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Environmental Influences on Benthic Community Structure in a Great Lakes Embayment

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ABSTRACT. *Biological and chemical measurements of sediment are useful parameters when establishing long-term assessment and monitoring tools for designated areas of concern (AOCs) in the Great Lakes. An intensified Environmental Monitoring and Assessment Program (EMAP) sampling grid in the St. Louis River estuary of western Lake Superior was used to assess the relationship between surficial sediment characteristics and benthic community structure. Ninety sites within two habitat classes (< 5.5 m and > 5.5 m depth) were randomly sampled. Sediment for chemical analysis was collected with a cylindrical drop core while benthic macroinvertebrate abundance and composition were determined from petite Ponar grab samples. Taxa richness was variable (1 to 25 taxa) among sites in the St. Louis River AOC. Oligochaeta were the most abundant taxa while Chironomidae larvae provided a majority of the taxa richness with 43 genera. Results from multivariate redundancy analysis (RDA) on 13 environmental parameters revealed that the majority of variation in benthic community structure was attributed to water depth and site distance from the headwaters. Although physical habitat alterations occur over large spatial scales and are more subtle than those conditions associated with chemically impacted sites, only a small portion of the variability in benthic community structure was explained by sediment chemistry variables. Variability in benthic community structure during this survey was best explained by physical habitat features and must first be quantified prior to understanding benthic response to contaminated sediments.*

INDEX WORDS: *Benthic macroinvertebrates, contaminants, dredging, Lake Superior, sediment.*

INTRODUCTION

Benthic invertebrates in the Great Lakes Areas of Concern (AOCs; International Joint Commission (IJC) 1989) are frequently used to evaluate overall ecosystem "health" (Flint 1979, Reynoldson and Zarull 1989, Rosenberg and Resh 1993, Reynoldson *et al.* 1995) because these communities are important to material cycling and secondary production, and are sensitive to environmental contaminants. To fully understand the extent to which anthropogenic disturbances affect benthic community health and distribution, it is important to measure the environmental factors that provide the basic ecological template structuring the benthic community. By identifying important environmental parameters, assessment and remediation plans can focus on those disturbances which are anthropogenically derived, eliminating the influence from other regional factors.

Previous investigations of benthic communities in the Great Lakes region were designed to examine specific habitats, determine the distribution and abundance of specific benthic fauna (Schneider *et*

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al. 1969, Flint 1979), or demonstrate cause and effect relationships for heavily industrialized areas (Reynoldson and Zarull 1989). Harbor and embayment sediment studies frequently determine benthic community responses to a host of endpoints, including habitat disturbance (Zajac and Whitlatch 1982a,b), *in situ* exposure to sediments (Bergen *et al.* 1993), and bioaccumulation of contaminants (Nelson *et al.* 1995). These studies are often designed to depict differences between impacted and non-impacted sites. However, this design may not be suitable for regional characterizations, nor provide the information necessary for identifying confounding influences caused by other large-scale environmental inputs. Identifying environmental gradients acting across an area such as a Great Lakes AOC requires benthic assessments that match the spatial area of the region in question.

Recently, alternative methods for characterizing sediment quality guidelines by utilizing predictive relationships between benthic fauna and a suite of environmental parameters have been suggested (Reynoldson *et al.* 1995). Understanding the extent to which dominant physical and chemical sediment characteristics influence benthic community structure should provide results that are not only good indicators of change, but are applicable to other systems. Such studies require sampling the full range of environmental conditions available because the spatial patterns of aquatic organisms can be affected by a variety of factors including fluctuations in stream flow, temperature, light, resource availability (Niemi *et al.* 1990, Palmer and Poff 1997), species interactions (Peckarsky *et al.* 1997), and habitat heterogeneity (Palmer and Poff 1997, Levin 1992, Valett *et al.* 1997).

Interactions between benthic communities and various sediment parameters have been examined in nearshore or estuarine areas of the Great Lakes, but few studies have attempted to develop regional biological sediment guidelines for impacted areas (Norris and Reynoldson 1993). The St. Louis River AOC is located at the western end of Lake Superior and is one of the busiest shipping ports in the Great Lakes. As recommended by the IJC for Great Lakes AOCs, a Remedial Action Plan (RAP) has been in place since 1992 (MPCA/WDNR 1992) to evaluate impaired uses in the lower St. Louis River and harbor. Contaminated sediments have contributed to fish consumption advisories, benthic community degradation, and dredging restrictions. Previous studies conducted at depositional sites within the lower St. Louis River AOC noted areas of degraded sediment quality based on chemical and toxicological analysis (Schubauer-Berigan and Crane 1996, 1997) and the associated benthic community (Crane *et al.* 1997). Sediments associated with the St. Louis River AOC were evaluated during *in situ* exposures with *Lumbriculus sp.* (Monson *et al.* 1995) and sampled to determine benthic community response (Crane *et al.* 1997). Although the interactions between indigenous organisms and underlying sediments are important for comparing habitat conditions, generating biological impact statements, or establishing sediment quality guidelines, these data provide limited information that identifies the important environmental parameters within the AOC, and the extent to which these parameters influence freshwater benthic communities. The objectives of this study were to: 1) conduct a systematic survey to identify the environmental parameters that have the greatest influence on benthic community structure, and 2) isolate specific biotic variables that best indicate changes occurring within the benthic community.

METHODS

Study Area

The St. Louis River is a fifth order river that lies on the western most expanse of Lake Superior (Fig. 1). The river drains a 5,861 km^2 watershed, following a 328 km course through northern Minnesota and northwest Wisconsin. The St. Louis River is the second largest tributary to Lake Superior, and the confluence forms a freshwater estuary that is approximately 4,856 ha. The terminus of two natural sand bars, one adjacent to Superior Bay (Minnesota Point) and the other along Allouez Bay (Wisconsin Point), form a natural opening to Lake Superior. An additional opening, a channel constructed through the base of Minnesota Point, provides easier access to slips in the northern section of the harbor. A biomonitoring study was conducted, incorporating sediment chemistry analyses with a benthic community survey to determine the dominant environmental factors influencing benthic community structure. This portion of the project focused on a series of randomized sites beginning below an upstream reservoir, extending throughout the estuary, and concluding at the openings to Lake Superior.

