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TOXICITY OF VERTICAL SEDIMENTS IN THE TRENTON CHANNEL, DETROIT RIVER, MICHIGAN, TO *CHIRONOMUS TENTANS* (INSECTA: CHIRONOMIDAE)

Cornell J. Rosiu† and John P. Giesy*
Pesticide Research Center
Department of Fisheries and Wildlife
Center for Environmental Toxicology
Michigan State University
East Lansing, Michigan 48824

Russell G. Kreis, Jr.
United States Environmental Protection Agency
Large Lakes Research Station
9311 Groh Road
Grosse Ile, Michigan 48138

ABSTRACT. The objective of this study was to determine the effects of sediment from various sediment core depths on survival and weight gain of larvae of the dipteran midge, Chironomus tentans, *during 10-d laboratory exposures. Sediment cores were collected from 12 sites in the Trenton Channel of the Detroit River in 1987 and sectioned into 5-cm intervals to a depth of 25 cm. Percent reductions in larval weight gain, relative to that in control sediment, were calculated for each interval. Two sites were classified as very toxic, three sites as toxic, three sites as slightly toxic, and four sites as good quality benthic habitat. The utility of sediment core toxicity profiling and the* C. tentans *bioassay for three-dimensional sediment quality assessment are discussed, as well as comparisons between the results of laboratory assays and field surveys of benthic macroinvertebrates. The assay results are used to estimate the volume of toxic sediment at eight sites and determine the costs of dredging and disposal of the toxic sediments. Preliminary estimates of remedial actions were developed to achieve several levels of mitigation of the toxicity of sediment to macrozoobenthic populations in the Trenton Channel.*
ADDITIONAL INDEX WORDS: Benthos, benthic environment bioassay, macroinvertebrates.

INTRODUCTION

Sediments can become contaminated by the introduction of persistent organic and inorganic substances into surface waters where they become associated with particles, which can settle and accumulate in bottom deposits. The rate at which this occurs and the degree of contamination of the sediments depend on many factors such as site-specific particle deposition rates, transport dynamics, and specific characteristics of the in-place contaminants and sediments. Within rivers, sediment bed entrainment and redeposition of contaminants from point sources can relocate sediments downstream such that a horizontal gradient of toxicity is observed. The subsequent depositional pattern of sediment-bound contaminants affects the vertical evolution of sediment toxicity.

The Detroit River and western basin of Lake Erie are known to be affected by industrial and municipal wastes, urban and agricultural runoff (IJC 1981). The concentrations of many contaminants in sediment exceed objectives and criteria for metals and organic xenobiotics set by the Ontario Ministry of the Environment (MOE) and International Joint Commission (IJC) (Hamdy and Post 1985, Fallon and Horvath 1985, Chau *et al.* 1985, Maguire *et al.* 1985, Kaiser *et al.* 1985, Platford *et al.* 1985,

*To whom correspondence may be addressed.
†Current Address: Marine Pathology Laboratory, Dept. of Fisheries, Animal and Veterinary Science, University of Rhode Island, Kingston, Rhode Island 02881

Pranckevicius 1986). The degree and distribution of toxicity of surficial sediments in the Trenton Channel, Michigan, has been previously reported (Giesy *et al.* 1988a, b).

Surveys of the distribution of benthic macroinvertebrates have been conducted on the Detroit River (Hiltunen and Manny 1982, Thornley and Hamdy 1984, Horvath and Kenega 1985) and have identified patterns of benthic habitat degradation and the limited recuperative capacity of the benthic macroinvertebrates in areas of the river (Thornley 1985). While the greatly contaminated areas in the lower Detroit River are a small portion of the aquatic habitat in the Great Lakes ecosystem, their local and overall importance to production in the system is probably greater than the affected areas suggest. Juvenile fishes use the regions of major sedimentation for nursery habitat (Goodyear *et al.* 1982) and depend on the successful rehabilitation of benthic invertebrate communities for reestablishing a naturally reproducing fishery (Hartman 1988).

Effective monitoring of benthic habitat quality for purposes of planning cost-effective remedial action necessitates a measurement resolution greater than qualifying statements such as "grossly contaminated," "unbalanced macrozoobenthic community structure," or "exclusionary to macrozoobenthos." The presence or absence of benthic macroinvertebrates is one possible indication of adverse conditions in the surficial sediments. However, synoptic survey data cannot determine the cause for the observed effects because the distributions of organisms are affected by numerous factors other than toxicity. In addition, survey data cannot identify the toxicity in deeper sediments where organisms do not naturally occur.

