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A Comparison of Macroinvertebrates of Two Great Lakes Coastal Wetlands: Testing Potential Metrics for an Index of Ecological Integrity

Donna R. Kashian* **and Thomas M. Burton****

*Departments of Fisheries and Wildlife and **Zoology*
Michigan State University
East Lansing, Michigan 48824

ABSTRACT. *The macroinvertebrates of two northern Lake Huron wetlands were compared to assess water quality and test potential metrics for an Index of Ecological Integrity (IEI) for Great Lake coastal wetlands. Macroinvertebrates were collected using sediment coring and dip-net sampling monthly from June through September 1996. One wetland was impacted by domestic wastewater from a lagoon, urban storm-water runoff, and local marina traffic. A nearby wetland with a similar size drainage basin, no wastewater or urban storm-water input or marina traffic served as a reference. Greatest differences in chemistry between sites occurred during lagoon discharge in September. Compared to the reference, the impacted wetland had higher Cl, NH_4-N, NO_3-N, soluble reactive P, conductivity and lower dissolved oxygen levels. There were fewer insects, especially Ephemeroptera and Trichoptera in the impacted wetland than in the reference wetland. A greater proportion of macroinvertebrates in the impacted wetland were Amphipoda, Isopoda, and Naididae. Observed differences in macroinvertebrate communities were used to test 38 metrics, used in indices of biological integrity for streams, to determine their potential as metrics for an index of ecological integrity for Great Lake wetlands. Invertebrate attributes sensitive to water quality changes were identified as candidate metrics if they exhibited low within-site variability and detected differences between wetlands for each sampling period. Candidate metrics included relative abundance of Ephemeroptera, Isopoda, Trichoptera, predators, collector-filterers, and herbivore/detritivore ratio.*

INDEX WORDS: *Macroinvertebrates, wetland, marshes, metrics, index of biological integrity, monitoring, Lake Huron.*

INTRODUCTION

Coastal wetlands of the Great Lakes provide diverse habitats for animals and plants and trap allochthonous and autochthonous materials (Gaudet 1974, Wetzel and Allen 1972). The macroinvertebrates of coastal wetlands are important food resources and contribute to widespread use of wetlands as fish spawning (Jude and Pappas 1992) and waterfowl breeding areas (Prince and Flegel 1995). Despite the apparent importance of macroinvertebrates, comparatively little is known of their distribution and ecology in Great Lakes coastal wetlands or in freshwater wetlands in general (Krieger 1992). In order to monitor ecological integrity of these ecosystems, distribution and life history data for macroinvertebrates are needed. Macroinvertebrate community structure may provide a sensitive index to pollution inputs to coastal wetlands (Burton *et al.* 1999). The use of freshwater macroinvertebrates as indicators of water quality in streams is well established (Karr 1991, Rosenberg and Resh 1993). However, the macroinvertebrate fauna of Great Lakes coastal wetlands has received limited study (Krieger 1992), and no widely used macroinvertebrate system for measuring ecological integrity has yet been developed. The goal of this research is the development of such an index (Burton *et al.* 1999).

Benthic macroinvertebrate assemblages can be used as an economical monitoring system in place of or to complement chemical monitoring of water quality (Plafkin *et al.* 1989, Rosenberg and Resh 1993, Karr 1993). Continuous sampling of water chemistry is usually too expensive to be practical, and periodic sampling may not detect episodic pol-

*Corresponding author. E-mail: drkashian@students.wisc.edu
Present address: Dept. of Zoology, University of Wisconsin, Madison, WI 53706

lution. Benthic macroinvertebrates are continuously exposed to water quality, and their responses to pollutants are often cumulative over the life of a cohort. Thus, invertebrate-based indices are able to detect both episodic and cumulative releases of chemicals (Olive *et al.* 1988, Plafkin *et al.* 1989, Barbour *et al.* 1996, Fore *et al.* 1993).

Comprehensive multimetric indices (Barbour *et al.* 1996) have been widely adopted for stream monitoring since a fish-based index of biological integrity (IBI) was first proposed by Karr (1981). They are currently used or are being developed in 48 states (Karr and Chu 1997, Barbour *et al.* 1996, Gerritsen 1995, Southerland and Stribling 1995, Kerans and Karr 1994, Ohio EPA 1987). Their use in wetlands is also being investigated by many states with these efforts coordinated by the Bioassessment Wetlands Working Group (BAWWG) of the U.S. Environmental Protection Agency's Wetland Office in Washington, D.C. Their use in Great Lakes wetlands was also advocated by the wetlands task force of the State of the Lakes Ecosystem Conference (SOLEC) held in Buffalo, NY in 1998.

Multimetric indices may encompass several attributes of the sampled assemblage, including taxa richness, indicator taxa or guilds (tolerant and intolerant groups), health of individual organisms, and assessment of processes (Karr and Chu 1997). A multimetric index including a variety of such metrics integrates information from ecosystem, community, population and individual organism levels (Gerritsen 1995, Karr 1991, Barbour *et al.* 1995). Community structure is widely used to assess biological conditions in streams because it is the result of both long-term environmental factors and short-duration conditions (Lenat 1993, 1988; Plafkin *et al.* 1989; Karr *et al.* 1986; Hilsenhoff 1977). Changes in physical, chemical, and biological conditions may result in changes of species composition, species richness, trophic structure, numbers of top carnivores and omnivores, and shifts from specialized to generalized feeding behaviors. Multimetric biological indices have proven effective in assessment of ecological conditions in a variety of management settings and in diverse geographic regions (Karr and Chu 1997).

Although predominantly used in assessment of lotic systems, multimetric indices may be useful in assessment of wetland systems (Burton *et al.* 1999). The objective of this study was to compare the macroinvertebrate fauna of a moderately-impacted northern Lake Huron coastal wetland to the fauna of a nearby, relatively pristine reference wetland. In addition to contributing to the limited knowledge of wetland macroinvertebrates (Gathman *et al.* 1999), the goal was to select and test metrics for development of a multimetric index of ecological integrity (IEI) for Great Lakes coastal wetlands.

MATERIALS AND METHODS

Study Sites

This study was conducted in two coastal wetlands, less than two km apart, along the north shore of Lake Huron (Fig. 1). The two wetland catchments were similar in size, aspect, and other physical characteristics. In the absence of water quality differences, the working hypothesis was that they would have supported similar macroinvertebrate communities (Barbour *et al.* 1992) and that any differences in macroinvertebrates between wetlands that was detected would reflect responses to differences in pollutant loads and disturbance regimes between sites. Both sites were lacustrine wetlands (approximately 900 m^2) and were part of the Les Cheneaux Islands wetland complex.

A small coastal wetland immediately adjacent to Cedarville, Michigan (population of 2,013) at the northwestern end of Cedarville Bay (45° 59′ N, 85° 21′ W) was selected as the impacted site. The wetland received treated discharge from the Cedarville wastewater lagoon treatment system. The lagoons were discharged once in spring and again in fall to Pearson Creek about 2 km upstream from the wetland. Pearson Creek entered the Cedarville wetland through a culvert under a road that isolated the emergent zone from the wet meadow. The wet meadow zone at the impacted site had been partially filled. The northwestern end of Cedarville Bay received storm-water runoff from Cedarville that was not intercepted by the sewer system. A marina with 57 boat slips and a heavily used, public boat ramp were located within 100 m of the sampled area.

A wetland of similar size at the northwestern end of Mackinac Bay (46° 00′ N, 85° 25′ W) was selected as the reference (Fig. 1). This wetland received no wastewater or urban storm-water discharge. Several houses along Mackinac Bay were greater than one km from the sampled area.

The two wetlands had similar geomorphological characteristics including stream discharge, size and shape of watersheds, and underlying geology. Land cover in both watersheds was a mixture of northern hardwood forest and dense stands of northern white

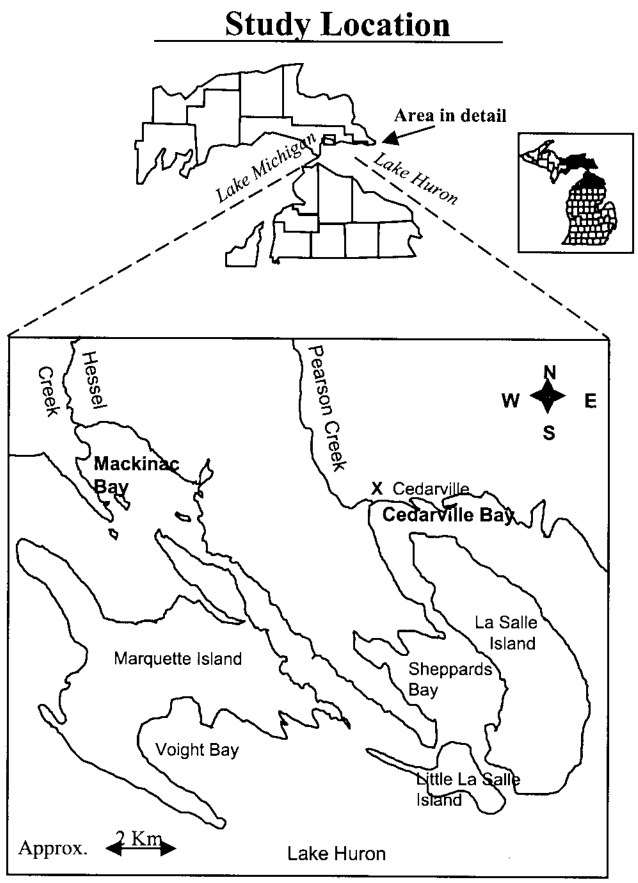

FIG. 1. Location of study sites in Northern Lake Huron Bioreserve, Cedarville and Mackinac wetlands, Lake Huron, Michigan.

cedar, balsam fir, and spruce (Albert *et al.* 1986). The catchments of both wetlands were drained by a single small, similar sized, first-order streams.

The study sites in both bays were located at similar depths in emergent plant zones dominated by *Scirpus acutus* Muhl (hardstem bulrush). *S. acutus* emerged above water in late May, reached maturity in late July, and started to senesce in August. The *S. acutus* zone at the impacted wetland extended 200 to 300 m from the road into Cedarville Bay. The *S. acutus* zone at the reference wetland extended an even greater distance from the wet meadow into Mackinac Bay. The density of *S. acutus* did not differ between the two wetlands, with an average of 292 stems per m^2. Submersed plant density at the impacted site was greater than at the reference site, and more exotic species such as *Myriophyllum spicatum* (Eurasian milfoil) were present. Macroinvertebrate abundance within the *S. acutus* zone increased from June to a peak in September at both sites. Both sites were protected from direct wave action from Lake Huron by a network of islands (the Les Cheneaux Islands).