Site Location

Site selection followed a randomized sampling design based on the format used in the Great Lakes Environmental Monitoring and Assessment Pro-

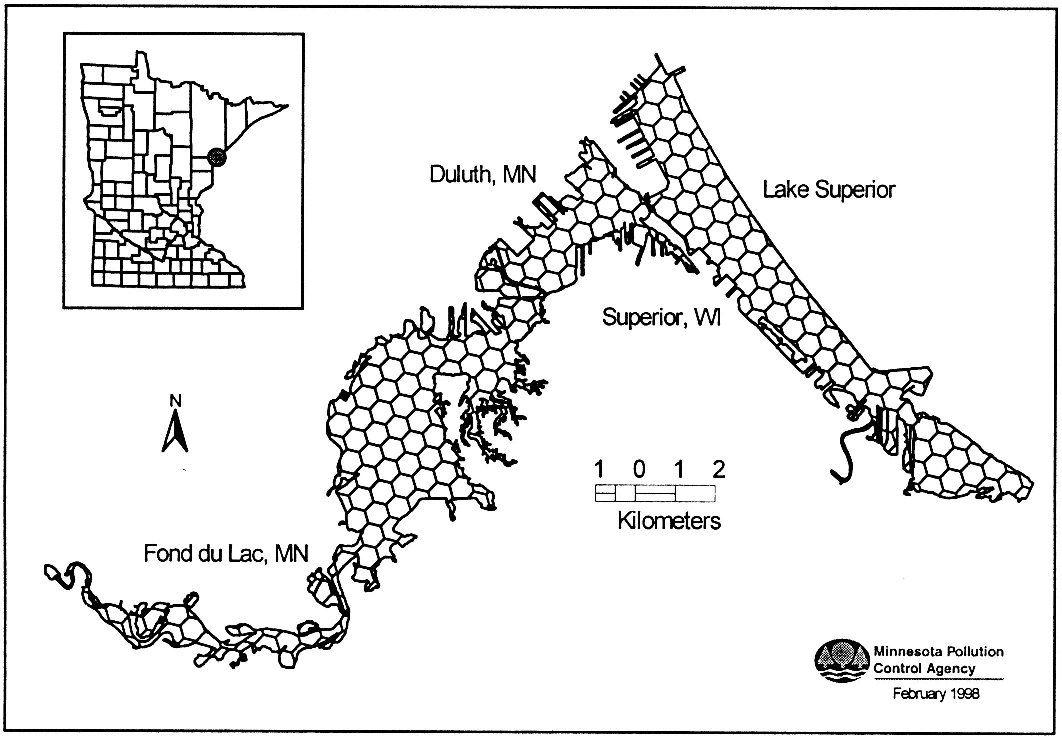

FIG. 1. a) Schematic diagram depicting the St. Louis River area of concern (AOC) within the Great Lakes. b). The St. Louis River AOC overlain with the hexagonal grid system used to select random sample locations.

gram (EMAP) system (U.S. EPA 1993). Accessible areas were digitally categorized as shallow, class 1 (water depth < 5.5 m), or channelized, class 2 (> 5.5 m in depth) habitats based on National Oceanic and Atmospheric Administration (NOAA) shipping charts. A 7^2-fold enhancement of the Great Lakes EMAP grid was applied to the accessible areas and the number of sites evaluated within each habitat class was based on the total surface area represented by that particular class. Ninety sample site locations were randomly selected from the resulting hexagonal grid, with each site positioned by a latitudinal/longitudinal coordinate (Fig. 1). Class 1 habitats (shallow) received 60 randomized sites, and class 2 habitats (channels) received 30 sites. To aid in summarizing data, site locations within the St. Louis River AOC were grouped, upstream to downstream, into areas noted as: 1) St. Louis River, 2) Spirit Lake, 3) St. Louis Bay, 4) Superior Bay, and 5) Allouez Bay (Fig. 2).

All samples were collected during a 30-day period, beginning in June, 1995. A research vessel was equipped with a Global Positioning System (GPS) to locate sites to within 30 m of the predetermined coordinate and provide post-survey differential correction. The research vessel position was held constant by anchor during sample collection. From the original 90 sample locations, sediment chemistry samples were obtained and analyzed for 87 sites, including 58 shallow and 29 channelized habitats. Benthic community samples were obtained from 89 of the 90 locations.

Habitat Parameters

In addition to collecting benthic and sediment samples, water depth and sediment depth were measured in the field, and site distance from the headwaters was generated in the laboratory. Water depth measurements were recorded at the predetermined coordinate using sonar. To determine sediment depth, a circular shoe (25 cm dia.) attached to the outside of a sampling rod (3.75 cm dia.) was used to determine the sediment/water interface. The rod was forced into the sediment to the point of resistance and a depth measurement was recorded. The difference between the sediment surface layer and

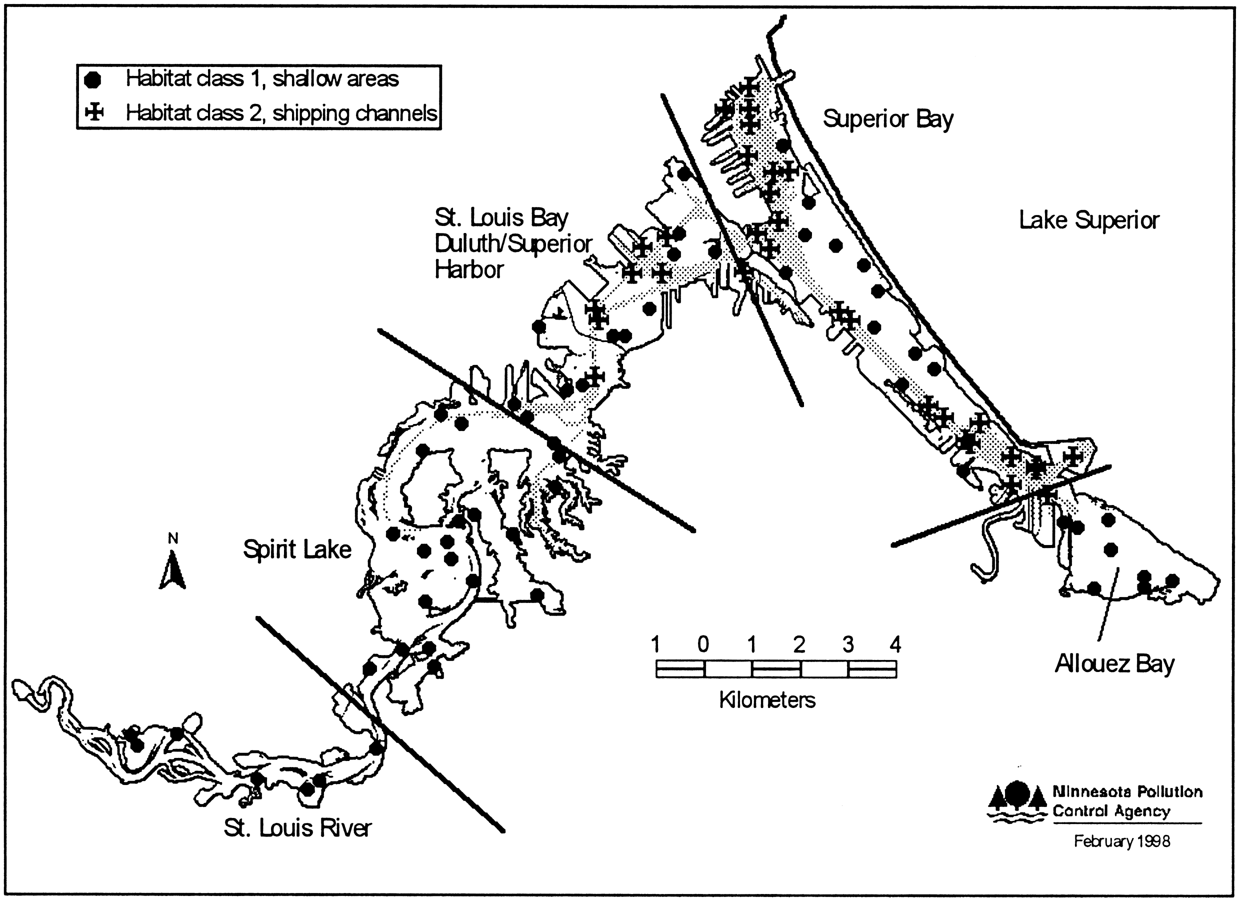

FIG. 2. The St. Louis River area of concern (AOC) longitudinally delineated into distinct habitat areas based on ecological conditions and anthropogenic modification.

the point of maximum resistance was used to estimate sediment depth. Study site distance downstream from the headwaters (longitudinal location) was determined from U.S. Geological Survey quadrangle maps (1:24,000 scale) and NOAA shipping charts (1:15,000 scale).