Because buried sediments have the potential to be uncovered and resuspended as a result of natural and human activities, characterizing the three-dimensional distribution of toxic sediments is necessary for the effective management of in-place pollutants and planning for possible remedial action and urban waterfront development. Sediment toxicity assays can provide necessary information by developing a cause-and-effect relationship and quantifying the horizontal and vertical profiles of sediment toxicity. If remedial dredging is indicated, the mass and volume of sediment which would need to be removed can be calculated. Such measurements can also provide insight into the relative duration between contaminant depositional events so one may determine if contamination is due to current occurrences or historical events.

The results of sediment bioassays have been shown to be correlated with the distribution of benthic organisms in sediments (Malueg *et al.* 1984a, Giesy *et al.* 1988a), although sediment bioassays can better resolve the degree of habitat quality of degredation among sites (Lamberson and Swartz 1988, Giesy *et al.* 1988a). Bioassay data can help one determine the dilution of a contaminant that is necessary to permit recolonization by indigenous aquatic insects (Giesy *et al.* 1990) and, as in the present study, demonstrate the potential toxicity of the uninhabited deeper sediments.

Chironomus tentans is widely distributed in freshwater sediments during its larval development and spends the majority of its lifecycle in a burrow within the upper few centimeters of the sediments (Adams and Heidolph 1985). Chironomids often comprise a significant proportion of the benthic biomass and are important in the cycling of residues to and from the sediments as a result of their bioturbation activities (Gerould *et al.* 1983). *C. tentans* can be satisfactorily reared in the laboratory and has been successfully used as a bioassay organism (Wentsel *et al.* 1977, 1978; Batac-Catalan and White 1983; Mosher 1982; Nebeker *et al.* 1984; Ziegenfuss and Adams 1985; Malueg *et al.* 1984b; Giesy *et al.* 1988a,b, 1990). Individuals of the family Chironomidae are typically found in Great Lakes sediments which have macrozoobenthic communities.

The objective of this study was to determine the chronic effects on growth of larval *C. tentans* in 10-d laboratory exposures of sediment from different depths in sedimentation zones of the Trenton Channel of the Detroit River, Michigan, USA. Results of the bioassays have been compared with the distribution of benthic macroinvertebrates in the Trenton Channel. The horizontal and vertical distribution of toxic sediments have been mapped and estimates made of the mass of toxic sediments that would need to be removed to allow rehabilitation of macrozoobenthic communities. We provide estimates of the cost of several levels of remedial action, based on current costs for handling toxic sediments.

MATERIALS AND METHODS

Sediment cores were collected in the summer of 1987, using 2-inch diameter stainless steel tubes with a Wildco Hand Core Sampler (Wildco Instruments, Saginaw, Michigan) from 12 locations within the Trenton Channel and Trenton Channel delta (Fig.