Study Design

Biological and environmental data were collected from nine stations along a 400 m fixed transect in each wetland. The transects ran parallel to shoreline, 55 m into the emergent plant community. The first sample station was established 10 m west of the point where each stream discharged into the marsh. Sampling stations were established every 50 m along the transect with the last station located 410 m west of the discharge point. This design minimized habitat variability between the two wetlands with sampling occurring at similar water depths (1 m ± 25 cm) in *S. acutus* dominated plant communities with similar emergent stem densities, at the same distance from shore. This design established a gradient away from the source of discharge. Currents within the bay are primarily determined by the strength and direction of the prevailing winds and due to the close proximity of these two wetlands both wetlands experience similar wind patterns. Circulation driven by stream flow is comparable in both wetlands since the streams from both wetlands enter at approximately the same orientation in relation to wind direction. Macroinvertebrate and water quality samples were collected on 24–25 June, 24–25 July, 18–19 August, and 28–29 September 1996.

Chemical Analysis

Water samples were collected in pre-rinsed polyethylene bottles at mid-depth (about 0.5 m) from each of the nine points along each sampling transect in each wetland and stored on ice during transport to the laboratory where they were frozen (except for a sub-sample for Cl) and stored for later analyses. Chemical parameters such as ammonium (NH_4-N), nitrite (NO_2-N), nitrate (NO_3-N), soluble reactive phosphate (SRP), alkalinity, and chloride (CL) were analyzed using a Lachat automated ion analyzer following standard procedures (APHA 1985). Known standards and duplicates were used for quality assurance; agreement was generally within 10%, and a limit of detection of 0.01mg/L was used. Dissolved oxygen (YSI model 51B oxygen meter), pH (Altec monitor II meter), and conductivity (YSI model 31 conductivity bridge) were determined *in situ*. Water temperatures were recorded to the nearest 0.1°C from mid-depth at each sampling point along each transect.

Macroinvertebrate Collection

Macroinvertebrate samples were taken at a random distance (0 to 5 m) and direction (0 to 360°) from each of the nine sampling points along each transect. On each sampling date, semi-quantitative macroinvertebrate samples consisted of two sets of standardized triplicate sweeps with 0.3-m wide D-frame dip nets. The first set of sweeps were made with a 1-mm mesh net which was used to prevent net clogging so more active animals could be captured. The second set of sweeps was made with a 250 µm mesh net to capture smaller macroinvertebrates. Each set of sweeps consisted of three sweeps at the surface, three at mid-depth and three along the sediment surface with all sweeps above the same substrate area. Each set of sweeps covered 0.45 m^2 of substrate (net width of 0.3 m × 0.5 m length of pass × 3 passes); therefore, the total composite sample taken from an area was approximately 0.90 m^2. All samples were washed through a 250 µm sieve in the field. Macroinvertebrates and debris left in the sieve after washing were preserved in 70% ethanol containing 100 mg/L of Rose Bengal to reduce sorting time.

Benthic invertebrates were sampled quantitatively using a corer made of a 1.22 m long, 5 cm (internal diameter) acrylic tube. The corer was shoved 15 cm deep into the sediment, capped with a rubber stopper, and pulled from the sediment. Each core included 15 cm of sediment plus the water column above it. Two cores comprised each sediment sample. No attempt was made to separate macroinvertebrates in water above the sediment surface in corers from those in sediments. All sediment samples were preserved in 70% ethanol containing 100 mg/L of Rose Bengal. Each sediment sample was divided into six sub-samples using a sub-sampler similar to the one described by Waters (1969). Sub-samples were picked with the aid of a dissecting microscope until at least 50 organisms were found.

Organisms were identified to the lowest operational taxonomic unit (usually species or genus, but tribe, family or order for some groups) where possible using Merritt and Cummins (1996), Wiggins (1995), Simpson and Bode (1980), Snider (1967), Weiderholm (1983), or specialist assistance. Taxa difficult to identify (Chironomidae, Oligochaeta) or not properly preserved (Nematoda) were classified at higher taxonomic levels. Chironomidae larvae were identified to sub-family or tribe. A subset of the June Chironomidae samples were identified to genus and, when possible, species, but were not used in calculation of metrics (Kashian 1998). Macroinvertebrates were further classified according to trophic status (omnivore, detritivore, herbivore, carnivore, scavenger), and functional-feeding groups (scraper, predator, collector, filterers, collector-gatherers, piercers, shredders) (Merritt and Cummins 1996).

Metric Selection and Testing

38 structural or functional attributes of macroinvertebrate assemblages were selected for testing as potential metrics for a wetland index of ecological integrity. The selected attributes had been cited as having potential use as metrics in bioassessment (Table 1) (Barbour *et al.* 1996, Resh *et al.* 1995, Kerans and Karr 1994, Kerans *et al.* 1992).

Metrics, based on community structure and composition, consisted of species richness, enumeration of individual taxa, and calculation of relative contributions of each taxon to the total fauna (taken from Kerans *et al.* 1992; Table 1). Richness was calculated for macroinvertebrate taxa that were known to be sensitive to anthropogenic disturbances in streams. One widely used stream metric, the presence of Ephemeroptera, Trichoptera, and Plecoptera (EPT) (Barbour *et al.* 1996, Plafkin *et al.* 1989), was modified to exclude Plecoptera, since they were rare in northern Lake Huron marshes.

Metrics reflecting trophic and functional composition of the assemblage were selected based on the premise that trophic metrics are surrogates of processes such as trophic interaction, production, and food source availability (Barbour *et al.* 1996, Cummins *et al.* 1989, Plafkin *et al.* 1989, Karr *et al.* 1986). The majority of trophic metrics were evaluated as relative abundance. In addition, four functional feeding group metrics suggested by Merritt *et al.* (1996) were used to evaluate several ecosystem attributes (Table 2).

Community diversity and similarity indices are thought to reflect the diversity of aquatic assemblages (Resh *et al.* 1995). Three diversity indices were applied, one evenness index, the Coefficient of Community Loss (Courtemanch and Davies 1987), and the Jaccard Coefficient (Jaccard 1912). The Shannon Index (Shannon and Weaver 1963) incorporates both richness and evenness in its formula. Simpson's Index (Simpson 1949) considers

TABLE 1. Potential metrics and screening criteria for use in an Index of Ecological Integrity. Sensitivity values were calculated from September and August 1996 macroinvertebrate data from Cedarville and Mackinac wetlands. Coefficients of variations are based on values from the reference wetland (Mackinac); low = (CV < .50), high = (CV > 50).

Potential Metrics	Sensitivity value		Coefficient of Variation		Agreement with expected	
	Aug.	Sept.	Aug.	Sept.	Aug.	Sept.
Community structure and composition						
Richness measures						
No. of Crustacea + Mollusca taxa	3*	3*	High	High	O	O
No. of Diptera taxa	0	0	Low	Low	N	N
No. of Ephemeroptera taxa	3*	3*	High	Low	A	A
No. of Ephemeroptera + Trichoptera taxa	3*	3*	High	Low	A	A
No. of families	3	0	Low	High	O	N
No. of Trichoptera taxa	0	2	Low	Low	N	A
Total abundance	3*	0	High	High	O	N
Total taxa richness	3	0	Low	High	O	N
Enumerations- Proportion of individuals as**:**						
Amphipoda	3*	3*	Low	Low	O	A
Chironomidae	0	0	Low	Low	N	N
Chironomini	0	1	Low	Low	N	A
Crustacea + Mollusca	3*	3*	Low	Low	O	O
Diptera	0	0	Low	Low	N	O
Dominant taxon	0	0	Low	Low	N	O
Ephemeroptera	3*	3*	Low	Low	A	A
Gastropoda	3*	0	Low	Low	O	N
Isopoda	3*	3*	Low	Low	A	A
Odonata	3*	0	Low	Low	O	N
Oligochaeta	0	0	Low	Low	N	N
Orthocladiinae	3*	3	Low	Low	A	O
Stylaria	2*	0	Low	Low	O	N
Sphaeriidae	3*	0	Low	Low	N	N
Tanytarsini	0	1	Low	Low	N	A
Trichoptera	3*	3*	Low	Low	A	A
Tubificidae	0	2	Low	Low	N	O
Trophic and Functional composition						
Collector-gatherers	3*	2*	Low	Low	O	O
Filterers	1	1	Low	Low	A	A
Habitat stability	3*	3*	Low	Low	O	O
No. of scrapers/collector-filterers	0	3*	High	High	N	A
No. of scraper + piercer taxa	3*	2	Low	High	O	O
Predators	3*	3*	Low	Low	A	A
Production/Respiration	3*	3*	Low	Low	A	A
Scrapers	3*	3*	Low	Low	O	O
Shredders	0	0	Low	Low	N	N
SPOM/BPOM	1	0	Low	Low	N	N
Top down	3*	3*	Low	Low	O	O
Community diversity and similarity indices						
Evenness	3*	1	Low	Low	O	A
Margalef diversity	1	3*	Low	Low	A	O
Shannon diversity	3*	3*	Low	Low	O	A
Simpson diversity	3*	3	High	High	A	A

*Significant value based on Mann-Whitney U test ($P < 0.05$)
+ O = opposite effect, N = no effect and A = agrees

TABLE 2. Relationships between macroinvertebrate functional groups and ecosystem attributes for which they can serve as analogs (modified after Merritt et al. 1996).

