediments during disposal (Sly, 1977) and their behaviour follows well-defined pH/redox potential controls; Zn and other heavy metals have also been found to increase in waters overlying dumped sediments, presumably from the same cause.

These observations confirm that small but measureable short-term contaminant release can take place from dumped material, and it is evident that, at least on a local basis, there is potential for an impact by macro- and micro-nutrients and persistent contaminants, particularly upon microplankton. Because of the relatively high productivity of nearshore areas these effects would be greatest in sheltered shallow water.

LONG-TERM EFFECTS—MERCURY EXAMPLE

Although contaminant uptake in fish occurs through more than one pathway, the presence of contaminants in ingested food is often the largest single contributory factor and, as demonstrated in Figure 1, many intermediate and top predators are sustained by food-web structures which include ingestion of benthic fauna. The proportion of diet represented by benthic feeding varies from species to species, with stage of development and in both space and time, and is therefore difficult to estimate (Elliott and Persson, 1978). The sediment biota, as stated by Johnson and Brinkhurst (1971b), are represented by at least four trophic levels; primary production is provided by chemosynthetic bacteria at depth in the sediments, oligochaete and chironomid detrivores form the major group of saprophytes, and there are at least two levels of consumer organisms, such as carnivorous chironomids and amphipods. Therefore, the more a fish population depends on benthic faunal diet, the more quickly it is likely to reflect bottom sediment contaminants uptake. This seems to be borne out by the work of Thommes *et al.* (1972), who showed that the highest (wet weight) levels of mercury occurred in sculpin and stickleback (shallow water benthic feeder) and the lowest levels in planktivores.

Studies on the Wabigoon River system by Jackson *et al.* (1982) have shown that binding sites discriminate between different forms of mercury and that concentrations of "dissolved" methyl mercury are seasonally variable. The production of methyl mercury is not related to total

mercury concentrations but, rather, to bacterial activity (Cooley and McCarty, 1976). This seems to be supported by the work of Skoch and Sikes (1973), who suggested that there was a relationship between the presence of methylating bacteria in Lake Erie sediment and the amount of absorbed mercury in chironomids. Their data indicated that the difference in mercury concentrations represented time-dependent absorption of mercury from the surrounding media (pore water), rather than absorption from ingested material. Since tissue retention of methyl mercury (Huckabee and Goldstein, 1975) appears to be longer than for inorganic forms in food supply, it is evident that, so long as mercury is present in sediments, its incorporation in top predator fish will be largely influenced by the rate of bacterial activity and the feeding pattern of inter-mediate predators. For contaminated sediments to provide a significant source of methyl mercury loading in fish the transfer mechanism must be continuous for some time; however, seasonal transfer rates will show considerable variation, with the highest rates occurring during the spring-summer period. It may be argued that concentrations of methyl mercury in fish are not related to the increased concentrations of total mercury in Great Lakes sediments. It is likely, however, that since mercury is widespread in Great Lakes sediments (Thomas, 1974), increased bacterial activity associated with the effects of cultural eutrophi-cation will increase the rate of methyl mercury production at source.

LONG-TERM EFFECTS—PCBs AND DDT EXAMPLES

The behaviour of persistent organochlorine con-taminants in aquatic systems differs markedly from that of mercury. The effects of lethal toxicities, which can be esti-mated from the concentration of the contaminant in water expressed in relation to its solubility (Hutchinson et al., 1980), tend to be low, but suppression of growth and other sublethal effects have been noted at observed environ-mental and laboratory concentrations (Pfister et al., 1970; Wyman and O'Connors, 1980). In some situations, the presence of organic contaminants may stimulate micro-bial activity (Pfister et al., 1970). Experimental evidence (Clayton et al., 1977; Scura and Theilacker, 1977) indicates that concentrations of organic contaminants in biota are related to equilibrium partitioning between water and the lipid pool (represented by n-octanol as a surrogate, as shown in Figure 5). Studies on zooplankton-phytoplankton relationships (Wyman and O'Connors, 1980) and on selected biota from Lake Ontario, as shown in Figure 6 (Borgmann and Whittle, 1983), seem to confirm this. However, the re-lationship is not entirely confirmed and studies by McLeese et al. (1980) imply that concentrations in some marine worms and shrimps may be otherwise controlled.