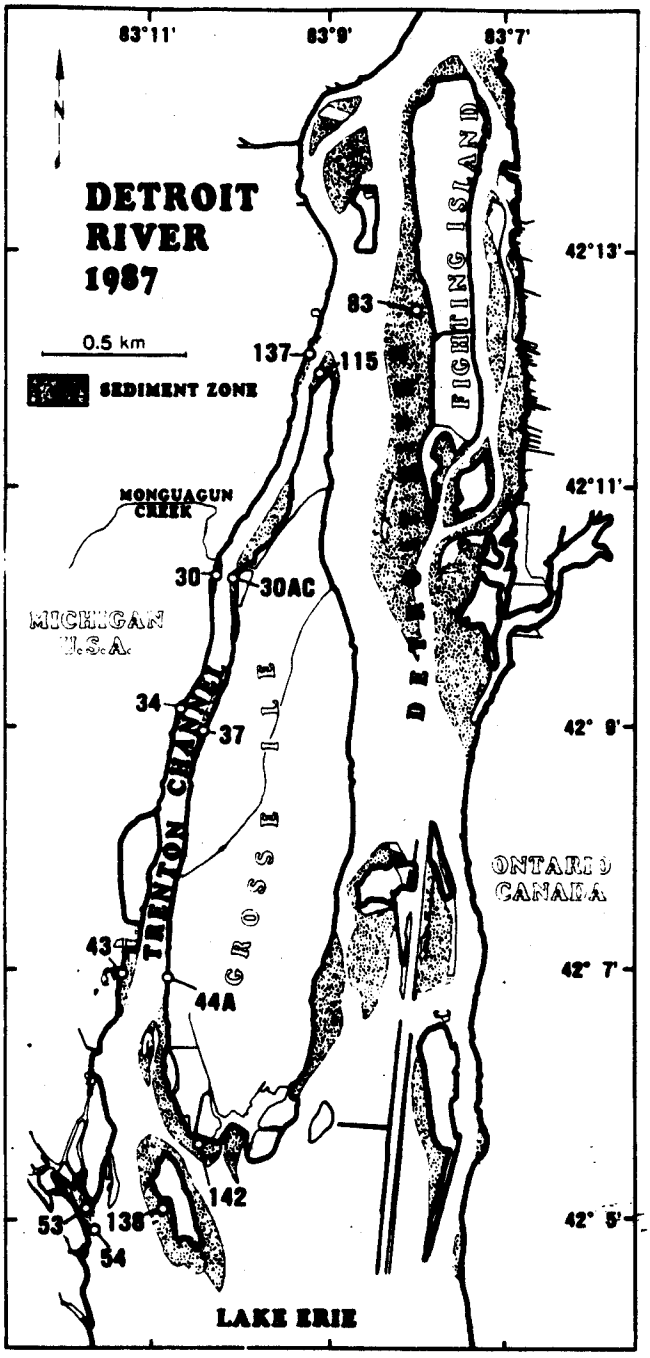

FIG. 1. Station numbers and locations for sediment-core samples and sedimentation zones in the Trenton Channel of the lower Detroit River. Station 83 was a reference (control) site.

1). Surficial sediment from an additional site (station 83) was used as a reference (control) sediment in this study. Major sedimentation zones have been defined by previous studies (Fallon and Horvath 1985) and minor sedimentation zones were identified using a National Ocean Service 1987 bathymetric map of the lower Detroit River. Cores extended to a depth of 25 cm or until a hard, compacted clay layer was reached; cores ranged from 10 to 25 cm in length. Five replicate cores were collected at each location and extruded from the core sampler into a clear, graduated Lexan plastic tube where they were sectioned at 5-cm intervals. The sediments from the five cores were composited respective to each 5-cm interval of sediment depth.

Sediments were visually characterized and samples placed into 950-mL glass bottles and stored on ice in coolers during transport to the laboratory. The following day, composite samples from the five depths (0–5 cm, 5–10 cm, 10–15 cm, 15–20 cm, and 20–25 cm) were manually homogenized per station. Large stones and fragmented macrophytes were removed. Aliquots of sediment were collected for chemical and biological characterization. The whole sediments were analyzed for percent water, insoluble materials, and carbon content. Organic carbon was calculated after combustion with a model 523, LECO Carbon Analyzer (Laboratory Equipment Corporation, St. Joseph, MI) as the difference between total carbon and inorganic carbon (Kolpack and Bell 1968). The chemical data are not presented in this paper, but can be obtained from the authors. Sediments were stored at 4°C until they were prepared for bioassays, which was always within 7 days of collection.

Detailed methods for the culture of *C. tentans* and the whole-sediment growth assay have been reported elsewhere (Giesy *et al.* 1988a). Briefly, tests were conducted with individual *C. tentans* larvae in 50-mL polypropylene centrifuge tubes (Corning Science Products). Each tube contained 7.5 g of sieved (1.00 mm opening), wet-weight sediment and 47 mL of deionized, reverse-osmosis treated tap water. Sediment and water were mixed and the suspension was allowed to settle out for 24 h. Each tube was then aerated with a 16 gauge stainless steel pipet and standard aquarium pump 24 h before one second instar larva (12 d post-ovaposition) was deposited into each tube. Each larva was fed 6 mg of Tetrafin® (TetraWerke, Inc., West Germany) per day. After 10 d, the individual *C. tentans* larvae were determined as found dead, not found, or recovered live. They were then dried at 80°C for 24 h in a convection oven, and then weighed to 0.01 mg with a Mettler Model H24 Balance. The lifecycle of *C. tentans* takes approximately 24 d to complete at 25°C. In culture, second instar larvae had an initial,

mean dry weight of 0.210 mg (n = 30, SD = 0.090 mg). Twenty-two day-old larvae grown on reference (control) sediment (station 83, Detroit River; see Giesy *et al.* 1988a) for 10 d had a final mean dry weight of 4.51 mg (n = 52, SD = 1.56 mg). Fifteen larvae were exposed to sediment from each depth interval at each location. This number of replications was determined by power analysis to provide detection of growth differences to 0.5 mg dry weight (Giesy *et al.* 1988a).

C. tentans growth in composites of sediment from each depth at each location was compared to growth in station 83 surficial sediment. Station 83 served as a control location where viable, balanced populations of indigenous benthic macroinvertebrates were observed, including *Hexagenia limbata* and species of the family Chironomidae (Thornley and Hamdy 1984, Horvath and Kenega 1985). Concentrations of contaminants measured in sediment from the reference (control) station were previously shown to be low or non-detectable (Kauss and Hamdy 1985) and did not cause lethality or reduced growth of *C. tentans* in earlier bioassay studies (Giesy *et al.* 1988a, 1990). The effects of whole sediments on growth of *C. tentans* are reported as percent reductions in weight gain, relative to surficial sediment from station 83. With the execution of each set of bioassays, a simultaneous assay with the reference sediment was conducted as a control. If the results of the control assays were determined to be statistically similar by multiple mean comparisons (α = 0.05), then the control data sets were pooled. Control mortality was always less than 10%. Lethality was included in the final growth data by the assignment of 100% reduction in weight gain to an occurrence of individual mortality (see Giesy *et al.* 1990 for rationale). The statistical relationships among stations and depths were investigated by general linear models (PROC GLM; SAS 1985) and Tukey's studentized multiple range test.

