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The Use of Macroinvertebrates as Indicators of Water Quality for Two Northern Lake Huron Coastal Wetlands

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Donna Rebecca Kashian

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Thomas M. Burton

Major professor

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THE USE OF MACROINVERTEBRATES AS INDICATORS OF WATER QUALITY FOR TWO NORTHERN LAKE HURON COASTAL WETLANDS

By

Donna R. Kashian

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ABSTRACT

THE USE OF MACROINVERTEBRATES AS INDICATORS OF WATER QUALITY FOR TWO NORTHERN LAKE HURON COASTAL MARSHES

By

Donna R Kashian

The macroinvertebrate fauna of a moderately impacted Northern Lake Huron coastal, emergent marsh was compared to the fauna of a nearby, relatively pristine, reference marsh. The impacted marsh received domestic wastewater twice per year from a lagoon system and was impacted by marina traffic and stormwater runoff from a nearby urban area. The reference marsh received no such impacts. Community structure of the macroinvertebrates was determined from sediment and dip-net samples collected from June-September in 1996. The invertebrate fauna demonstrated moderate impairment at the impacted marsh with fewer insects present and a greater portion of the fauna existing as Amphipoda and Isopoda. There was a diverse Ephemeroptera community of 7 species in the reference marsh for all sampling dates whereas Ephemeroptera were present only in June with 4 species at the impacted marsh. Trichoptera exhibited lower abundances and species richness at the impacted marsh compared with the reference marsh. Differences in the macroinvertebrate communities were used to test 38 potential metrics as indicators of water quality. Ten metrics appeared to be good water quality indicators including the relative abundance of Ephemeroptera, Isopoda, Trichoptera, predators, filterers and a ratio of grazers to detritivores.

ACKNOWLEDGMENTS

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CHAPTER ONE

A COMPARISON OF TWO NORTHERN LAKE HURON COASTAL MARSHES

Introduction

The Great Lakes coastal marshes are an integral part of the Great Lakes ecosystem. They are important as fish spawning grounds (Jude and Pappas 1992) and waterfowl breeding grounds (Prince and Flegel 1995). Not only do the coastal marshes provide a diverse habitat for animals and plants but they also act as a sieve and a trap for allochthonous and autochthonous materials (Gaudet 1974, Wetzel and Allen 1972). Coastal marshes have the potential of contributing to the total production of the lake ecosystem and regulate, at least in part, the metabolism of many lakes (Jude and Pappas 1992, Howard-Williams and Lenton 1975, Wetzel and Allen 1972). The Great Lakes contain about 200 species of fish with most of these species using the coastal marshes during some part of their lives (Whillans 1990, Jude and Pappas 1992, Brady 1992). Despite the apparent importance of the macroinvertebrates, comparatively little is known of their distribution and ecology in Great Lakes coastal marshes or in freshwater marshes in general (Krieger 1992). Historical data reveals that, since the mid-1800's, Michigan has lost approximately seventy percent of its original coastal marshes (Prince and Flegel 1995, Jaworski and Raphael 1978). Encroachment upon these ecosystems through human development has resulted in a subsequent increase of human impact upon them (Krieger 1992). The need to protect and monitor the health and productivity of these coastal marshes is of great importance in management and recreational planning.

Traditional approaches for collecting water quality data have been based on chemical monitoring (e.g. APHA 1985). While chemical monitoring of water quality is important, there are several inherent problems associated with its use. A number of forms of degradation imposed on aquatic systems are not fundamentally chemical but are instead physical, such as habitat disturbance. The results of chemical monitoring may also overlook significant discharge loads that occur between periods of data collection (Ramm 1988). Rankin et al. (1990) found that water chemistry data failed to detect 50% of the impairment in Ohio surface waters that was detected with integrated biological and chemical monitoring. Karr (1993) defined biological monitoring as "the use of a biological entity as a detector and its response as a measure to determine environmental conditions." The use of natural benthic macroinvertebrate assemblages is one of the best understood and most economical water quality monitoring systems, and it can be used to complement chemical monitoring of water quality (e.g. Plafkin et al. 1989, Rosenberg and Resh 1993, Karr 1993). A major advantage of this system is that, because benthic macroinvertebrates are continuously exposed and influenced by the environment, they continuously "monitor" water quality and can reveal effects of episodic as well as cumulative pollution and habitat alteration (Olive et al. 1988, Plafkin et al. 1989, Barbour et al. 1996, Fore et al. 1993). Benthic macroinvertebrates may reflect long-term water quality conditions and serve as an integrated measure of discharge effects through changes in community structure and composition caused by these effects (Ramm 1988). Macroinvertebrates have proven to be good biomonitoring tools because they are numerous in almost every aquatic system, are readily collected and identified, and are not very mobile (Chandler 1970, Gaufin 1973, Roback 1974, Hilsenhoff 1982, Rosenberg and

Resh 1993). Macroinvertebrates generally have life cycles of weeks to years in duration, and this is important in assessing past perturbations of short duration; once a macroinvertebrate is eliminated from the ecosystem it will not reappear until the next generation (Hilsenhoff 1977).