ECOSYSTEM ATTRIBUTES	METHODS	FUNCTIONAL GROUP RATIOS	EXPECTED RATIOS[1]	
			Reference	Impacted
Herbivore *As a proportion of* detritivore as a surrogate for P/R	P/R measurements per unit area on a daily basis	Shredders (Live Vasc. Plants) + Scrapers *As a proportion of* Shredders (CPOM Detritivores) + Total Collectors	Heterotrophic < .75	Autotrophic ≥ .75
Suspended Particulate Organic Matter *As a Proportion of* Deposited (Benthic) Particulate Organic Matter SPOM/BPOM	SPOM Measured per Unit Volume and BPOM Measured per Unit Area	Filtering Collectors *As a Proportion of* Gathering Collectors	Not Enriched in Suspended Particulate Organic Matter < .50	Enriched in Suspended Particulate Organic Matter ≥ .50
Habitat (Substrate) Stability HABITAT STABILITY	Measures Available Surfaces for Stable Attachment (Sediment Coarser than Moved by Maximum Transport Velocity, Large Woody Debris, Rooted Vascular Hydrophytes)	Scrapers + Filtering Collectors *As a proportion of* Total Shredders + Gathering Collectors	Stable Substrates > .60	Unstable Substrates ≤ .60
Top Down Control TOP DOWN	High ratio of slow turnover predators indicates high proportion of fast turnover	Predators *As a proportion of* Total of All Other Functional Feeding Groups	Normal Top Down Predator Control < .15	Sensitive Species Affected ≥ .15

[1] The proposed ratios are based on values calculated from the literature for lotic habitats (Merritt *et al.* 1996).

both abundance patterns and species richness (Cao *et al.* 1996). The Margalef formula (Margalef 1957) differs from Simpson's and the Shannon index in that it does not contain an evenness component. Jaccard's Coefficient of Community similarity measures the degree of similarity in taxonomic composition between two sites in terms of taxon presence or absence and discriminates between highly similar sites. Coefficient values increase as degree of similarity with the reference station increases (Jaccard 1912). The Coefficient of Community Loss Index measures loss of taxa between a reference site and the site of interest. It is an index of compositional dissimilarity and communities are expected to become more dissimilar as stress increases (Courtemanch and Davies 1987).

Metric Testing

The goal of this study was to develop a sampling protocol that would maximize sampling efficiency and minimize sample processing. The August and September samples contained the greatest abundance and diversity of macroinvertebrates and also contained a higher proportion of larger, late instars that were easier to identify and process. The June and July samples were also analyzed, but no additional metric information was gained from them. Thus, results will emphasize samples from August and September.

The low abundance and diversity of macroinvertebrates in sediment samples made it impractical to use them to compare potential metrics. Therefore, macroinvertebrates, collected with dip nets from August and September, were used to test potential metrics as water quality indicators. August samples were used to test whether long-term effects would be detectable, since no wastewater discharge had occurred since 5 June. September samples were used to reflect both short- and long-term impacts of wastewater discharge, since wastewater was being discharged while samples were taken.

Low within-wetland variability was an important criterion for determining the potential usefulness of a metric in detecting anthropogenic impacts. Coefficients of variation were used as a measure of within-site variability (Karr and Chu 1997). A potential metric with a coefficient of variation greater than 50% in either August or September for the reference wetland was deemed unlikely to be useful in detecting differences between impaired and unimpaired (Barbour *et al.* 1996). Potential metrics with coefficients of variation less than 50% were retained for additional testing. Sewage effluent may be classified as a point-source disturbance that could potentially cause high within-site variability at the impacted wetland, therefore, only coefficient of variations for the reference wetland were used to screen potential metrics.

After elimination of potential metrics with high within-wetland variances, the remaining metrics were evaluated for their ability to detect the hypothesized effect of impact. Metrics were calculated for each wetland by pooling macroinvertebrates from all nine stations. Pooled data were used because no consistent patterns in abundance or diversity of macroinvertebrates were detected statistically in response to distance from discharge. Even though chemical differences between the two sites were only detectable for the first 210 to 260 m along the sampling gradient at the time of sampling, the entire emergent zone was subjected to stormwater runoff and marina traffic as well as redistribution of contaminated sediments by winds, storm surges, and boat traffic near and through the wetland.

Results from the two wetlands were compared graphically by examination of "box-and-whisker" plots of median values ± quartile differences in variance between sites (Barbour *et al.* 1996). The positioning of the box-and-whisker plots was used to determine if each metric agreed with the hypothesized predictions of impact, measured no effect of impact, or measured the opposite effect. Metrics that detected the hypothesized effect of impact in August and September were selected as metrics likely to detect long-term impacts. Metrics that detected the hypothesized effect of impact during wastewater discharge in September but measured no effect in August were selected as potential metrics of short-term impacts. Potential metrics which responded in an opposite direction than was hypothesized were eliminated from further consideration. However, hypothesized direction of response was based on stream literature, and metrics that respond in the opposite direction in wetlands may prove to be useful indicators of impact after data are collected from more sites.

Potential metrics were assigned a sensitivity value of three if no overlap in interquatile range existed. A sensitivity value of two was assigned if overlap occurred but did not extend to the medians. A sensitivity value of one was assigned if a moderate overlap of interquartile ranges occurred but at least one median was outside the range; and a sensitivity of zero was assigned if interquartile overlap was considerable, with no discrimination between reference and impaired sites (Barbour *et al.* 1996).

The non-parametric Mann-Whitney U test was used to detect significant differences between sites because metrics were often expressed as proportions and did not tend to be normally distributed. Statistical comparisons were not used as a means of metric elimination, but were instead used to describe differences between fauna of the two wetlands and as a means to support previously determined sensitivity values. Final selection of metrics was based on graphical analysis.

RESULTS

Comparisons of Water Quality Data

The impacted wetland was characterized by higher Cl, inorganic nitrogen and soluble reactive phosphorus (SRP) (Fig. 2). Higher conductivity, turbidity, and alkalinity levels also occurred in the impacted wetland coinciding with lower dissolved oxygen levels during June and September but not during July and August. These differences in water quality during June and September were greatest near the mouth of Pearson Creek (source of wastewater discharge), and decreased to levels similar to those in the reference wetland between 210 and 260

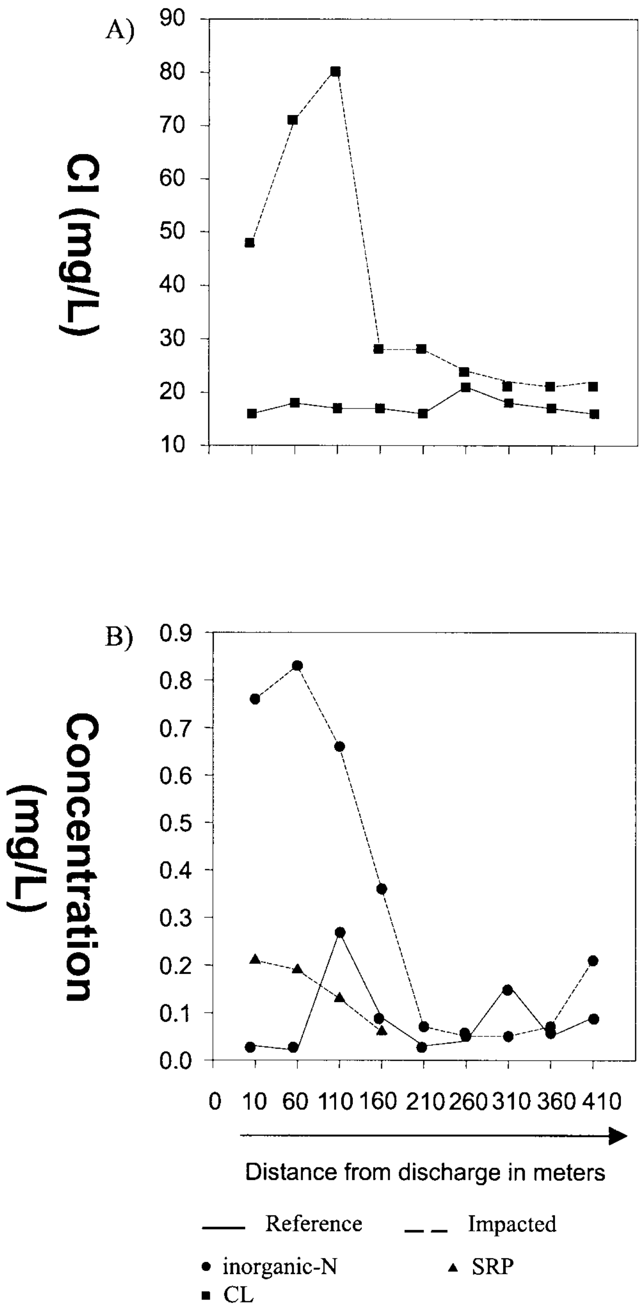

FIG. 2. Changes in Chloride (Panel A) and Inorganic Nitrogen and soluble reactive P (SRP) (Panel B) concentrations along invertebrate sampling transects that extend parallel to shore from Pearson Creek in Cedarville Bay (impacted wetland), and Mackinac Bay (reference wetland) for 410 meters, 12 Sept. 1996. * Limit of detection for SRP = 0.01 mg SRP/L.

m from the creek mouth. No obvious spatial or temporal patterns occurred with pH, turbidity and temperature in either of the wetlands (Table 3). June samples taken approximately two weeks after lagoon discharge indicated that water quality in the wetland had returned to near-background levels.

Cl as high as 81 mg/L near the mouth of the creek occurred during or immediately following discharge from the wastewater treatment lagoon at the impacted wetland in June and September. Cl decreased to background levels (12 to 21 mg/L) at 260 m from the mouth of the creek, indicating rapid dilution and mixing. Cl concentration remained relatively constant spatially and temporally in the reference wetland during the sampling season (Fig. 2).

Turbidity was low at the impacted and reference wetlands, never exceeding five nephelometric turbidity unit (NTUs). The highest turbidity at the impacted wetland occurred during June and September, perhaps indicating higher turbidity associated with wastewater discharge. Trends were inconclusive on other sampling dates.

Dissolved oxygen ranged from 61% saturation to supersaturated values of 114% from June through August at the impacted and reference wetlands. No consistent differences were observed between sites. In September, dissolved oxygen was significantly lower at the impacted wetland during wastewater discharge. Levels below 5 mg/L occurred at sampling stations within 210 m of the creek mouth; DO less than 5 mg/L are known to stress many fish and invertebrates (Wetzel 1983).

During wastewater discharge in September, soluble reactive phosphorus (SRP) was 0.21 mg/L at the station 10 m from the source of discharge on 29 September 1996. SRP decreased with distance from the stream mouth until it decreased below 0.01, the detection limit, at 210 m from discharge (Fig, 3). SRP was always below limits of detection (0.01 mg/L) in the reference wetland.