Chemical / Physical Sediment Parameters

Sediment samples were collected using a cylindrical (48 cm × 4.1 cm dia.) drop core (Wright 1980). A 2 liter composite sample, including the top 5 cm sediment layer from several cores, was homogenized and appropriate aliquots were individually stored for future physical and chemical analysis. Samples were immediately placed on ice prior to permanent storage (4°C). The physical and chemical parameters that were measured are listed in Table 1.

Sediment samples were analyzed for seven particle size classes based on distributions given by Folk (1980). Excess water was removed from the samples and the contents were homogenized using a jar mill. Three subsamples were removed with a sample thief, or spatula, and wet weights were recorded. One sample was dried (105°C) to a constant weight to determine moisture content. A dispersal agent (0.2% $^{w}/v$ sodium polyphosphate in ultra-pure water) was added to the remaining samples, which were then shaken briefly by hand. Size separation was achieved by passing each sample through a 1 mm sieve followed by elutriation through a 63 µm sieve. Sediment particles were continuously suspended by pumping air into the elutriation sieve until the content was clear. The 1,000-63 µm fraction retained in the elutriation sieve was transferred to a sample bottle and the excess dispersant was removed by pipette. Solids

TABLE 1. Environmental variables collected during the St. Louis River AOC survey.

			SEDIMENT CHEMISTRY		
FIELD MEASUREMENTS			PAH analytes (µg/g)	Metals	
Latitude/Longitude	lat/lon	degrees			
Longitudinal location	long loc	km	Napthalene	Cadmium	µmol/g
(Distance from headwaters)			Acenaphthylene	Copper	µmol/g
Water depth	watz	m	Fluorene	Nickle	µmol/g
Sediment depth	sedz	m	Phenanthrene	Lead	µmol/g
			Anthrocene	Zinc	µmol/g
			Fluoranthene	Total SEM	mg/kg
SEDIMENT SIZE CLASSES			Pyrene	Mercury	mg/kg
			Benz(a)anthracene		
Median diameter	meddia	µm	Chrysene		
Gravel, course sand	gvcs%	> 1,000 µm	Benzo(b)fluoranthene		
Course sand	csnd%	500–1,000 µm	Benzo(k)fluoranthene	OTHER MEASUREMENTS	
Medium sand	msnd%	250–500 µm	Perylene		
Fine sand	fsnd%	120–250 µm	Indeno(1,2,3-cd)pyrene	Ammonium	mg/kg
Very fine sand	vfsd%	63–120 µm	Dibenzo(a,h)anthracene	TOC	%
Silt	silt%	< 63 µm	Benzo(g,h,i)perylene		
			Total PAHs		
			Screening PAHs		

were covered with a 75% (^{v}/v glycerol) dispersant solution. Solids from this fraction were then suspended by gently swirling the mixture to avoid creating air bubbles, and transferred to a Horiba LA-900 Laser-Scattering Instrument. This instrument provided detailed analysis of the 1,000-63 µm fraction (Table 2). The instrument's performance was checked with a silica subsample (Fisher S150-3).

Sediment chemistry parameters were chosen due to elevated concentrations observed in the St. Louis River AOC (MPCA/WDNR 1992). The difference between simultaneously extracted metal (SEM, sum of Zn, Cu, Cd, Ni, Pb concentrations in µmol/g) and acid volatile sulfides (AVS) concentrations were used to evaluate the potential for metal availability. SEM-AVS values for determining bioavailability of metals to benthic organisms has been well documented (Di Toro *et al.* 1990, Ankley *et al.* 1994, Casas and Crecilius 1994, and Hansen *et al.* 1996). Percent total organic carbon (TOC), KCl extractable ammonium, and mercury (Hg) concentrations also were measured. Analytical procedures for AVS, SEM, KCl extractable ammonium, and Hg measurements followed standard operating procedures (SOP) described by Owen and Axler (1995). Gas chromatography/mass spectrometry (GC/MS) was used to examine concentrations of seventeen polycyclic aromatic hydrocarbons (PAH) at forty-two of the ninety sites (Moser and Lodge 1996). In addition, a screening PAH (fluorescence) procedure (Filkins 1992, Owen *et al.* 1995) was performed on sediment from all sites. Total PAH values from GC/MS data were regressed against PAH fluorescence values associated with the same site, providing predicted total PAH values for those sites having no GC/MS data.

$$\log \text{GC/MS} = B_0 + B_1 (\log \text{fluorescence}) \qquad (1)$$

where $B_0 = -1.02$, and $B_1 = 0.89$. The regression was significant ($p < 0.05$) with a $R^2 = 0.83$.

Benthic Sampling

Benthic macroinvertebrates were sampled in triplicate using a 0.023 m^2 petite Ponar grab sampler. Samples were rinsed through a 500 µm mesh sieve and the remaining debris was preserved in 10% formalin (^{v}/v). Samples were washed in the laboratory and re-preserved in 70% ETOH (^{v}/v) for long-term storage. Preserved samples were stained with Rose-Bengal solution and hand picked using a 2× magnification lens. A dissecting microscope was used to identify individual organisms to the lowest taxonomic level (Hilsenhoff 1981, Wiederholm 1983, Merritt and Cummins 1996). Midge larvae (Chironomidae and Ceratopogonidae: Diptera) were permanently slide mounted with euparal mounting

TABLE 2. Mean physical variables grouped by longitudinal location and habitat class within the St. Louis River AOC. Values are means ± 1 standard error. Physical variable abbreviations are listed in Table 1.