RESULTS AND DISCUSSION

The effects on growth of *C. tentans*, due to exposure to the upper 5 cm portion of the sediments from all locations, were similar to those observed in bioassays of surficial sediments from the same stations conducted in 1986 (Giesy *et al.* 1988a). In that study, a 30% reduction in weight gain of *C. tentans* in the laboratory, relative to the control, was statistically significant (ANOVA, $P<0.05$) and was observed to be the critical threshold toxicity of sediment. That is, above a 30% reduction in weight gain, naturally reproducing indigenous macrozoobenthos were also observed to be restricted in their colonization or were grossly unbalanced in community structure. In the study reported here, we observed a similar threshold toxicity of approximately 25% reduction in weight gain relative to control. A reduction in *C. tentans* growth greater than 25% was statistically different from control growth and this same level of reduction in weight gain, which represents statistical significance relative to control, was also similar to the growth reduction threshold for significant ecological effects.

Significantly (ANOVA, $P<0.05$) smaller weight gains, relative to control, ($\geq$25%) were observed among *C. tentans* exposed to sediments from one or more depths at eight of the 12 stations. However, weight gains on sediment from stations 44A, 138, or 54 were not significantly different ($P>0.05$) from control and were therefore determined to be non-toxic, as was station 37 which stimulated growth for all depths tested, over that of control (Table 1). For purposes of ranking station locations according to toxicity, bioassay data were averaged over all depths by station. The site mean (all depths averaged) reductions in weight gain for sediment from stations 53, 30AC, and 142 were 13.4%, 16.7%, and 23.6%, respectively. While slightly toxic to *C. tentans*, these site mean values were less than 25% reduction in weight gain relative to the control and were not significantly different from one another or surficial control sediment (Tukey's studentized test; α = 0.05, total df = 737, n = 15, k = 13, minimum significant difference = 27.9). The site mean (all depths averaged) reductions in weight gain of *C. tentans* for sediment from stations 137, 43, and 115 were 35.7%, 37.5%, and 37.9%, respectively. The growth responses of *C. tentans* to sediment from these three locations were not statistically different from one another, but all were above the 25% threshold and were classified as toxic.

Weight gains (all depths averaged) on sediment from stations 34 and 30 reduced by 82.8% and 83.7%, respectively. Weight gains from these two sites did not differ from one another significantly, and were considered to be very toxic.

Comparison of Bioassay Results for Surficial Sediments with Benthic Community Structure

Community and population descriptors of benthic macroinvertebrates are often used as indicators of

TABLE 1. Mean percent reduction in weight gain (% REDUCTION) of **C. tentans** *relative to station 83 (control) sediment and percent macroinvertebrate biomass field data (Horvath and Kenaga 1985) for each sediment-core sampling location. In bioassays, 15 individuals were exposed to sediment from each depth at each location. Negative values represent growth stimulus relative to control sediment. Individual lethality is included as 100% reduction in weight gain for an individual observation. Dash symbol (-) represents absence of the macroinvertebrate.*