NH_4-N levels generally ranged from 0.03 to 0.07 mg/L at the reference and impacted wetlands from June through August. NH_4-N was elevated to 0.48 mg/L in September at the impacted wetland near the source of discharge but decreased to background levels of 0.06 mg/L at 210 m from discharge. NH_4-N concentrations were relatively constant in the reference wetland.

NO_3-N was detected at the impacted wetland during June and September but was below the limits of detection (0.01 mg/L) in July and August. During September, NO_3-N was 0.32 mg/L near the source of discharge and decreased with distance from dis-

TABLE 3. Water quality data (mean + S.E.) for Cedarville and Mackinac wetland, Lake Huron 1996. Means calculated from nine sampling stations. * No data.

	Alkalinity (mg $CaCo_3$/L)	Conductivity (uS/cm)	% Dissolved Oxygen	pH	Turbidity (NTUs)	H_2O Temp. (* Celsius)
Reference Wetland (Mackinac)						
June	152 ± 9.7	342 ± 16.4	94 ± 2.9	*	2 ± .03	22 ± 0.3
July	145 ± 6.4	313 ± 10.9	92 ± 3.1	7 ± 0.1	2 ± 0.3	20 ± 0.3
August	176 ± 12.5	387 ± 12.7	100 ± 2.7	8 ± 0.1	1 ± 0.1	22 ± 0.2
September	139 ± 5.6	355 ± 59.0	61 ± 2.1	9 ± 0.1	2.2 ± 0.6	12 ± 0.2
Impacted Wetland (Cedarville)						
June	87 ± 5.6	207 ± 11.9	106 ± 1.7	*	4.1 ± 0.2	21 ± 0.4
July	86 ± 4.3	248 ± 20.0	86 ± 2.11	8 ± 0.02	1 ± 0.2	21 ± 1.2
August	83 ± 7.4	344 ± 53.8	85 ± 4.7	7.6 ± 0.3	1 ± 0.1	24 ± 0.9
September	114 ± 7.8	316 ± 30.0	60 ± 5.2	7 ± 0.4	2 ± 0.4	13 ± 0.3

charge until no longer detected at 210 m. NO_3-N was usually below the limit of detection in the reference wetland.

NO_2-N was only detected in the impacted wetland during September at the three stations nearest the source of discharge. Levels never exceeded 0.03 mg/L. NO_2-N was never detected at the reference wetland.

Comparisons of Macroinvertebrate Community Structure

Macroinvertebrates of plant and sediment associated communities tended to reach peak abundance in September with numbers increasing from June through September. Aquatic insect richness was higher and insects were significantly ($p < 0.05$) more abundant in the plant and sediment associated macroinvertebarte communities in the reference wetland than they were in the impacted wetland (Fig. 3). The decreased abundance of insects in the impacted wetland resulted in dominance by non-insect invertebrates such as Amphipoda, Oligochaeta, Isopoda, Gastropoda, and Nematoda (Fig. 3).

There were 73 aquatic insect taxa collected from the plant associated community from the reference wetland compared to 64 taxa from the impacted wetland (Table 4). Aquatic insect taxa richness from sediment samples followed a similar pattern with 34 taxa from the reference and 22 taxa from the impacted wetland (Table 4, Kashian 1998).

The Chironomidae, Ephemeroptera, and Trichoptera were dominant insect groups in plant and sediment associated communities at the reference wetland. Ephemeroptera and Trichoptera were less important in the impacted wetland with Chironomidae becoming more dominant. Chironomidae represented 70 to 90% of aquatic insects from the impacted wetland compared to 50 to 60% of insects from the reference wetland.

Even though Chironomidae relative abundance was greater in the impacted wetland, actual abundance was comparable for sediment and plant associated communities with no consistent, significant differences ($p < 0.05$) occurring between the two wetlands. Thus, differences in percent composition of insect communities for the two sites (Fig. 3) resulted from decreases in abundance of Ephemeroptera, Trichoptera, and other insects in the impacted wetland compared to the reference wetland rather than from increases in Chironomidae abundance.

Even though there were no consistent, significant ($p < 0.05$) difference in total abundance of Chironomidae larvae between the two wetlands, the taxonomic composition of the Chironomidae did differ. Chironomini and Orthocladiinae (including *Corynoneura* sp.) were more dominant in the plant associated samples from the impacted wetland while Tanypodinae and Tanytarsini were more dominant in the reference wetlands (Kashian 1998). Chironomini were the most common midge larvae collected from both wetlands making up 50 and 43% of total Chironomidae collected from the plant associated samples from the impacted and reference wetlands respectively. These trends also occurred in sediment samples with Chironomini being even more dominant in sediments than they were in the plant associated samples (Kashian 1998).

Ephemeroptera abundance was significantly

III.

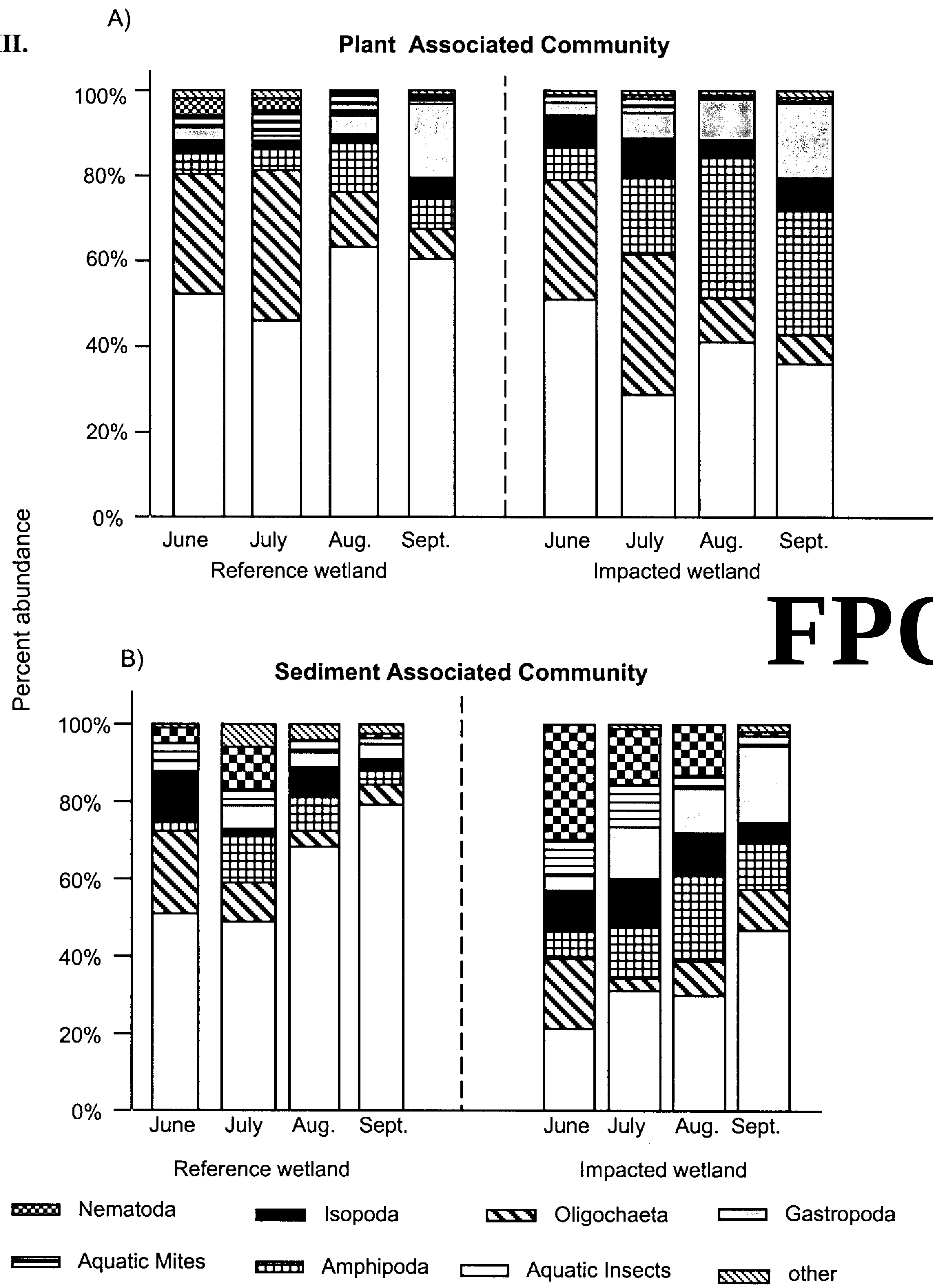

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FIG. 3. Community composition of the macroinvertebrate community at Cedarville (impacted) and Mackinac (reference) wetlands, Lake Huron, 1996. A) Plant associated community (dip-net samples) and B) Sediment associated community (core samples).

TABLE 4. Mean abundance of macroinvertebrates in the plant associated community (standardized dip-net; 0 ± S.E .per sample) and sediment associated community (core sample; 0 ± S.E. per m2) from Mackinac and Cedarville wetlands in 1996. * Not present; + present but mean < 1.

INSECT	Mackinac		Cedarville	
TAXON	Dip-Net	Core Samples	Dip-Net	Core Samples
COLEOPTERA				
Chrysomelidae				
Donacia spp.	+	21 ± 21	+	35 ± 21
Nehaemonia spp.	+	*	+	*
Curculionidae	+	*	*	*
Dytiscidae				
Agbus spp.	+	*	*	*
Elmidae				
Duiraphia spp.	+	*	*	*
Gyrinidae				
Dineutus spp	+	*	+	*
Gyrinus spp.	+	*	+	*
Haliplidae				
Halplus spp.	+	*	+	*
Peltdytes spp.	*	*	+	*
Ptilidae	+	*	*	*
COLLEMBOLA				
Isotomidae				
Isotomurus tricolor	*	*	+	*
Sminthuridae				
Entomobrya nivalis	*	*	*	+
Pseudobourletiella spinata	+	*	+	*
DIPTERA				
Ceratopogonidae				
Bezzia/Palpomyia spp	+	148 ± 72	1 ± 1	21 ± 21
Culicoides spp.	+	*	*	21 ± 21
Probezzia spp.	+	*	*	*
Serromyia spp.	*	7 ± 7	*	*
Sphaeromias spp.	*	21 ± 21	*	*
Chironomidae*				
Chironominae				
Chironomini	75 ± 32	4,135 ± 1,783	118 ± 31	5,605 ± 2,445
Tanytarsini	39 ± 14	693 ± 266	27 ± 11	657 ± 577
Orthocladiinae				
Corynoneura spp.	4 ± 1	6 ± 64	18 ± 6	99 ± 24
Others	26 ± 14	495 ± 213	55 ± 12	417 ± 155
Tanypodinae	39 ± 25	2325 ± 1389	22 ± 4	1,329 ± 674
Empididae	+	134 ± 53	+	*
Ephydridae				
Hydrellia spp.	*	*	+	*
Sciomyzidae	*	*	+	*
Stratiomyidae				
Odontomyia spp.	+	*	+	*
Tabanidae				
Chrysops spp.	*	42 ± 42	+	14 ± 14
Haematopota spp.+		*	*	*
Hybomitra spp.	+	*	+	*
EPHEMEROPTERA				
Baetidae				
Callibaetis ferrugineus	4 ± 2	127 ± 55	2 ± 1	*
Procleon viridocularis	+	*	*	*

(Continued)

TABLE 4. Continued.