PCBs and similar persistent contaminants are strongly hydrophobic and usually enter aquatic systems in associa-tion with particulates, or rapidly become associated with them. PCBs, however, are quite volatile (Millard et al., in preparation) and this allows rapid uptake by phyto-plankton, through loss of PCBs absorbed onto the outer sur-face of a particle and transfer to the lipid pool by diffusion through the cell wall. The affinity for the lipid pool means that PCBs are transferred to the biota from both inorganic and organic detritus. Transfer may also take place during transient contact between particles (Harding and Phillips, 1978). PCBs adsorbed in the interstices of particles are largely retained and the concentrations of PCBs in seston and lake sediments tend to be similar (Glooschenko et al., 1976). The rates of uptake in small organisms with a high area-to-volume ratio (Harding and Phillips, 1978) can approach an asymptote in less than 2 h. In fish, PCBs accumulate by rapid exchange across the gill surface and pass into the lipid pool via the circulatory system; as much as 40% of PCBs uptake in fish may be accounted for by this mechanism (Thomann, 1977). Although the desorption/transfer process could account for the total accumulation of PCBs in the lipid pool of the biota, the rate of uptake is strongly influenced by the ingestion pathway. For example, copepods feeding on PCB-contaminated phytoplankton reached asymptotic PCB body burdens in less than 2 days under experimental conditions (Wyman and O'Connors, 1980). Hence, direct adsorption or food-web pathways may contribute to the final PCB body burden in widely differing proportions.

For organochlorine and similar halogenated contami-nants which are less volatile than PCBs, such as DDT, the principal pathway of bioaccumulation is that of the food-web (Reinert et al., 1974). DDT, with very low solubility in lake water (Fig.5), is retained almost indefinitely as body burden, but aldrin, dieldrin and endrin, which have higher solubilities (Weber, 1977), will tend to decline in uncon-taminated conditions.

All known halogenated organic contaminants are bio-degradable, but the degradation rates are extremely variable both in terms of the compound and the environmental conditions. DDT, for example, is degraded much more rapidly under anoxic conditions (-150 mV) than under weak oxic conditions (+50 mV), and is nearly stable under normal oxidizing conditions (Gambrell et al., 1981). DDD domi-nates the breakdown products of DDT but is not stable. Kepone, on the other hand, appears to be nearly stable under a wide range of naturally occurring redox and pH conditions, as is Mirex.
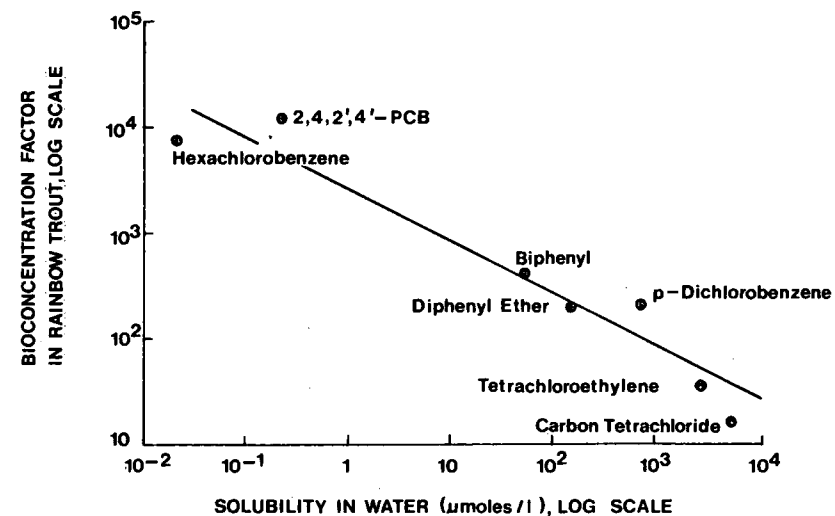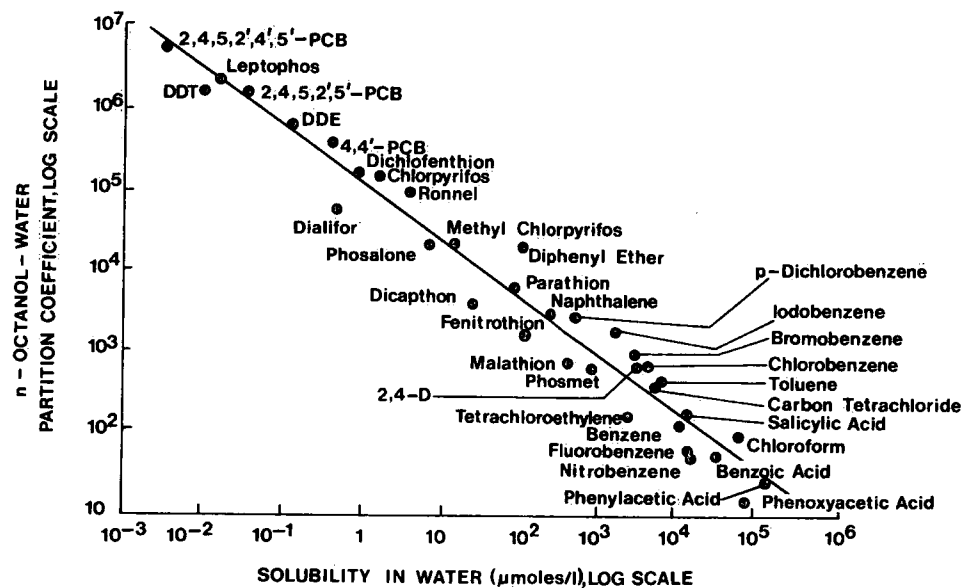
Figure 5. Lipophility and bioaccumulation. Both *n*-octanol-water partition coefficient and bioconcentration factor in rainbow trout are inversely proportional to aqueous solubility (Reprinted with permission from Chiou, C.T., Freed, V.H., Schmedding, D.W. and Kohnert, R.L., 1977. Partition coefficient and bioaccumulation of selected organic chemicals. Environ. Sci. Technol. 11:475-478. Copyright 1977, American Chemical Society.)
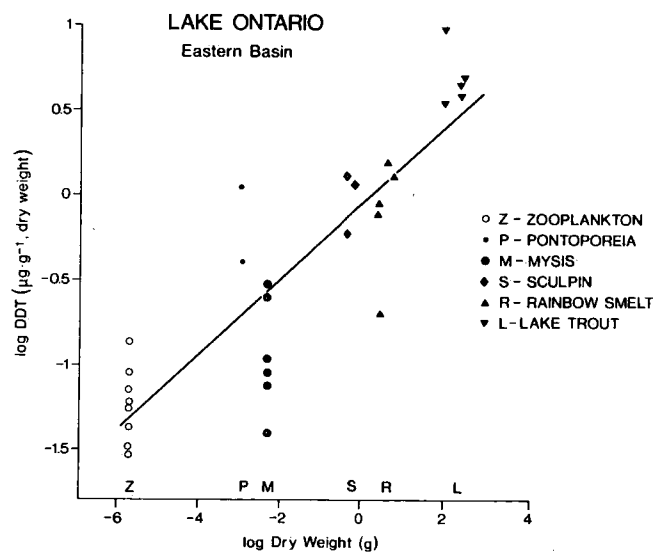