STA.	CORE DEPTH (cm)	% REDUCTION	MACROINVERTEBRATES (% BIOMASS) (A) Hexa.	other(B) mayflies	(C) Tricop.	(D) other	(E) Amphip.	(F) Chiro.	(G) Olig.
30	0- 5	76.4 ± 5.0	-	-	-	-	-	-	-
	5-10	85.4 ± 5.0							
	10-15	89.8 ± 5.0							
	15-20	83.4 ± 8.3							
34	0- 5	87.1 ± 5.9	-	-	-	-	0.1	0.1	99.8
	5-10	81.8 ± 11.4							
	10-15	91.6 ± 8.1							
	15-20	83.4 ± 9.9							
	20-25	70.1 ± 20.5							
115	0- 5	32.7 ± 19.4	-	-	1.5	0.5	8.1	1.3	88.6
	5-10	79.4 ± 14.2							
	10-15	55.1 ± 18.2							
	15-20	25.8 ± 18.2							
	20-25	-9.9 ± 21.5							
43	0- 5	10.2 ± 25.2	-	-	11.6	7.6	3.0	5.0	72.8
	5-10	71.4 ± 12.2							
	10-15	52.4 ± 17.9							
	15-20	14.0 ± 13.5							
137	0- 5	34.2 ± 21.5	-	-	-	-	-	-	-
	5-10	32.3 ± 18.3							
	10-15	40.7 ± 19.9							
142	0- 5	-6.9 ± 19.3	5.4	-	4.4	3.5	18.3	13.6	54.8
	5-10	8.9 ± 28.4							
	10-15	46.7 ± 21.4							
	15-20	36.6 ± 24.0							
30AC	0- 5	-21.8 ± 25.8	-	-	0.1	-	0.1	1.1	98.7
	5-10	8.1 ± 22.8							
	10-15	52.8 ± 21.3							
	15-20	27.5 ± 15.7							
	20-25	5.4 ± 23.6							
53	0- 5	4.8 ± 21.1	-	-	6.2	4.6	2.0	11.2	76.0
	5-10	57.1 ± 15.0							
	10-15	10.1 ± 13.2							
	15-20	1.5 ± 26.2							
	20-25	-9.0 ± 21.5							
44A	0- 5	22.1 ± 32.6	-	-	-	-	0.4	4.7	94.9
	5-10	-7.2 ± 23.8							
138	0- 5	4.1 ± 32.8	-	-	-	-	3.3	0.9	95.8
	5-10	0.20 ± 19.6							
	10-15	-1.7 ± 29.8							
	15-20	-30.1 ± 66.5							
	20-25	8.1 ± 19.3							
54	0- 5	-20.6 ± 32.2	-	-	6.2	4.6	2.0	11.2	76.0
	5-10	-11.8 ± 32.2							
	10-15	6.2 ± 49.4							
	15-20	0.16 ± 30.8							
	20-25	0.81 ± 26.0							
37	0- 5	-114.3 ± 65.2	-	-	-	61.8	-	-	38.2
	5-10	-64.4 ± 42.9							
	10-15	-44.2 ± 50.6							
	15-20	-94.1 ± 47.2							
83	-	-	2.1	5.6	5.4	-	15.4	26.2	45.3

stream pollution (Goodnight 1973, Washington 1984, Tolkamp 1985, Hruby 1987). Several assumptions are made when one makes a comparison of the results of macrozoobenthos surveys. They include: 1) the chemical (excluding toxics) and physical substratum parameters are identical among comparison sites; 2) the major sources of colonizing individuals are the same; and 3) acutely toxic events which leave no residue record in sediments do not differentially occur among locations. In all likelihood, one or more of these assumptions are invalidated when surveying complex aquatic systems. This fact necessitates the use of laboratory bioassays to test, under controlled conditions, the assumptions used in surveys of benthic invertebrate communities. It is not our intention to provide a detailed discussion of the benthic macroinvertebrate field survey conducted in conjunction with this, and the 1986 bioassay study. Instead, we will make a general assessment of the predictive power of laboratory bioassays as they relate to corresponding observations of benthic communities in the Detroit River.

Site-specific benthic macroinvertebrate species composition and biomass were determined within the lower Detroit River (Horvath and Kenega 1985). Observations on the macroinvertebrate communities were made within the same sedimentation zones as the bioassays. In general, surficial sediments which caused a significant reduction in growth of *C. tentans* relative to reference (control) sediment under laboratory conditions, corresponded to those areas where the species composition of organisms were different from the nontoxic reference location (Table 1 and Fig. 2).

Excessive abundance of tubificid worms (>80% of the total number or organisms recovered) has been used as a qualitative measure of the extent of degradation of benthic habitat in the Detroit River (Thornley 1985). At stations 115 and 34, where a statistically significant reduction in weight gain of *C. tentans* was observed, oligochaete worms from the respective field stations represented a greater proportion of the total macrozoobenthos biomass (88.6% and 99.8%, respectively) when compared to the control station (45.3%; Table 1). At three of the nine locations from which surficial sediment caused sub-threshold reductions in weight gain of -21.8%, 4.1%, and 22.1% (30AC, 138, and 44A, respectively), oligochaetes also accounted for greater than 80% of the total number of benthic macroinvertebrates.