INSECT TAXON	Mackinac Dip-Net	Mackinac Core Samples	Cedarville Dip-Net	Cedarville Core Samples
Caenidae				
Caenis amica	25 ± 22	1,230 ± 1,061	+	42 ± 42
Caenis latipennis	9 ± 6	770 ± 659	+	*
Caenis youngi	17 ± 5	1,887 ± 398	*	*
Ephemerellidae				
Eurylophella funeralis	1 ± 1	7 ± 7	+	*
Ephemeridae				
Hexagenia limbata+	212 ± 185		*	*
HEMIPTERA				
Belostomatidae				
Lethocerus spp.	+	21 ± 21	*	*
Corixidae				
Hesperocorixa kennicotti	*	*	+	*
Sigara transfiturata	+	*	*	*
Sigara variabillis	1 ± 1	*	*	*
Palmacorixa buendi	+	21 ± 21	+	*
Trichocorixa sexcincta	*	*	+	*
Others	+	7 ± 7	1	21 ± 21
Hebridridae				
Merragata spp.	+	*	*	*
Nepidae				
Ranatra spp.	+	*	+	*
Mesoveliidae				
Mesovelia mulsanti	+	*	+	21 ± 21
Gerridae				
Gerris comatus	+	*	+	*
Trepobates spp.	*	*	+	*
LEPIDOPTERA				
Noctuidae				
Bellura spp.	+	*	+	*
Pyralidae				
Acentria spp.	+	*	2	21 ± 21
Munroessa/Synclita/Neocataclysta spp.	*	*	1 ± 1	*
Parapoynx spp.	2 ± 1	7 ± 7	+	21 ± 21
NEUROPTERA				
Sisyra spp.	*	*	+	*
ODONATA				
Ashnidae				
Anax junius	+	*	*	*
Aeshna eremita	*	*	+	*
Aeshna spp.	*	*	+	*
Coenagrionidae				
Enallagama boreale	+	*	*	*
Enallagma ebrium/hageni	4 ± 4	177 ± 92	6 ± 5	205 ± 106
Enallagma geminatem	+	*	+	*
Enallagma vernale	+	*	*	*
Ischnura verticalis	30 ± 16	21 ± 21	19 ± 11	14 ± 14
Others	2 ± 1	*	2 ± 2	*
Corduliidae				
Epitheca spp. + *	+	*		
Cordulia shurleffi	1 ± 1	*	+	*
Gomphidae				
Arigomphus cornutus+		*	*	*

(Continued)

TABLE 4. Continued.

INSECT TAXON	Mackinac Dip-Net	Mackinac Core Samples	Cedarville Dip-Net	Cedarville Core Samples
Libellulidae				
Celithemis spp.	+		*	* *
others	+	*	1	*
Lestidae				
Lestes spp.	+	*	+	*
TRICHOPTERA				
Hydropsychidae				
Ceratopsyche spp.	+	*	+	*
Cheumatopsyche campyla	+	*	*	*
Hydroptilidae				
*Agraylea multipuntata**	21	4 ± 3	35 ± 21	
Hydroptila spp.*	*	4 ± 4	*	
Orthotrichia spp.	*	7 ± 7	*	*
Oxyethira spp.	9 ± 5	28 ± 20	6 ± 4	*
Leptoceridae				
Ceraclea spp.	+	*	*	*
Mystacides interjecta	2 ± 2	64 ± 41	+	*
Mystacides sepulchralis	8 ± 3	20 ± 20	2 ± 1	35 ± 35
Nectopsyche spp.	+	*	+	*
Oecetis cinerascens	1 ± 1	*	1 ± 1	14 ± 14
Oecetis inconspicua	+	*	*	*
Oecetis osteni	*	42 ± 42	*	*
Oecetis persimilis	+	*	*	*
Oecetis spp.	9 ± 3	127 ± 127	2 ± 1	21 ± 21
Triaenodes aba	+	*	+	*
Limnephilinae				
Lilmnephilus spp.	+	*	*	*
Nemotaulius hostilus	*	*	+	*
Molannidae				
Molanna tryphena	+	*	*	*
Molanna spp.	+	57 ± 20	*	*
Phryaneidae				
Agrypni improba	+	*	+	*
Fabria spp.	+	*	+	*
Phryganea cinera	14 ± 11	42 ± 42	1 ± 1	21 ± 21
Polycentropodidae				
Cernotina spp.	*	*	+	*
Polycentropus spp.	1 ± 1	134 ± 98	3 ± 1	163 ± 64
Aphids	*	21 ± 21	4 ± 4	*
NON-INSECT TAXON				
ANNELIDA				
OLIGOCHAETA				
Naididae				
Stylaria spp.	68 ± 23	353 ± 145	66 ± 31	551 ± 468
Others	20 ± 9	982 ± 266	59 ± 6	1,548 ± 744
Tubificidae	2 ± 1	454 ± 234	5 ± 3	375 ± 128
POLYCHAETA				
Manayunkia speciosa	*	*	+	*

(Continued)

TABLE 4. Continued.

INSECT TAXON	Mackinac Dip-Net	Mackinac Core Samples	Cedarville Dip-Net	Cedarville Core Samples
HIRUDINAE	*	*	*	*
Erpobdellidae	+	*	*	*
Mooreobdella spp.	*	*	+	*
Glossiphoniidae				
Alboglossiphonia heteroclita	*	*	+	*
Batrachobdella phalera	*	*	+	*
Helobdella stagnalis	*	*	+	*
Theromyzon spp.	+	*	*	*
CRUSTACEA				
AMPHIPODA				
Crangnox spp.	*	*	+	*
Gammarus spp.	3 ± 1	247 ± 45	136 ± 52	1,937 ± 799
Hyallella azteca	37 ± 15	884 ± 213	72 ± 27	1,449 ± 542
DECAPODA				
Orconectes propinquus	*	*	+	*
ISOPODA				
Lirceus lineatus	12 ± 10	156 ± 107	44 ± 12	721 ± 231
Racovitzai racovitzai	9 ± 3	912 ± 396	12 ± 3	1,534 ± 317
MOLLUSCA				
GASTROPODA				
Pulmonata				
Ancylidae				
Ferrissa parallela	+	*	10 ± 8	254 ± 227
Lymnaeidae				
Acella haldemani	+	42 ± 42	+	64 ± 41
Fossaria spp.	+	*	*	*
Physidae				
Aplexa elongata	*	*	+	*
Physa gyrina	5 ± 3	63 ± 41	7 ± 2	134 ± 108
Planorbidae				
Gyraulus deflectus	2 ± 2	*	+	*
Gyralus parvus	41 ± 35	445 ± 112	42 ± 20	1,958 ± 524
Promenetus exacuous	1 ± 1	42 ± 42	6 ± 3	270 ± 127
Prosobranchia				
Bithyniidae				
Bithvnia tentaculata	*	*	+	57 ± 32
Hydrobiidae				
Amnicola limosa	8 ± 6	184 ± 47	23 ± 9	728 ± 214
Valvata bicarinata	*	*	+	*
OTHERS				
NEMATODA	9 ± 2	587 ± 292	3 ± 1	3,011 ± 872
PELECYPODA				
Sphaeriidae	5 ± 2	523 ± 131	2 ± 1	78 ± 60
ARACHNOIDEA (MITES)				
Oribatei	5 ± 2	424 ± 95	3 ± 1	1,096 ± 359
Hydrocarina	13 ± 2	184 ± 29	10 ± 5	346 ± 72
Spider				
Tetragnatha laboriosa	+	*	*	*
TUBELLARIA				
Tricladida	+	21 ± 21	+	*
Dugesia tigrina	*	*	1 ± 1	*

higher ($p < 0.05$) in plant and sediment associated communities in the reference wetland than in the impacted wetland. Plant associated Ephemeroptera never comprised more than 2% of the aquatic insect community in the impacted wetland even though they represented more than 10% of the fauna on every sampling date for the reference wetland. In fact, mayflies increased from 15 per sweep net sample in June to 155/sweep net sample in September in the reference wetland, while the highest number ever collected from the impacted marsh was 4/sweep net sample in July. Likewise, sediment associated Ephemeroptera comprised 20% or more of insects collected from the reference wetland on every sampling date, but were completely absent from impacted wetland sediments in July, August, and September and were only present in low numbers in June. These extreme differences in abundance suggest that mayflies should be excellent indicators of water quality in these wetlands.

Caenidae was the predominant Ephemeroptera family in plant and sediment associated communities of the reference wetland with three species comprising from 80 to 96% of total mayfly numbers collected. *Callibaetis ferrugineus* was the most abundant Ephemeroptera in the impacted wetland, yet only an average of two individuals per sample were collected.

Over all sampling periods, Trichoptera were significantly ($P < 0.05$) less abundant in plant and sediment communities in the impacted wetland than they were in the reference wetland (mean of 21/dip-net vs. 47/dip net for plant associated and 290/m^2 vs. 643/m^2 for sediment samples for impacted vs. reference wetlands). These differences were even greater during June and September, shortly after or during wastewater lagoon discharge. Trichoptera peak abundance in the reference wetland was 2.6 times greater in sediment and 3.3 times greater in the plant associated communities than in the impacted wetland during June and September.