Habitat Area		Water depth (m)	Sediment depth (m)	Median dia. (µm)	gvcs% > 1,000 µm	csnd% 500–1,000 µm	msnd% 250–500 µm	fsnd% 120–250 µm	vfsd% 63–120 µm	silt% < 63 µm
St. Louis River	mean	1.6	3.3	99.8	0.5	1.5	17.7	7.7	7.6	65.0
	std. err.	(0.2)	(0.6)	(38.2)	(0.2)	(0.6)	(10.4)	(2.7)	(4.1)	(10.8)
	n = 9									
Spirit Lake	mean	1.7	3.1	96.4	3.3	4.6	11.8	12.5	5.7	62.0
	std. err.	(0.2)	(0.8)	(15.1)	(1.3)	(1.9)	(2.6)	(3.6)	(1.1)	(6.6)
	n = 18									
St. Louis Bay	mean	4.0	4.8	131.2	1.5	6.4	21.1	8.6	4.5	57.9
	std. err.	(0.8)	(1.7)	(27.7)	(0.7)	(1.6)	(6.0)	(2.2)	(0.8)	(7.5)
	n = 18									
Superior Bay	mean	6.2	4.1	154.2	0.5	6.1	22.7	20.8	5.2	44.6
	std. err.	(0.6)	(1.0)	(18.8)	(0.1)	(1.2)	(3.6)	(3.8)	(0.8)	(6.5)
	n = 33									
Allouez Bay	mean	2.3	3.9	153.1	6.9	9.0	18.3	12.7	6.2	46.9
	std. err.	(0.9)	(1.8)	(38.6)	(2.9)	(2.2)	(3.6)	(3.2)	(1.8)	(9.8)
	n = 9									
All Habitats	mean	4.0	3.8	131.8	2.0	5.8	19.2	14.4	5.5	53.2
	std. err.	(0.4)	(0.5)	(11.2)	(0.5)	(0.7)	(2.2)	(1.8)	(0.6)	(3.6)
	n = 87									

Habitat Classes		Water depth (m)	Sediment depth (m)	Median dia. (µm)	gvcs% > 1,000 µm	csnd% 500–1,000 µm	msnd% 250–500 µm	fsnd% 120–250 µm	vfsd% 63–120 µm	silt% < 63 µm
Shallow < 5.5 m	mean	1.7	3.8	123.0	2.3	5.4	19.0	12.6	5.4	55.4
	std. err.	(0.1)	(0.5)	(12.9)	(0.6)	(0.9)	(2.8)	(1.9)	(0.8)	(4.3)
	n = 58									
Channel > 5.5 m	mean	8.3	N/A	151.4	1.3	6.6	19.6	18.0	5.6	48.9
	std. err.	(0.3)	N/A	(21.3)	(0.7)	(1.4)	(3.6)	(3.8)	(0.8)	(6.5)
	n = 29									

medium (BioQuip Products, Inc., Gardena, CA 90248) and identified to genus using a compound microscope.

Invertebrate data were used to create a set of twenty derived variables based on specific trophic level associations, functional feeding groups, behavioral mechanisms, and taxonomic categories (Table 3). Invertebrate classifications were based on information from ecological texts (Hilsenhoff 1982, Brinkhurst 1986, Pennak 1989, Thorp and Covich 1991, Merritt and Cummins 1996). These categorizations are frequently employed to examine the structure of benthic communities and to provide a means for understanding functional relationships to environmental conditions that are not inherent in pure taxonomic classifications. In a preliminary multivariate analysis, the strength of environmental and invertebrate associations were compared between derived biological variables and raw species abundances, and the identification of important environmental variables were found to be very similar.

Statistical Analysis

The number of environmental variables was reduced by conducting a Spearman rank order correlation analysis (SYSTAT 1992) and eliminating highly correlated parameters ($r \geq 0.75$). An easily identifiable parameter was chosen to represent a group of highly correlated values. For example, total SEM-AVS values were highly correlated with Zn, Hg, Cu, Cd, Ni, and Pb ($r \geq 0.75$). Consequently, the individual metal concentrations were eliminated from further analysis. The environmental variable data set was reduced from 40 to 13 parameters (Tables 2 and 4).

Relationships were examined between the benthic invertebrate variables and the chemical and physical sediment data sets using Redundancy Analysis (RDA), a canonical extension of principal component analysis. Redundancy analysis is a form of direct gradient analysis (ter Braak and Prentice 1988) that describes variation between two multivariate data sets. Specifically, a matrix of predictor variables (physical and chemical sediment variables) is used to quantify variation in a matrix of response variables (benthic community matrix). Richards *et al.* (1996) has used this technique to describe the variability in the benthic community structure within streams draining large agricultural watersheds.

In RDA, the site scores from a principal component analysis (PCA) are regressed iteratively against a set of environmental variables, and the fitted values of the regression become new site scores (Jongman *et al.* 1987). The PCA is thus constrained by the environmental or predictor variables. The RDA was performed using the program CANOCO (ter Braak 1991). Two important outputs from this method are: 1) the interset correlations of environmental variables with the RDA axes which identify environmental variables that have the strongest influence on the ordination; and 2) the fraction of total variance for each predicted variable explained by the RDA axes (Jongman *et al.* 1987). In RDA, the explained variance is derived from the sum of squares of the regression (ter Braak 1991).

Monte Carlo permutation tests were used to determine the statistical validity of the associations between the biological and environmental data sets. These tests were conducted by randomly permuting the sample numbers within the environmental variables. The environmental variables (or predictor variables) were then randomly linked with the species (or predicted variable) data, and a new ordination was calculated. This procedure was repeated 100 times to develop a population of eigenvalues. If the species variables respond to the environmental variables, then the test statistic calculated from the observed data will be larger than the data derived from most of the random simulations. An association was considered significant if the observed eigenvalue was within the five largest simulated values ($p < 0.05$).

RESULTS

Sediment Characteristics

Physical measurements on the St. Louis River AOC included eleven parameters consisting of water and sediment depth, river distance from the headwaters, and sediment size classes (Table 2). Site conditions ranged in water depth from 0.73 to 11.1 m with a clear demarcation between shallow habitats (mean of 1.74 ± 0.1 m) and channelized areas (mean of 8.3 ± 0.3 m).

Some upstream to downstream trends were evident in physical variables among the site locations. Water depth was shallow upstream (St. Louis River and Spirit Lake) and deepest in the St. Louis Bay and Superior Bay regions (Table 2). Sediment characteristics were relatively similar among the locations, however, upstream locations had smaller mean particle diameters. Superior Bay and Allouez Bay had the lowest proportion of silt, and these lo-

TABLE 3. Benthic invertebrate variables* grouped by longitudinal location and habitat class within the St. Louis River AOC. Values are means ± 1 standard error.