The presence of mayflies (particularly *Hexage-*

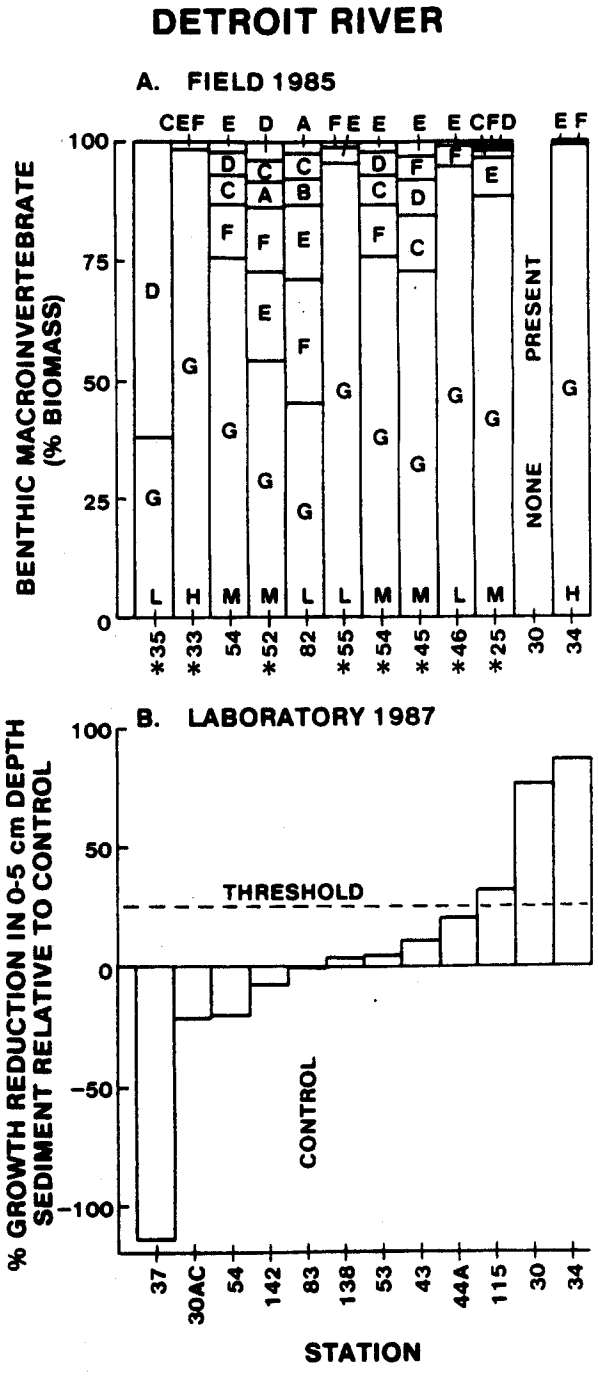

FIG. 2. Comparisons of field survey data (A) and **Chironomus tenans** *bioassay data (B) for surficial sediment (0–5 cm interval) from the Trenton Channel of the lower Detroit River. Station 137 is absent from the data sets. See Table 1 for letter definitions for benthic macroinvertebrate species. Total Biomass: L = 0–4.0 g/m, M = 4.1–9.0 g/m, H = >9.1 g/m. * = Station numbers for the respective field survey locations within corresponding sedimentation zones (Horvath and Kenega 1985).*

nia spp.), and sometimes caddisflies (Trichoptera), in synoptic surveys of Detroit River sediments is indicative of quality benthic invertebrate habitat (Thornley and Hamdy 1984, Hiltunen and Manny 1982). Macrozoobenthic biomass at station 83 was composed of 7.7% mayflies (2.1% *Hexagenia* spp.) and 5.4% caddisflies (Table 1). At station 142, *Hexagenia* spp. was 5.4% of the macrozoobenthic biomass while at other locations such as stations 115 and 34, where surficial sediments were classified as toxic based on *C. tentans* laboratory assays, no mayflies or caddisflies were observed. The survey of station 30 sediment did not recover any benthic macroinvertebrate organisms. We have used a value of 25% relative inhibition of *C. tentans* weight gain as a threshold marker for sediment toxicity and exclusion of pollution sensitive species as corroborated by the available field survey data.

Trenton Channel Sediment Core Toxicity Profiles

The use of surficial sediment sampling alone is limited in its ability to answer the questions necessary for the informed management of contaminated sediments. Measurements of the vertical extent of toxicity in sediments can be used to determine the potential toxic effect of resuspension events resulting from dredging operations. In addition, the historical record of contamination can be examined to determine the duration and extent of previous contamination events and estimate whether the observed effects are due to "new" or "old" contamination.

Sediments from stations 30, 34, and 137, where a uniform toxicity profile was observed at all depths surveyed (Fig. 3), suggest a continuing and current source of contamination to these sediments or recent mixing of sediments at these locations. In such a case as this, the management of continuing point-source pollutants should be the first priority in remediation. A toxicity profile of a steadily increasing pattern of toxicity with increasing depth could be due to chemical processes, or confirm the success of management strategies for the reduction of contaminant inputs or the redistribution of sediments within the system at any given depositional zone.