There were 19 species of Trichoptera collected in plant associated samples from the reference wetland while 16 species were collected from the impacted wetland (Table 4). Differences in species richness were greater for sediment samples with 10 taxa collected from the reference wetland while only six were collected from the impacted wetland. Four of the six species collected from the impacted wetland occurred in only one month of sampling. *Polycentropus* sp. and *Agraylea multipuntata* were the only species that consistently occurred in sediment samples from the impacted site, and their abundance was greater at the impacted site than at the reference site. *Polycentropus* spp., for example, made up 56% (162/m^2) of total Trichoptera in the impacted but only 21% (134/m^2) of total Trichoptera in the reference wetland

Phryganea cinera, Oxyethira spp., *Oecetis* spp., and *Mystacides sepulchralis* were the most abundant Trichoptera species in plant and sediment associated communities in the reference wetland. These same species were present in the impacted wetlands but occurred at much lower density than in the reference wetland (Table 4). During wastewater discharge in September, *P. cinera* were significantly ($p < 0.05$) more abundant in the plant associated samples from the reference wetland (14.1/dip net) than in samples from the impacted wetland (1.5/dip net).

Odonata represented approximately 15% of plant associated macroinvertebrates in both wetlands with the majority being Coenagrionidae damselflies. *Ischnura verticalis* was the dominant damselfly in each wetland. There were 13 Odonate taxa collected from the reference wetland with 10 collected from the impacted wetland (Table 4). Odonata abundance was comparable for sediment and plant associated communities in the reference and impacted wetlands with no significant ($p < 0.05$) differences occurring for any sampling date. Odonata were uncommon in sediment samples of both wetlands. The only two Odonata in the sediments were *Enallagma* sp. (*ebrium/hageni*) and *Ischnura verticalis* with *Enallagma* being the most common (Table 4).

The remainder of the aquatic insects in the plant and sediment associated samples occurred infrequently and in low numbers in the two wetlands (Table 4).

Non-insect macroinvertebrates were significant components of the invertebrate fauna in both wetlands (Table 4). Their occurrence was significantly ($p < 0.05$) higher in sediment and plant associated communities of the impacted wetland than in the reference wetland. A total of 8,619 non-insect invertebrates representing 23 taxa were collected from the plant associated community of the reference wetlands, while 10,985 representing 30 taxa were collected from the impacted wetland (Table 4). There were 17 taxa collected from sediments from the reference wetland, while 18 taxa were collected from the impacted site. Oligochaeta, Amphipoda, Isopoda, and Gastropoda were among the most common taxa collected from both sites. Most taxa were collected in both sediment and dip net samples; the exception was leeches (Hirudinae) which were only collected with dip nets (twp species from the reference and four from the impacted site with no species overlap between the two sites).

Gastropoda were more abundant in sediment and plant associated communities at the impacted wetland, with significant differences ($p < 0.05$) occurring in July and August in the plant associated community, and August and September in the sediment associated community. Gastropoda was the most diverse group of non-insects in the sediment and plant associated communities with 10 species collected from the impacted and 8 species collected from the reference wetland (Table 4). *Gyraulus parvus*, *Amnicola limosa*, and *Physa gyrina* were the most common species occurring in the plant and sediment communities in both wetlands.

Although Oligochaeta represented a large portion of the macroinvertebrate community there were no significant differences ($p < 0.05$) in Oligochaeta abundance between the two wetlands on any sampling date (Table 4). The percent abundance of Oligochaeta was remarkably similar between the two wetlands and sampling regimes. Oligochaeta represented 16% of the plant associated community at both the impacted and reference wetlands and 10% of the sediment associated community in both wetlands. Most of the Oligochaeta collected from the reference and impacted wetlands were Naididae. Tubificidae are often classified as very pollution tolerant but no consistent differences occurred between the impacted and reference sites.

Amphipoda were more abundant in the sediment and plant associated communities at the impacted wetland compared with the reference wetland. Significant differences ($p < 0.05$) between the two wetlands occurred in July, August, and September in the plant associated community, and in September in the sediment associated community. The overall dominant Amphipoda in sediment and plant associated communities of the impacted wetland was *Gammarus* spp., while *Hyalella azteca* was dominant in the reference wetland. Sediment associated Amphipoda density at the impacted wetland was greater than at the reference wetland on every sampling date and reached a maximum mean density of approximately 4,500/m^2 in September, the only date where differences were significant at the $p > 0.05$ level. *Gammarus* spp. abundance was 45 fold greater at the impacted wetland in plant associated samples and 8 fold greater at the impacted wetland sediment samples compared to the reference wetland and these differences were significant at the $p > 0.05$ level.

Isopoda were much larger components of the plant-associated community at the impacted wetland than at the reference wetland with over 2,000 individuals collected, compared with only 730 individuals collected in the reference wetland. Even so, differences between the two sites were only significantly different during August because of high variance. Isopoda abundance was also higher (1,534/m^2) in the sediment community of the impacted wetland compared to the reference wetland (912/m^2) but these differences were not significant ($p > 0.05$). Two species of Isopoda were collected in the wetlands, *Lirceus lineatus* and *Racovitzai racovitzai*. *L. lineatus* was the dominant Isopoda in the plant associated community, while *R. racovitzai* was the dominant species in the sediment community of both wetlands.

Metric Results

From an original suite of 38 metrics, six metrics met the strict criteria of low within-site variability in the reference wetland, no overlap of interquartile ranges between the two wetlands for both months, and were assigned a sensitivity value of three (Table 5, Fig. 4). These metrics were able to separate the impacted wetland from the reference wetland and are strong candidate metrics for use as indicators of long-term impacts on water quality in northern Lake Huron coastal wetlands (Fig. 4).

Ephemeroptera and Trichoptera were sensitive indicators of water quality with proportions decreasing in the impacted wetland (Table 1, Fig. 4). Isopoda increased with impacts reflecting their known tolerance to pollution. Three functional feeding group metrics showed promising results. Filterers were sensitive indicators of water quality with proportions decreasing in the impacted wetland (Fig. 4). The proportion of fauna as predators decreased (Fig. 4), perhaps reflecting loss of sensitive top predators. The increase in the herbivore/detritivore ratio at the impacted wetland (Fig. 4) is consistent with an expected increase in primary production due to nutrient enrichment (Table 1).

Another potentially useful metric was the number of Trichoptera taxa (Table 5). This metric was assigned a sensitivity value of zero and showed no effect in August based on examination of the box-and-whisker plots and p-values. However, this metric was assigned a sensitivity value of two for September (Table 1). Although inconsistent be-

TABLE 5. Candidate metrics for use in northern Lake Huron coastal marshes.

Candidate Metrics of Long-Term Impact—Metrics which detected the hypothesized effect of impact in August and September

Proportion of individuals as Ephemeroptera
Proportion of individuals as Trichoptera
Proportion of individuals as Isopoda
Proportion of individuals as Filterers
Proportion of individuals as Predators
Production/Respiration

Candidate Metrics of Short-Term Impact—Metrics which detected the hypothesized effect of impact during wastewater discharge in September, but measured no effect of impact in August

Number of Trichoptera taxa

Metrics for Further Analysis—Metrics consistently demonstrating results opposite than predicted in stream systems

Proportion of individuals as scrapers
Habitat Stability
Top Down

tween months, the number of Trichoptera taxa did determine differences in these coastal wetlands during the period of wastewater discharge. Therefore, it may be a valuable metric for detection of short-term impacts.

Three additional metrics, the proportions of individuals as scrapers, metrics of habitat stability, and top down control demonstrated potential for use as indicators of water quality in northern Lake Huron coastal wetlands (Table 5). However, these metrics consistently showed the opposite effects of impact than were predicted based on stream literature (Table 1). These metrics may prove useful after further investigation. The opposite of what is predicted in the stream literature may consistently hold true in Great Lakes coastal wetlands. However, they were not recommended as candidate metrics in this study because they exhibited an unpredicted response.

In a direct comparison between sites, the Jaccard Coefficient of Community Similarity mean value was 0.48 in August and 0.50 in September. The Jaccard Coefficients indicated a moderate degree of dissimilarity between the reference and impacted wetlands. The mean Coefficient of Community Loss was 0.34 in August and 0.42 in September. The higher value in September indicated a loss of benthic taxa at the impacted wetland.

DISCUSSION

Water chemistry differences between the reference wetland (Mackinac Bay) and the impacted wetland (Cedarville Bay) indicated that impacts on Cedarville Bay wetland were likely moderate. Differences in the macroinvertebrate community existed in both community structure and composition within the plant and sediment associated communities between the marshes.

Chemical and land-use differences were reflected in 10 metrics and may be indicators of the integrity of Great Lake coastal wetlands (Table 5). Six metrics including the proportion of individuals as Ephemeroptera, Trichoptera, Isopoda, filterers, predators, and herbirove/detritivore ratio consistently detected changes in the invertebrate community and are strongly recommended as candidate metrics of long-term impacts for use in development of a multimetric index of ecological integrity. An additional metric, number of Trichoptera taxa, was less consistent in detecting impacts but is recommended as a metric capable of detecting short-term impacts. The proportion of individuals as scrapers, the ecosystem attribute metrics of habitat stability, and top down control, gave consistent results. Although, they were in the opposite direction from those predicted for streams (Kerans and Karr 1992).

The remaining 28 potential metrics derived from indices developed for wadeable streams and rivers (Kerans and Karr 1992, Plafkin *et al.* 1989, Ohio EPA 1988) did not appear to be useful for Great Lakes coastal wetlands (Table 1).

Ephemeroptera, an order of aquatic insects commonly used to monitor water quality based on their sensitivity to disturbance (Plafkin *et al.* 1989), had significantly ($p < 0.05$) lower taxa richness and were significantly ($p < 0.05$) less abundant in the impacted marsh compared with the reference marsh. Ephemeroptera taxa richness at the reference wetland was significantly higher ($p < 0.05$) in September and August than it was at the impacted wetland, with a sensitivity rating of 3 (Table 1). However, it was not selected for a candidate metric because of high within-site variability in August. This metric may still be useful but further testing across a wider array of wetlands with varying degrees of impacts is required.