Habitat Area		Ampodar No./m²	Chidae No./m²	Dipera No./m²	Ephera No./m²	Isooda No./m²	Molsca No./m²	Olieta No./m²	Triera No./m²	Total Ab No./m²
St. Louis River	mean	35.1	1,680.0	1,745.3	207.2	97.3	275.8	1,716.1	62.1	4,503.9
	std. err.	(29.9)	(346.9)	(359)	(122.1)	(90.7)	(104.9)	(526.0)	(17.1)	(1,067.9)
	n = 9									
Spirit Lake	mean	51.0	1,712.2	1,744.1	76.0	34.3	223.9	2,419.3	110.7	5,041.6
	std. err.	(50.2)	(399.7)	(403.2)	(13.8)	(32.6)	(43.2)	(1,329.5)	(40.4)	(1,570.7)
	n = 18									
St. Louis Bay	mean	8.0	1,022.6	1,054.5	31.1	4.8	390.6	2,493.5	28.7	5,387.2
	std. err.	(6.5)	(152.5)	(152.8)	(14.1)	(4.8)	(126.8)	(407.1)	(8.0)	(815.7
	n = 18)									
Superior Bay	mean	0.0	1,003.5	1,037.8	10.0	0.0	320.4	2,710.1	9.6	4,473.6
	std. err.	(0.0)	(180.3)	(184.1)	(3.8)	(0.0)	(66.6)	(376.5)	(2.8)	(555.4)
	n = 33									
Allouez Bay	mean	6.4	857.7	886.4	236.0	3.2	141.9	1,038.2	43.0	2766.4
	std. err.	(4.2)	(242.8)	(254.6)	(64.5)	(3.2)	(78.8)	(387.9)	(16.0)	(868.5)
	n = 9									
All Habitat	mean	16.7	1,209.0	1,244.9	71.3	18.7	291.6	2,329.3	43.3	4,606.7
	std. err.	(11.0)	(122.4)	(124.3)	(15.9)	(11.0)	(40.0)	(328.8)	(9.7)	(445.1)
	n = 87									
Habitat Classes		**Ampodar No./m²**	**Chidae No./m²**	**Dipera No./m²**	**Ephera No./m²**	**Isooda No./m²**	**Molsca No./m²**	**Olieta No./m²**	**Triera No./m²**	**Total Ab No./m²**
Shallow < 5.5 m	mean	24.7	1,299.2	1,336.0	104.6	27.7	207.8	1,804.3	58.6	4,264.1
	std. err.	(16.5)	(165.7)	(168.1)	(22.8)	(16.5)	(26.8)	(445.7)	(14.1)	(603.8)
	n = 58									
Channel > 5.5 m	mean	0.0	1,028.6	1,062.7	5.9	0.0	460.2	3,379.4	12.9	5,291.7
	std. err.	(0.0)	(159.8)	(163.2)	(1.8)	(0.0)	(100.4)	(366.4)	(4.1)	(569.6)
	n = 29									

(Continued)

TABLE 3. *Continued.*

Habitat Area		Carnivore % Abundance	Detritivore % Abundance	Herbivore % Abundance	Omnivore % Abundance	Erosional % of Taxa	Depositional % of Taxa	Burrowing Taxa	Climbing Taxa	Clinging Taxa	Sprawling Taxa	Total Taxa
St.	mean	3.3	77.6	2.7	16.5	10.1	51.9	5.4	1.8	1.9	3.5	13.5
Louis	std. err.	(1.3)	(4.3)	(1.0)	(2.9)	(1.6)	(3.9)	(0.4)	(0.2)	(0.7)	(0.6)	(1.6)
River	n = 9											
Spirit	mean	2.1	82.2	1.8	11.5	10.8	47.5	5.3	1.3	1.7	2.3	11.5
Lake	std. err.	(0.6)	(3.9)	(0.6)	(1.8)	(1.4)	(2.7)	(0.4)	(0.1)	(0.2)	(0.3)	(1.0)
	n = 18											
St.	mean	3.3	82.9	2.2	12.7	9.4	51.3	4.6	1.1	0.8	2.1	9.1
Louis	std. err.	(1.3)	(2.7)	(0.7)	(2.0)	(1.3)	(2.0)	(0.3)	(0.1)	(0.1)	(0.3)	(0.7)
Bay	n = 18											
Superior	mean	2.1	79.2	1.2	17.6	11.8	46.8	3.6	1.1	0.8	2.2	8.1
Bay	std. err.	(0.4)	(2.5)	(0.3)	(2.2)	(1.0)	(1.2)	(0.0)	(0.1)	(0.1)	(0.2)	(0.5)
	n = 33											
Allouez	mean	6.6	66.7	6.4	24.8	9.0	55.7	4.0	0.7	1.2	2	8.4
Bay	std. err.	(2.6)	(6.1)	(2.8)	(3.4)	(1.9)	(3.5)	(0.6)	(0.2)	(0.5)	(0.3)	(1.4)
	n = 9											
All	mean	2.9	79.1	2.3	15.7	10.6	49.3	4.4	1.2	1.2	2.3	9.6
Habitats	std. err.	(0.5)	(1.7)	(0.4)	(1.1)	(0.6)	(1.0)	(0.2)	(0.1)	(0.1)	(0.1)	(0.4)
	n = 87											
Habitat Classes		**Carnivore % Abundance**	**Detritivore % Abundance**	**Herbivore % Abundance**	**Omnivore % Abundance**	**Erosional % of Taxa**	**Depositional % of Taxa**	**Burrowing Taxa**	**Climbing Taxa**	**Clinging Taxa**	**Sprawling Taxa**	**Total Taxa**
Shallow	mean	3.4	76.8	2.7	17.0	10.9	49.3	4.7	1.2	1.4	2.3	10.2
< 5.5 m	std. err.	(0.5)	(2.1)	(0.5)	(1.6)	(0.7)	(1.4)	(0.2)	(0.1)	(0.1)	(0.2)	(0.5)
	n = 58											
Channel	mean	2.1	83.7	1.4	13.3	10.0	49.3	3.7	1.1	0.7	2.3	8.2
> 5.5 m	std. err.	(0.8)	(1.8)	(0.4)	(1.4)	(1.3)	(1.4)	(0.2)	(0.1)	(0.1)	(0.2)	(0.5)
	n = 29											

*Amphipoda, Chironomidae, Diptera, Ephemeroptera, Isopoda, Mollusca, Oligochaeta, Trichoptera, and Total Abundance are in number of individuals/m^2. Carnivores, Detritivores, Herbivores, and Omnivores are designated as % of total abundance. Erosionial and Depositional are % of taxa richness associated exclusively with erosional or depositional habitats. Burrowing, Climbing, Clinging, and Sprawling Taxa are behavioral attributes (Merritt and Cummins 1996).

TABLE 4. Mean sediment chemistry parameters grouped by longitudinal location and habitat class within the St. Louis River AOC. Values are means ± 1 standard error.

Habitat Area		Predicted PAH (µg/g)	SEM/AVS value	TOC %	KCl Ammonium (mg/kg)
St. Louis River	mean	3.57	–3.60	3.05	2.50
	std. err.	(0.75)	(2.61)	(0.54)	(0.52)
	n = 9				
Spirit Lake	mean	3.46	–2.91	5.16	2.23
	std. err.	(1.13)	(1.79)	(1.11)	(0.38)
	n = 18				
St. Louis Bay	mean	12.47	–0.97	3.20	5.49
	std. err.	(3.67)	(0.47)	(0.71)	(2.63)
	n = 18				
Superior Bay	mean	3.47	–0.52	1.83	12.13
	std. err.	(0.56)	(0.78)	(0.27)	(4.71)
	n = 33				
Allouez Bay	mean	5.71	–0.06	7.37	2.76
	std. err.	(1.89)	(0.20)	(2.54)	(1.01)
	n = 9				
All Habitats	mean	5.57	–1.36	3.49	6.74
	std. err.	(0.93)	(0.55)	(0.43)	(1.93)
	n = 86				
Habitat Classes		**Predicted PAH (µg/g)**	**SEM/AVS value**	**TOC %**	**KCl Ammonium (mg/kg)**
Shallow < 5.5 m	mean	6.73	–2.07	4.42	3.10
	std. err.	(1.36)	(0.81)	(0.61)	(0.81)
	n = 58				
Channel > 5.5 m	mean	3.24	0.01	1.67	14.01
	std. err.	(0.48)	(0.11)	(0.22)	(5.32)
	n = 29				