At stations 115, 43, 142, 30AC, and 53, the profiles peak in the level of toxicity at the mid-core intervals (5–10 cm or 10–15 cm) and then diminish with increasing core depth (10–25 cm) to levels near or below that of control (Fig. 3). For example, the surficial sediment from stations 115

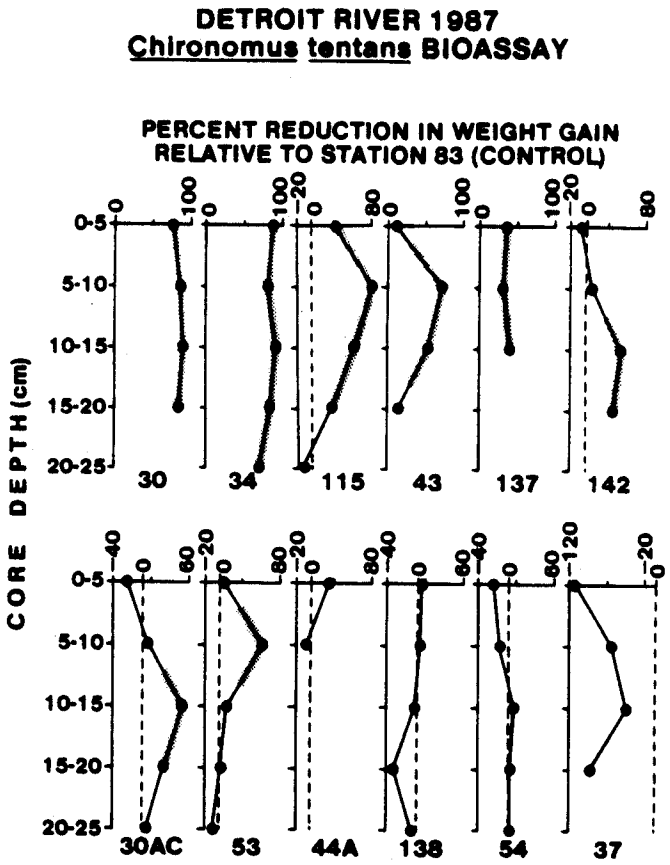

FIG. 3. Vertical toxicity profiles of sediment core depth vs. percent reduction in weight gain of C. tentans *relative to control. Highlighted lines profile growth inhibition greater than threshold toxicity (25%). Negative values represent a growth stimulus relative to control growth on surficial station 83 sediment.*

or 43 did inhibit growth of *C. tentans* relative to control, however, the sediments immediately below the surficial zone (5–10 cm) inhibited growth of *C. tentans* more than the surficial sediments (Fig. 3). Without sediment core toxicity data, the distribution of benthic macroinvertebrate organisms in surficial sediments at these locations would lead one to underestimate the toxic potential of the underlying sediments at these locations. The composition of benthic species at station 43 suggests a questionable habitat quality, yet still better quality than is suggested by the toxicity data for *C. tentans* at station 43 in 5–10 cm core depth sediment (Fig. 3). At stations 142 and 30AC the effect was even more dramatic. For example, a maximum effect on *C. tentans* of 46.7% reduction in weight gain was observed at station 142 at the 10–15 cm depth,

however, the overlying surficial sediment was non-toxic at the 0–5 cm (-6.9%) and 5–10 cm (8.9%) depths and supported a diverse community of macrozoobenthos. Therefore, at station 142 sediments that were determined to be toxic by the *C. tentans* bioassay were observed buried under as much as 10 cm of non-toxic sediments. At nine of the 12 locations, a pattern of diminishing toxicity with increasing depth suggests that a 25-cm core length was sufficient to describe the entire range of toxicity observed at those locations in the Trenton Channel.

Due to diagenetic and bioturbation processes, it is often very difficult to interpret the observed distribution of toxicity of contaminant concentrations and care must be taken in interpreting such results. However, we feel that coupled with sediment chemistry, bioassays can be used to successfully determine the extent of contaminated sediments for remedial action planning. With sufficient sampling, sediment core profiles of toxicity can provide data from which estimates of the spatial extent and volume of contaminated sediment in a region can be calculated. The required costs of dredging and volume of appropriate containment facilities can then be determined.

Remediation of Sediment Toxicity in the Trenton Channel

Because the presence of contaminated sediments in the Trenton Channel is confirmed and an adverse effect on the macrozoobenthic community has been observed, we calculated the volume of toxic sediment which would need to be removed to restore naturally reproducing, indigenous communities of benthic macroinvertebrates at those locations. In addition, the burial of toxic sediments in the Trenton Channel is not necessarily beneficial due to human activities and the natural erosional capacity of the river.

Outside the major sedimentation zones, the rate of sedimentation in the Trenton Channel is not great due to the channel configuration and the velocity of the flow (Kreis 1988). Suspended particles which enter the Trenton Channel have an average travel time of 8 h before they are deposited in the delta and the western basin of Lake Erie. Contaminants originating on the western channel shoreline remain isolated near that shore and decrease downstream as a toxic gradient (Giesy *et al.* 1988a, b). The total number of sedimentation zones in the channel is small because much of the channel bottom is composed of hard clay, rock or large gravel and cobble (Fig. 1). In waterways with more extensive or complex sedimentation, more sampling may be necessary to accurately ascertain the three-dimensional distribution of toxicity.