All of the metrics that included Gastropoda, in-

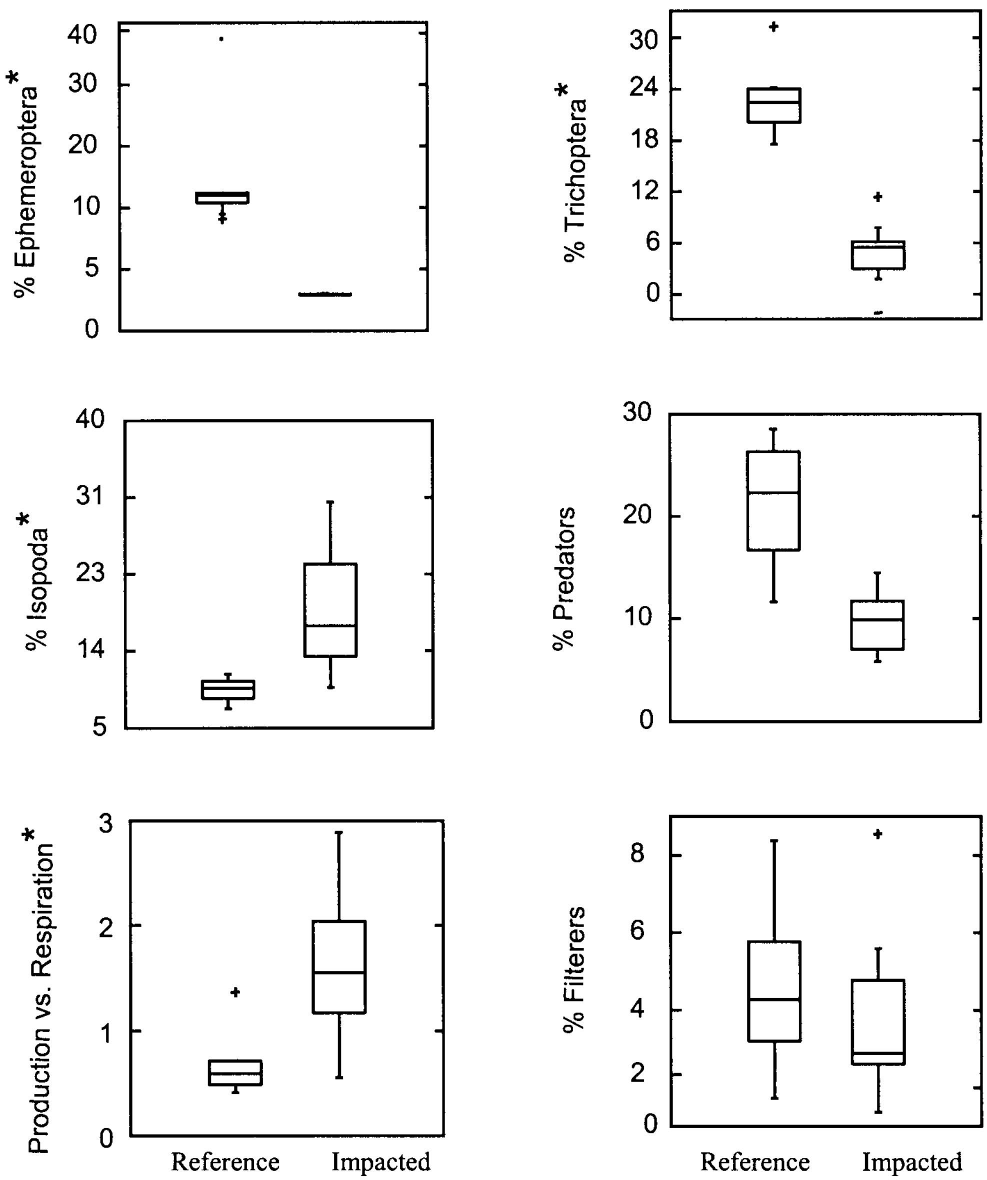

FIG. 4. A comparison of candidate metrics of long-term impact, for use in northern Lake Huron coastal wetlands. Range bars show maximum and minimum of non-outliers; solid lines inside the box are medians; boxes are interquartile ranges (25%ile to 75%ile); dots are outliers. * Indicates significance at p = 0.05 (Mann-Whitney U).

cluding measures of taxa richness, showed results opposite from those predicted from the stream literature or showed no effect of impact. Based on values obtained from the stream literature these results indicated that metrics using Gastropoda may not be useful indicators of water quality in these Great Lake coastal wetlands. However, the macroinvertebrate data collected from these two wetlands showed that indeed significant differences ($p < 0.05$) in the Gastropoda community did exist between the impacted and reference wetland. Gastropoda were more diverse and abundant in the impacted marsh than they were in the reference marsh.

Significant differences between the reference and impacted wetland suggest that Gastropoda may be sensitive enough to detect differences in water quality. A possible explanation for this metrics failure to reflect the predictions derived from the stream literature is that the increase in abundance and diversity of Gastropoda in the impacted wetland may have resulted from increased nutrients stimulating periphyton growth, a major food source for snails resulting in increased presence of pollution-tolerant snails. All the Gastropoda metrics that included calculations involving abundances or richness of the family Gastropoda, including functional feeding metrics involving scrapers, were expected to decline with increasing anthropogenic impacts. For example, Fore *et al.* (1996) suggested that the number of Crustacea plus Mollusca taxa was a useful measure of calcium-dependent taxa, and should decrease with an increase in disturbance due to the presence of sensitive Crustacea and Mollusca taxa. Very few sensitive Crustacea or Mollusca were collected in either the reference or the impacted wetlands, and the abundance of pollution-tolerant snails increased at the impacted wetland. The inability of this metric to detect the hypothesized impact on northern Lake Huron may be due to the lack of sensitive stream Crustaceans and the presence of pollution-tolerant snails. Therefore, metrics involving Gastropoda may be useful indicators of disturbance in coastal wetlands.

The moderate inputs of organic pollution in these wetlands may be sufficient to increase the food source for some of the moderately tolerant macroinvertebrates without causing negative impacts, therefore resulting in greater richness. However, these results could prove to be consistent across an array of coastal wetlands and varying degrees of degradation, and therefore should not be neglected**.**

The metrics based on functional feeding groups may provide information not readily obtained from taxonomic metrics (Plafkin *et al.* 1989, Barbour *et al.* 1996, Kerans *et al.* 1992). Useful measures of trophic and functional composition included measures of relative abundance of predators, filterers, and the ratio of herbivores to detritivore, a surrogate for the primary production to community respiration ratio (P/R). Wallace *et al.* (1977) suggested that filter feeders are sensitive to pollution in low-gradient streams. The metric based on proportion of filter feeders at the impacted wetland as compared to the reference wetland indicated that they also are sensitive in wetlands and may be a useful indicator of water quality in coastal wetlands. The metric based on the herbivore/detritivore ratio is used as a surrogate for gross primary production to community respiration ratio (P/R) and evaluates the balance between autotrophy and heterotrophy (Merritt *et al.* 1996). The herbivore/detritivore ratio indicated that the impacted wetland was an autotrophic system while the reference wetland was a heterotrophic one. The autotrophic nature of the impacted wetland appears to be the result of increased nutrient levels associated with the sewage effluent.

The ecosystem attributes measure of top down control (Merritt *et al.* 1996), predicted that normal top down predator control as the ratio of predators to all other functional feeding groups should be less than 0.15 for Florida streams. This metric consistently showed the reference wetland to be greater than 0.15 and consistently less than 0.15 in the impacted wetland. This metric was not selected as a candidate metric as a result. The ratio threshold calculated to be 0.15 for South Florida wetlands (Merritt *et al*. 1996) may differ regionally for Great Lakes coastal wetlands.

Two direct comparisons between the reference and the impacted sites were calculated using the Jaccard Coefficient and the Coefficient of Community Loss (Jaccard 1912, Ohio EPA 1987, Courtemanch and Davies 1987). These coefficients may be valuable in the assessment of water quality in coastal wetlands, but values obtained from a comparison of two wetlands provided little insight into ecological condition. It is plausible that the values obtained from these metrics are normal for this area, but this cannot be assessed without a larger sample size.

Several metrics commonly used in lotic systems appear to be less hardy or practical for use in coastal wetlands. For example, the metric of suspended particulate organic matter (SPOM) as a proportion of deposited particulate organic matter (BPOM) was not useful in differentiating between wetlands. However, information can still be drawn from this metric because the ratio of SPOM/BPOM proposed by Merritt *et al.* (1996) indicated that both wetlands were enriched in suspended particulate organic matter.

This study supports the suggestion of Karr and Chu (1997) that diversity indices are often inconsistent because they respond erratically to changes in assemblages. Diversity indices may therefore produce a different rank order of the same series of sites, making it impossible to compare the site's biological condition (Karr and Chu 1997). The use of diversity indices in this study demonstrated conflicting results within the same site and month (Table 1). For example, in September, the Shannon diversity index indicated higher diversity at the reference wetland, while the Margalef diversity index indicated higher diversity at the impacted wetland. Diversity indices are influenced by both number of taxa and their relative abundances. Some are more sensitive to rare taxa, and others to abundant taxa. Therefore, different indices measure slightly different aspects of assemblages, and can lead to different results. Eveness indices were also inconsistent from month to month. For example, evenness indicated that the invertebrate community was more evenly distributed at the reference wetland in September during wastewater discharge but that the impacted wetland was more evenly distributed in August. These results are opposite from stream predictions.

Chironomidae are fairly tolerant of pollution, and high abundances of Chironomidae are often used as indicators of degradation (Barbour *et al.* 1995, Plafkin *et al.* 1989). Kerans and Karr (1994) suggested that Chironomidae variability and the need to identify them to lower taxonomic units make them of limited use as indicators of water quality. Chironomidae were identified to species on one sampling date, but this level of taxonomic detail did not provide any better indication of discharge impacts than identification to the level of chironomidae tribe (Kashian 1998). Metrics based on total abundances and abundance of tribes generally showed no effect of degradation in this study, supporting the suggestion of Kerans and Karr (1994).

Development of an IEI for biomonitoring based on invertebrates appears to be feasible for northern Lake Huron coastal wetlands (this study; Burton *et al.* 1999). The ten metrics selected in this study need to be tested and validated across a gradient of human influence for Great Lakes coastal wetlands before they can be widely adopted or discarded. In addition, the proposed metrics may need to be modified for particular ecoregions. The incorporation of additional metrics based on attributes of several other species groups (fish, algae, macrophytes) offers an additional means of strengthening the proposed IEI for these wetlands. Thus, testing and validation of the recommended ten metrics and addition of metrics based on other taxonomic groups are recommended as next steps in development of an IEI for Great Lakes coastal wetlands.