cations, along with St. Louis Bay, had somewhat greater proportions of medium sand. The wider expanse and slower currents commonly experienced lower in the harbor may allow larger particles to settle out as finer particles are transported into Lake Superior (Keillor and Ragotzkie 1976). Sediment movement within the harbor has been primarily attributed to specific wind direction, with secondary suspension resulting from shipping traffic (Erdmann *et al.* 1994). Stortz and Sydor (1980) developed a hydrodynamic model for the Duluth/Superior Harbor and determined that coarse material resuspended by ship traffic settled rapidly, with only small sediment loads of the coarse material reaching Lake Superior. Besides the clear difference in depth between ship channels and non dredged habitats, no other distinct differences exist among sediment variables between shallow and channel habitats.

Sediment Chemistry

Sediment chemical concentrations were highly variable throughout the AOC and no upstream to downstream trends were evident. Total PAH concentrations were highest (54.03 µg/g) in St. Louis Bay, but concentrations were lower than concentrations reported at other industrialized harbors in the Great Lakes (Fallon and Horvath 1985). On average, lower PAH concentrations were observed at these randomized locations (Table 4) within the St. Louis River AOC than were reported during an earlier study on sediments in the harbor (Crane *et al.* 1997). Percent TOC averaged less than 4.0% for all locations and was highest (7.3%) in Allouez Bay. Mean KCl-extractable ammonium levels recorded within the St. Louis River AOC were relatively low, with Superior Bay reaching a mean of 12 mg/kg.

The mean total SEM-AVS value was low throughout the AOC. One site in St. Louis Bay had a value of 1.42, suggesting the AVS binding potential had been exceeded and metal bioavailability was possible. Individual concentrations of the five metals used to derive the total SEM-AVS value in this study (Zn, Cu, Cd, Ni, and Pb) were lower than the mean values reported during a Detroit River sediment study (Hamdy and Post 1985). Channel habitats in the St. Louis River AOC contained sites with positive total SEM-AVS values, although mean values were near zero (Table 4). Notable differences were found between shallow and channel habitats with respect to all chemical parameters. Shallow habitats had the highest mean concentrations for PAH and %TOC, and negative SEM-AVS values. Channel habitats contained the highest ammonium concentrations and, on average, near zero SEM-AVS values.

Benthic Invertebrates

Benthic invertebrate community abundance was highly variable throughout the St. Louis River AOC, with sample abundances between sites reaching orders of magnitude in difference. A similar finding was reported in a 1994 hotspot assessment of Duluth/Superior Harbor sediments (Crane *et al.* 1997) and during other Great Lakes benthic surveys (Reynoldson *et al.* 1989). Generic level identifications for most individuals resulted in 80 taxa, representing 18 orders and 34 families. Taxa richness ranged from 1 to 25 throughout the entire study area, with a mean of 9.6 ± 0.4 taxa per site. During a previous study in the Duluth/Superior Harbor (Crane *et al.* 1997), a similar range of taxa (1 to 18) was reported. However, more detailed, specific identifications would have increased mean numbers of taxa when compared to generic level identifications reported here. The comparatively lower taxa richness and high variability (8 ± 4 taxa per site) reported by Crane *et al.* (1997) may be explained by their sampling design, which targeted sites containing poor sediment quality.

About 85% of the taxa identified belonged to the class Insecta. Midge larvae (Chironomidae: Diptera) represented a majority of the total taxa richness with 43 genera. Oligochaeta worms (Enchytraeidae, Naididae, and Tubificidae: Annelida) were analyzed at the class level and represented the most abundant group. Tubificidae and Chironomidae also dominated the benthic fauna at contaminated sites in the harbor during an earlier study (Crane *et al.* 1997). Similarly, benthic studies in other Great Lakes AOCs (Buffalo River, NY; Indiana Harbor, IN; Saginaw River, MI) showed Oligochaeta and Chironomidae comprising over 90% of the invertebrate abundance collected in depositional areas (Canfield *et al.* 1996).

Mean taxa richness steadily declined along the upstream to downstream gradient, with the mean number of taxa at the upper most sites (13.5 ± 1.6) significantly greater than those values (8.1 ± 0.5) occurring in the lower portions of the harbor (Table 3). Ephemeroptera declined from a mean of 207.2 ± 122.1 /m^2 at the upstream sites to 10.0 ± 3.8 /m^2 in Superior Bay. Ephemeroptera abundances were also high in Allouez Bay (236.0 ± 64.5 /m^2), although numbers were highly variable. Trichoptera abundance increased from 62.1 ± 17.1 to 110.7 ± 40.4 /m^2 in the St. Louis River to sites in Spirit Lake, but declined to 28.6 ± 8.0 and 9.6 ± 2.8 /m^2 in the harbor and Superior Bay, respectively. Trichoptera abundances, like Ephemeroptera abundances, were also higher in Allouez Bay, reaching a mean of 43.0 ± 16.0 /m^2. Among *Caenis* sp. (Caenidae: Ephemeroptera), mean abundances fell steadily from 55.8 ± 28.6 /m^2 to 21.5 ± 5.0, 19.9 ± 12.1, and 2.6 ± 1.3 /m^2 from the upstream sites to Spirit Lake, St. Louis Bay, and Superior Bay sites, respectively. Similarly, *Hexagenia* sp. (Ephemeridae: Ephemeroptera) numbers, which constitute a large portion of Ephemeroptera sampled in the AOC, dropped from means of 141.9 ± 76.2 /m^2 in the upstream sites, to 50.2 ± 12.3 and 9.6 ± 5.6 /m^2 moving downstream to the estuary and harbor sites, respectively. *Hexagenia* sp. abundances were at a low of 7.4 ± 3.8 /m^2 in Superior Bay, but a notable exception occurred in Allouez Bay, where numbers reached a maximum (229.7 ± 62.8 /m^2). Allouez Bay is an embayment formed by the confluence of Superior Bay with the Nemadji River (Fig. 2). Although this habitat is located furthest downstream, its proximity to more riverine type habitats and potential sources of macroinvertebrate emigration, as