Maps of horizontal and vertical sediment toxicity were used to calculate the maximum and minimum sediment volume which would need to be removed from the Trenton Channel to improve the quality of the benthic habitat to a specified level. Previous studies of the toxicity of surficial sediment in the channel have indicated that the majority of these sediments have demonstrated deleterious effects on aquatic life (Kreis 1988). Therefore, it was reasoned that since the present study did not examine all the major sedimentation zones in the Trenton Channel, the optimal mass of contaminated sediment requiring removal from the channel would be between some minimum and maximum value. For example, a major sedimentation zone north of station 30AC (Fig. 1), approximately 2,100 m^2 in area, was not sampled in this study. If it had been included, it could have greatly affected the estimated total sediment mass that exceeded the critical threshold toxicity in this study.

The maximum volume of sediment which would need to be removed from the Trenton Channel is approximately 231,000 m^3, determined by hydrographic survey maps and calculation. This would remove all sediments causing a statistically significant inhibition of growth in *C. tentans*, relative to reference (control) sediment. This maximum volume includes sedimentation zones surrounding stations 115, 43, 53, 44A and 142 to depth of 0.5 m, a ribbon of sedimentation averaging 20 m wide and 0.5 m deep for the remaining eastern and western lengths of the Trenton Channel, and the submerged delta at Gibraltar Bay, Michigan, to a depth of 1 m. Dredging is limited in its ability to effectively remove less than some minimum depth of material. The 0.5-m dredge depths used in the calculations are approximately that minimum operational limit and constitute an over-estimate of the actual volumes of toxic sediment requiring removal.

The minimum volume of sediment which would need to be removed to restore quality benthic habitat to the Trenton Channel is approximately 4,246 m^3 (Table 2) which is based on known volumes of sediment within the zones tested. This calculation includes those sediments which were found to exceed the toxic threshold for the *C. tentans* bioassay, plus non-toxic sediments overlying toxic sedi-

TABLE 2. Minimum area, volume, and cost estimates for dredging toxic sedimentation zones in the Trenton Channel, having greater than a 25% reduction in weight gain by the C. tentans *bioassay.*

STATION	AREA (m^2)	DREDGE VOLUME (m^3)	COST[1] ($US)
137	588	294	$105,137.
115	780	390	$138,468.
30	540	270	$ 96,555.
30AC	687	344	$123,018.
34	519	260	$ 92,979.
43	1,268	634	$226,725.
142	2,855	1,428	$510,667.
53	1,251	626	$223,864.
Sediment removed from toxic sedimentation zones	TOTALS:	4,246	$1,518,413.

[1]Cost estimates based on a cost of $357.61/$m^3$ ($327/$yd^3$) for toxic sediment removal and confined disposal.

ments at stations 30AC, 142, and 53 (Fig. 3). This minimum volume is the total of the volumes for sedimentation zones represented by stations 137, 115, 30, 30AC, 34, 43, 142, and 53 to a sediment depth of 0.5 m (Table 2).

At the combined current dredging and disposal cost of $357.61/$m^3$ for contaminated sediment ($327/$yd^3$; personal communication with John Adams, U.S. Army Corps of Engineers, Buffalo, NY, 1988), the maximum cost of remediation is approximately $82.6 million. The cost to remove the minimum volume of toxic sediment would be approximately $1.5 million. These amounts are approximately 100 times greater than the costs for dredging and disposing of uncontaminated sediments, due to a confined disposal facility (CDF) required for storage of contaminated materials or sediments. The toxic nature of these sediments may also require specialized dredging operations (i.e., non-overflow hydraulic removal) which would reduce the resuspension of sediment, but also increase the costs of dredging. As a result, cost-effective remediation is best achieved by the removal of sediment that has been identified as contaminated material.

More extensive sediment-core sampling of the channel would yield more accurate results, but we feel that this study provides an acceptable initial estimate of the mass of sediment that would need to be removed and the approximate cost. More importantly, this report presents an analysis of the data necessary for the process of effective remediation. In turn, the informed implementation of remedial actions can assist the recovery of the currently degraded benthic habitat of the lower Detroit River.

CONCLUSIONS

Laboratory bioassays of the effects of sediments and their associated contaminants on *C. tentans* can provide ecologically relevent information which can be used to identify the relative toxicity of sediments, set priorities, and guide remedial actions. Also, due to the large difference in the costs of dredging and storing contaminated sediments, bioassays are a cost-effective method for determining which sediments must be handled as toxic sediments and confined in specially designed containment facilities, and which sediments can be safely disposed of in a more conventional manner.

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