Figure 6. DDT concentration as a function of body size (after Borgmann and Whittle, 1983 ; reprinted with permission from Can. J. Fish. Aquatic Sci.)

SUMMARY

In summarizing present biological concerns about the use of "open water disposal" it is assumed that materials are not lethally toxic to the biota and that dumping will not occur at sites or during periods when important biological events are taking place. Also, to avoid redistribution, disposal of fine materials should not occur in areas of non-deposition or of active erosion by bottom currents.

1. Unless specifically intended to disperse (e.g. beach nourishment), and then only if clean, materials should not be disposed of at depths where bottom sediments will be reworked and redistributed by wave or ice action.

2. Whenever possible, dumped materials should be disposed of at depths where biological productivity is least; i.e. in deep water (typically 100 m in Lake Ontario). If materials have to be "dumped" in intermediate depths, between about 20 and 100 m, the disposal project must be designed to cause minimal impact on the biota of the zone. From a biological point of view, disposal in this zone is considered most undesirable.

3. Dumped materials, except for (1) above, should not be spread over a wide area; they should be concentrated to affect the smallest area of benthic habitat.

4. Materials having some form of contaminant present should not be allowed to disperse during settlement, particularly if they contain quantities of PCBs or similar halogenated organic compounds. If the dump sites are not in areas of naturally high rates of sedimentation, "contaminated" materials should be sealed off by subsequent deposition of an inert or benig cover.

5. Coarse materials can be used to modify bottom sediment micro-relief, and to enhance habitat for some members of the benthic community (e.g. sculpins).

6. No lessening of biological concern should be expected in the future with regard to the impact of toxic contaminants. However, some relaxation of concerns over the presence of macro- and micro-nutrients may take place, especially if biological enhancement appears possible and desirable.

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