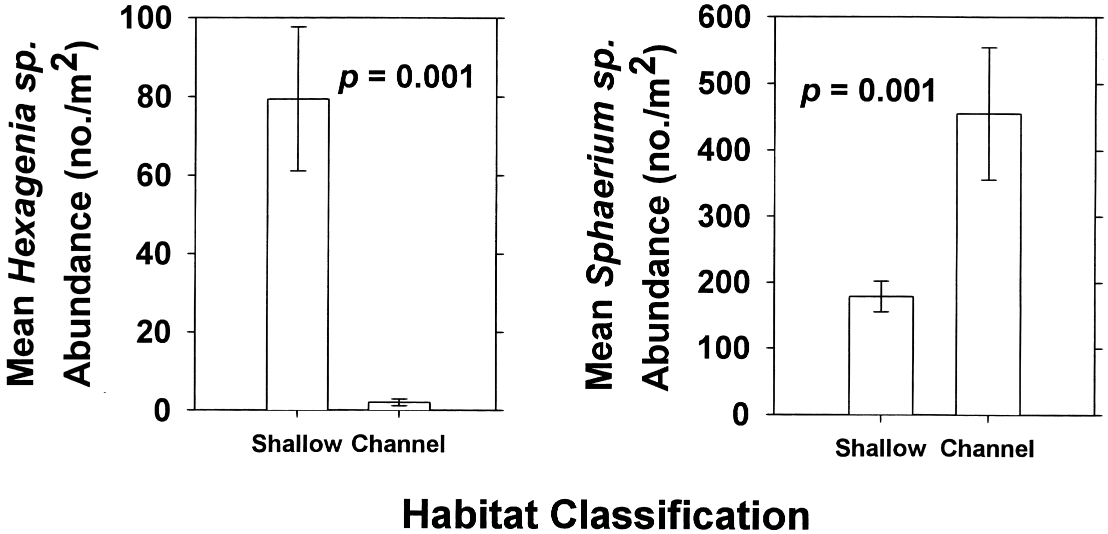

FIG. 3. The mean number of **Hexagenia** *sp. (Ephemeridae: Ephemeroptera) and* **Sphaerium** *sp. (Sphaeriidae: Mollusca) occurring at shallow and channelized habitats within the St. Louis River AOC. Bars represent the mean number of organisms per* m^2 *± 1 standard error. Value of p is from the overall ANOVA* ($p \leq 0.05$).

well as less industrial/ residential development, provides this area with characteristics more similar to the upper St. Louis River and Spirit Lake.

With few exceptions, most taxa decline in mean numbers when comparing shallow and channelized habitats (Fig. 3). The exceptions include *Sphaerium* sp. (Sphaeriidae: Mollusca) abundances, which nearly double from shallow to channel habitats (Fig. 3). Likewise, Oligochaeta abundance associated with channel habitats (3,379.4 ± 366.4 /m^2) was significantly greater than mean abundance (1,804.3 ± 445.7 /m^2) in shallow habitats (Table 3). However, some shallow habitat sites contained Oligochaeta abundances (31,864 /m^2) that far exceeded maximum values associated with channelized areas (10,851 /m^2).

RDA Analysis

The 13 environmental parameters (predictor variables) explained 28% of the total variation associated with the benthic community variables (response variables). Total variation explained for the benthic variables ranged between 11 and 43% (Table 5). An identical test was conducted using the same environmental parameters and 80 individual macroinvertebrate taxa. The data set using more specific identifications was used to determine if subtle associations could be better identified, as opposed to that explained by the higher systematic categories. The explained variability associated with individual taxa reached 24%. Overall, each ordination axis was driven by the same environmental parameters and the variation explained by the individual taxa ranged between 8 and 45%. A Monte Carlo permutation test of both ordinations was significant at $p = 0.05$. Water depth and distance downstream from the headwaters were the two redictor variables that explained most of the variance associated with the benthic community. These environmental parameters had strongest correlations with the first RDA axis (Table 5).

Eleven out of the 20 benthic invertebrate variables had the majority of their explainable variation described by the first axis (Table 5). Seventy-nine percent of the explainable variation in taxa richness was associated with the first axis (Table 5), and variability in mayfly (Ephemeroptera) and caddisfly (Trichoptera) abundance also was best described by the first axis (86 and 84% of explainable variation, respectively). These results suggest that community structure responds to subtle changes along the upstream/downstream gradient, and that more pronounced physical conditions, such as water depth, are important factors associated with benthic varia-

TABLE 5. Percent of explainable variance described on the first three RDA axes for each benthic variable. The correlation of environmental variables best describing each RDA axis are listed above each column. The total variance explained by the RDA for each variable is listed in the last column.

Benthic Variables	Measurement	RDA AXIS 1 Water Depth (501) Long Loc. (445)	RDA AXIS 2 SEM-AS (283) LONG. LOC. (241)	RDA AXIS 3 Fsnd % (223) SEM-AVS (–284)	% Total Variance Explained
Amphipoda	Abundance	55.79	0.73	32.39	30.04
Chironomidae		69.88	20.77	0.13	23.64
Diptera		70.10	21.93	0.00	23.98
Ephemeroptera		86.11	2.38	2.22	43.34
Isopoda		50.65	9.02	25.64	31.59
Mollusca		9.11	18.35	25.75	15.26
Oligochaeta		17.85	70.72	3.62	43.13
Trichoptera		83.85	0.12	2.64	40.92
Total Abundance		0.12	83.15	2.78	25.87
Carnivores	%Abundance	26.68	39.14	3.54	18.93
Detritivores		52.28	44.97	1.01	35.71
Herbivores		3.72	61.40	2.13	25.78
Omnivores		54.29	42.08	0.92	37.10
Erosional	% of Taxa	6.10	3.27	53.59	17.71
Depositional		13.58	0.37	5.02	10.75
Burrowers	Taxa	76.47	7.97	5.08	28.35
Climbers		25.55	48.74	6.13	24.15
Clingers		86.58	0.74	0.00	30.99
Sprawlers		17.99	39.41	2.17	21.62
Total Taxa		78.73	18.00	1.39	32.39

tion. In general, benthic invertebrate variables decreased with increasing depth and increasing distance downstream.

The relationship between distance along the upstream/downstream gradient and community composition was more fully explored by regressing taxa richness against distance from headwaters (Fig. 4). Because most of the deep water in the study region exists in dredged channels in the lower half of the study region, deep water habitats were excluded from the analysis in order to concentrate on changes occurring along the upstream/downstream gradient. These data suggest that the variance explained by upstream/downstream position along the river course can be associated with habitat alterations occurring as the AOC changes from a riverine system in the headwaters to an embayment near Lake Superior. The importance of water depth comes mostly from the contrast between dredged and undredged regions in the lower portions of the AOC. Periodic dredging to maintain shipping channels occurs exclusively at sites lower in the AOC, which substantially alters natural water depths and disturbs natural habitat characteristics.