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REFERENCES

Albert, D.A., Denton, S.R., and Barns, B.V. 1986. *Regional Landscape Ecosystems of Michigan*. University of Michigan, Ann Arbor.

American Public Health Association (APHA). 1985. *Standard Methods for the Examination of Water and Wastewater*, 16th Edition. American Public Health Association, Washington, D. C.

Barbour, M.T., Plafkin, J.L., Bradley, B.P., Graves, C.G. and Wisseman, R.W. 1992. Evaluation of EPA's rapid bioassessment benthic metrics: Metric redundancy and variability among reference stream sites. *Environ. Toxicol. Chem.* 11:437–49.

_____, Stribling, J.B., and Karr, J.R. 1995. Multimetric approach for establishing biocriteria and measuring biological condition. In *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making*, eds. W.S. Davis and T.P. Simon, pp. 63–77. Boca Raton, FL: Lewis.

_____, Gerritsen, J., Griffith, G.E., Frydenborg, E., McCarron, R., White, J.S., and Bastian, M.L. 1996. A framework for biological criteria for Florida streams using benthic macroinvertebrates. *J. N. Am. Benthol. Soc.* 15:185–211.

Burton, T.M., Uzarski, D.G., Gathman, J.P., Genet, J.A., Keas, B.E., and Stricker, C.A. 1999. Development of a preliminary invertebrate index of biotic integrity for Lake Huron coastal wetlands. *Wetlands* 19(4): 869–882.

Cao,Y., Bark, A.W., and Williams, P.W. 1996. Measuring the responses of macroinvertebrate communities to water pollution: a comparison of multivariate approaches, biotic and diversity indices. *Hydrobiologia* 34:1–19.

Courtemanch, D.L., and Davies, S.P. 1987. A coefficient of community loss to assess detrimental change in aquatic communities. *Water Research* 21:217–22.

Cummins, K.W., Wilzbach, M.A., Gates, D.M., Perry, J.B., and Taliaferro, W.B. 1989. Shredders and riparian vegetation. *Bioscience* 39:24–30.

Environmental Protection Agency. 1986. *Quality criteria for water 1986*. EPA-440/5-86-001. U.S.

Fore, L.S., Karr, J.R., and Conquest, L.L. 1993. Statistical properties of an index of biological integrity used to evaluate water resources. *Can. J. Fish. Aquat*. Sci. 51:1077–087.

______, Karr, J.R., and Wisseman, RW. 1996. Assessing invertebrate responses to human activities: Evaluating alternative approaches *J. N. Am. Benthol. Soc*. 15:212–231.

Gathman, J.P., Burton, T.M., and Armitage, B.J. 1999. Coastal wetlands of the Upper Great Lakes: Distribution of invertebrate communities in response to environmental variation, In *Invertebrates in Freshwater Wetlands of North America. Ecology and Management*, eds. D.P. Batzer, R.B. Rader, and S.A. Wissinger, pp. 949–994. New York: John Wiley & Sons, Inc.

Gaudet, J.J. 1974. The normal role of vegetation in water. In *Aquatic Vegetation and Its Use and Control*, D.S. Mitchell (ed), pp. 24–37. Paris: Unesco.

Gerritsen, J. 1995. Additive biological indices for resources management. *J. N. Am. Benthol. Soc.* 14(3): 451–57.

Hilsenhoff, W.L. 1977. *Use of arthropods to evaluate water quality of streams*. Technical Bulletin No. 100. Department of Natural Resources, Madison, Wisconsin.

Jaccard, P. 1912. The distribution of flora in the alpine zone. *New Phytologist* 11:37–50.

Jaworski, E., and Raphael, C.N. 1978. *Fish, Wildlife, and Recreational Values of Michigan's Coastal Marshes. Phase I.* Prepared for Great Lakes Shorelands Section, Division of Land Resource Programs, Michigan Department of Natural Resources, Lansing, MI.

Jude, D.J., and Pappas, J. 1992. Fish utilization of Great Lakes Coastal marshes: Current knowledge and research needs. *J. Great Lakes Res.* 18(4):651–72.

Karr, J.R. 1981. Assessment of biotic integrity using fish communities. *Fisheries.* 6(6):21–27.

______. 1991. Biological integrity: a long neglected aspect of water resource management. *Ecological Applications* 1:66–84.

______. 1993. Defining and assessing ecological integrity: beyond water quality *Environ. Toxicol. Chem.* 12:1521–1531.

______, and Chu, E.W. 1997. *Biological monitoring and assessment using multimetric indexes effectively.* EPA 235-R97-001. University of Washington, Seattle, Washington.

______, Fausch, K.D., Angermeier, P.L., Yant, P.R., and Schlosser, I.J. 1986. *Assessing Biological Integrity in Running Waters a Method and its Rationale.* Special Publication 5. Illinois Natural History Survey, Illinois.

Kashian, D.R. 1998. The use of macroinvertebrates as indicators of water quality for two Norhtern Lake Huron coastal wetlands. M.S. thesis, Michigan State Univ., East Lansing, MI.

Kerans, B.L., and Karr, J.R. 1994. A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. *Ecological Applications* 4(4):768–85.

______, Karr, J.R., and Ahlstedt, S.A. 1992. Aquatic invertebrate assemblages: spatial and temporal differences among sampling protocols. *J. N. Am. Benthol. Soc.* 11:377–90.

Krieger, K.A. 1992. The ecology of invertebrates in Great Lakes coastal marshes: Current knowledge and research needs. *J. Great Lakes Res.* 18(4):634–50.

Lenat, D.R. 1988. Water quality assessment of streams using a qualitative collection method for benthic macroinvertebrates. *J. N. Am. Benthol. Soc*. 7: 222–33.

______. 1993. A biotic index for the southeastern United States: derivation and list of tolerance values, with criteria for assigning water-quality ratings *J. N. Am. Benthol. Soc.* 12(3):279–90.

Margalef, D.R. 1957. Information theory in ecology. *Yearbook of the Society for General Systems Research* 3:36–71.

Merritt, R.W., and Cummins, K.W. (eds.). 1996. *An introduction to the aquatic insects of North America*, 3d edition. Kendall/Hunt Publ. Co., Dubuque, IA.

______, Wallace, J.R., Higgins, M.J., Alexander, M.K., Berg, M.B., Morgan, W., Cummins, K.W., and Vandeneeden, B. 1996. Procedures for the functional analysis of invertebrate communities of the Kissimmee river-floodplain ecosystem. *Florida Scientist*. 59: 216–74.

Ohio EPA (Environmental Protection Agency). 1987. *Biological criteria for the protection of aquatic life.* Volume 1–3. Monitoring and Assessment Program, Surface Water Section, Division of Water Quality, Columbus, Ohio.

______. 1988. *Biological Criteria for the Protection of Aquatic life*, volumes 1–3. Ecological Assessment Section, Division of Water Quality Monitoring and Assessment, Ohio Environmental Protection Agency, Columbus, Ohio.

Olive, J.H., Jackson, J.L., Bass, H., Holland, L., and Savisky, T. 1988. Benthic macroinvertebrates as indexes of water quality in the Upper Cuyahoga River. *Ohio J. Sci*. 88:91–98.

Plafkin, J.L., Barbour, M.T., Porter, K.D., Gross, S.K., and Hughes, R.M. 1989. *Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish*. EPA/440/4-89-001. Office of Water, US Environmental protection Agency, Washington, DC.

Prince, H.H., and Flegel, C.S. 1995. Breeding avifauna of Lake Huron. In *The Lake Huron Ecosystem: Ecol-*

ogy, Fisheries, and Management, eds. M. Munawar, T. Edsall, and J. Leach, pp. 247–272. The Netherlands: SPB Academic Publishing, Amsterdam.

Resh, V.H., Norris, R.H., and Barbour, M.T. 1995. Design and implementation of rapid assessment approaches for water resource monitoring using benthic macroinvertebrates. *Aust. J. Ecol*. 20:108–121.

Rosenberg, D.M., and Resh, V.H. (eds.) 1993. *Freshwater biomonitoring and benthic macroinvertebrates*. New York: Routledge, Chapman & Hall.

Shannon, C.E., and Weaver, W. 1963. *The mathematical theory of communication*. Urbana, IL: University of Illinois Press.

Simpson, E.H. 1949. Measurement of Diversity. *Nature* 163:688.

Simpson, K.W., and Bode, R.W. 1980. *Common Larvae of Chironomidae (Diptera) from New York State Streams and Rivers, with particular reference to the fauna of artificial substrates.* Bulletin No. 439, New York State Museum, The State Education Department, Albany, New York.

Snider, R.J. 1967. An annotated list of the Collembola (springtales) of Michigan. *The Michigan Entomologist* 1:179–234.

Southerland, M.T., and Stribling, J.B. 1995. Status of biological criteria development and implementation. In *Biological Assessment and Criteria. Tools for Water Resource Planning and Decision Making*, eds. W.S. Davis and T.P. Simon, pp. 81–96. Boca Raton, Florida: Lewis Publishers.

Wallace, J.B., Webster, J.R., and Woodall, W.R. 1977. The role of filter feeders in flowing waters. *Archiv fur Hydrobiologie* 79:506–32.

Waters, T.F. 1969. Subsampler for dividing large samples of stream invertebrate drift. *Limnol. Oceanogr*. 14(5):813–815.

Weiderholm, T. (ed.) 1983. Chironomidae of the Holarctic Region: Keys and Diagnoses. Part 1. Larvae. *Entomological Scandinavica*, Supplement No. 19.

Wetzel, R.G. 1983. *Limnology*, 2nd edition. New York: Harcourt Brace College Publ.

_____, and Allen, H.L. 1972. Functions and interactions of dissolved organic matter and the littoral zone in lake metabolism and eutrophication. In *Productivity Problems of Fresh Waters*, Z. Kajak and A. Hillbricht-Ilkowska (eds.), pp. 333–347. Warszawa-Krakow, Poland.

Wiggins, G.B. 1995. *Larvae of the North American Caddisfly Genera* (2nd ed.). University of Toronto Press, Toronto.

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