The second RDA axis was best explained by the variability associated with total SEM-AVS and site distance from the headwaters (Table 5). Four of the biological variables (total abundance, Oligochaete abundance, climbing taxa, % herbivores) had the majority of their explainable variation accounted for on this axis. Eighty-three percent of the explainable variation associated with total macroinvertebrate abundance was described on the second axis. Because Oligochaeta worms were the most common and abundant taxa, their numbers closely relate to the total abundance variable (Table 3). Over 70% of the explainable variation in Oligochaeta abundance was attributed to the second RDA axis. Taxa behaviorally defined as climbers had over 48% of their explainable variation accounted for on the second RDA axis. Similarly, the variance within the benthic community trophic structure, both herbivores and carnivores, also were best described

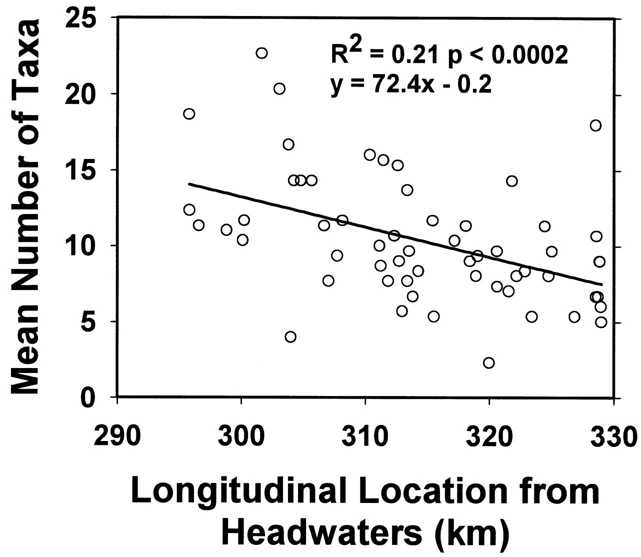

FIG. 4. Mean number of macroinvertebrate taxa occurring in shallow habitats along the upstream / downstream gradient within the St. Louis River area of concern (AOC). Taxa richness is negatively related (slope = – 0.198, p = 0.0002) to distance from the headwaters.

on the second axis (61 and 39% explained, respectively).

The third RDA axis was best described by a decreasing SEM-AVS value and fine sand (Table 5). The benthic community variable best explained on this axis was the percent of erosional taxa. Amphipoda, Isopoda, and Mollusca also had a large portion of their explainable variation described on the third axis. From the benthic invertebrate data available, Amphipoda, Isopoda, and Mollusca appear to be more closely associated with sediment chemistry variables than any other invertebrates.

DISCUSSION

Physical disturbances and natural environmental gradients were the most important factors regulating the abundance of benthic invertebrates in the St. Louis River AOC. Physical disturbance in lotic systems is inextricably linked to environmental dynamics and resulting ecosystem and biotic interactions (Powers *et al.* 1988, Resh *et al.* 1988). Similarly, benthos of larger embayments within the Great Lakes are strongly influenced by physical conditions (Rice and White 1987). In the St. Louis River AOC, dredging operations associated with shipping traffic cause a variety of physical and chemical disturbances that influence the distributions of aquatic organisms. Channels are dredged to maintain a 6 m depth for ship traffic, and these areas provide the only habitat deeper than two meters in the relatively shallow estuary. In addition, ship traffic and channel maintenance result in localized disturbance and increased sediment deposition in shallow areas adjacent to channels (Erdmann *et al.* 1994). These influences are superimposed on seasonal fluctuations in biota, discharge, and flow dynamics of the river. This study indicates that channelized habitats have more impoverished benthic communities than non channelized areas based on lower species richness and lower total abundance. These differences in abundance have also been shown to occur seasonally throughout the harbor (Swanson 1999). This occurrence is typical of fauna indigenous to disturbed lotic systems (Poff and Ward 1989), emphasizing the importance of dredging activities on benthic communities in the AOC.

The upstream to downstream gradient was an important parameter describing benthic community variation in the St. Louis River AOC. Site location along this gradient acted as a surrogate for several environmental factors within the AOC that were not directly measured during the study. For example, the St. Louis River immediately upstream from the study area contains reaches with high gradient, greater amounts of erosional substrate, and a greater diversity of riverine habitats than those found immediately downstream in the study area. These upstream areas act as a colonizing source for a variety of taxa occurring in the upper study reach that are not found in the lower portions of the study area because of the increased distance from the colonizing source. Consequently, higher taxa richness occurred in the most upstream portions of this study, partly as a result of the proximity of colonizing sources. This occurrence also has been reported in previous work that examined seasonal changes in benthic community structure in the St. Louis River AOC (Swanson 1999). Current velocity, although generally low throughout the study area, is somewhat higher in upper portions of the study area and favors the presence of some riverine taxa. Furthermore, the physical characteristics of the study region are complicated by seiche oscillations from Lake Superior. Seiche activity can cause periodic flow reversal within the study area, and may be more intense in downstream areas. Finally, commercial and residential development along shorelines and within the watershed is more prevalent when moving from upstream to downstream. Consequently, nonpoint source impacts associated with these types of land use patterns, as well as disturbances by recreational and commercial water craft, increase in a downstream direction and may contribute to environmental disturbances not measured in this study.

In addition to longitudinal location and depth, substrate particle size also had an influence on benthic macroinvertebrate distributions in the St. Louis River. The percentage of fine sand was important to the abundance of amphipods, isopods, and molluscs; however, this influence was small and difficult to separate from other sediment-related factors (Table 5). Some variation in substrate characteristics were observed, particularly in median particle size and percentage of silt, although these characteristics were not clearly related to macroinvertebrate distribution. Watershed development issues identified above may influence sediment characteristics. Sediment particle size (< 63 μm), and %TOC and other detrital materials have a large impact on abundance and distribution of macrobenthic fauna in other Great Lakes embayments (Schneider *et al.* 1969). In this study, the highest TOC levels were associated with the highest densities of *Hexagenia* sp. and *Caenis* sp. mayflies.

The relatively small proportion of variation in macroinvertebrate assemblages that could be accounted for by chemical-related sediment variables indicates that on a system-wide scale, these variables do not have a large influence in the St. Louis River estuary. This does not mean that contaminated sediments do not have localized influences on benthic organisms. Several studies have documented acute toxic effects of sediments on invertebrates in the St. Louis River estuary (Schubauer-Berigan and Crane 1997, Crane *et al.* 1997). However, these 10-day sediment toxicity tests revealed minimal impact at a majority of the sites sampled, with significant acute effects occurring at only a few contaminated locations.

Contaminated sediments are a major source of toxic substances and represent a continued risk to aquatic ecosystems within the Great Lakes (Landrum and Robbins 1990). Although there has been considerable progress in most RAPs, there is still a large gap in knowledge about the extent and magnitude of impact caused by contaminated sediments. In this study, a randomized design based on a systematic grid was used (Larsen *et al.* 1991, Messer *et al.* 1991) which allowed characterization of habitat constraints at both the local and the sub-basin level. Systematic field surveys examining macroinvertebrate communities are an essential risk assessment tool along with chemical analyses and laboratory toxicity tests. Knowledge of the important environmental constraints at several hierarchical levels will enhance the development of an effective survey design within the all of Great Lakes AOCs.

The spatial resolution mandated by the Regional Environmental Monitoring and Assessment Program (R-EMAP) sampling protocol had limited power to "see" local deleterious impacts. The hexagonal grids used in this sampling scheme were 0.267 km^2, which may not sufficiently match the spatial impact of local effluents or contaminated sediments in the harbor. Consequently, regional surveys, such as the present study, are not sufficient to identify "hot spots" for monitoring purposes. Such studies require more specific stratified study designs. Nonetheless, the R-EMAP approach does provide a means for depicting the dominant environmental influences on benthos in the region and establishes a benchmark for comparing data from site specific studies.

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