The use of biological monitoring to evaluate water quality in streams has a long history beginning with the work of Kolkwitz and Marsson (1908) and Forbes (1913). These early biological monitoring programs emphasized an organism's tolerance of organic pollution (Kolkwitz and Marsson 1908, Chutter 1972, Hilsenhoff 1987). Advances in community and population ecology led to the use of diversity indices (Wilhm and Dorris 1968, Hughes and Gammon 1987) which acted as measures of species richness and evenness (review in Fausch et al. 1990). These early biomonitoring ideas began when investigators discovered a number of biological patterns associated with increased human disturbance within a watershed. Such patterns included the decline of a number of species (Metcalfe 1989), the disappearance of a small group of intolerant species (Hilsenhoff 1977), and a decline of trophic specialists while trophic generalist increase (Metcalfe 1989). It is now widely accepted that pollution of a stream reduces the number of species in the stream while frequently creating an environment that is favorable to a few species (Hilsenhoff 1977). Total taxa richness and taxa richness of intolerant groups such as mayflies, caddisflies, and stoneflies often decline as human influences increase (Lenat 1988, Ohio EPA 1987, Plafkin et al. 1989). In contrast, relative abundance and dominance of Chironomidae and Oligochaeta often increase with increasing human influences (Lenat 1988, Ohio EPA 1987, Ford 1989, Barbour et al. 1996).

In the last decade, biological monitoring and indices have been increasingly used to interpret how similar an assemblage at a site is to the potential assemblage that would have occurred there if the site were undisturbed (Karr 1991, Gerritsen 1995, Rosenberg and Resh 1993). This interpretation relies on use of regional, reference data for relatively undisturbed sites. The indices currently used are variations of the Index of Biotic Integrity (IBI) developed by Karr and his colleagues (e.g., Karr 1981, Karr et al. 1996) for fish communities of streams. The concept was extended to the benthic macroinvertebrate assemblages of streams by the Ohio Environmental Protection Agency (Ohio EPA 1987), and by Plafkin et al. (1989) and Kerans and Karr (1994). Elements of community structure and composition are the most widely used assessments of stream biological conditions because community structure is a result of both long-term environmental factors and critical conditions of short duration (e.g., Hilsenhoff 1977; Karr et al. 1986, Lenat 1988, 1993; Plafkin et al. 1989). These monitoring programs and indices are currently used by several states for assessment and management of streams (Southerland and Stribling 1995, Ohio EPA 1987, Kerans and Karr 1994, Barbour et al. 1996, Gerritsen 1995).

In comparison to the large body of literature that exists on the role of freshwater macroinvertebrates in stream communities as they apply to their use as water quality indicators (Karr 1991), the macroinvertebrate fauna of Great Lakes coastal marshes has received only limited study (Krieger 1992). As in stream systems, the macroinvertebrate community structure in coastal marshes may provide a sensitive index to pollution inputs. In order to monitor the health and productivity of these coastal marshes, it is necessary to first obtain descriptive data on the distribution and life history of the macroinvertebrate community and to examine the response of these communities to pollution.

Objectives

The overall goal of this study was to document anthropogenic impacts on the macroinvertebrate fauna of a Great Lakes coastal marsh by comparing it to a nearby reference marsh which appeared to have been less impacted by human activities. Specific objectives were to:

1) Compare the diversity and community composition of the macroinvertebrates in a "reference" Great Lakes coastal marsh with diversity and community composition of a nearby human impacted marsh.

2) Determine which macroinvertebrates in these coastal marsh were most susceptible to pollution from a small community such as Cedarville, Michigan.

3) Select and test various metrics for the use in the future development of an multimetric index of ecological integrity for use in northern Lake Huron coastal marshes.

Description of Study Sites

This study was conducted in two protected bays along the northern Lake Huron shoreline at the southeastern shore of Michigan's Upper Peninsula (Figure 1). The study sites were selected based on the premise that test sites which have similar characteristics in the absence of water quality impairments would be expected to yield equivalent macroinvertebrate communities (Barbour 1992). The two sites selected were coastal, or lacustrine, marshes, that were less than 2 km apart and were part of the Les Cheneaux Islands marsh complex (Figure 1).

A small coastal marsh immediately adjacent to the town of Cedarville, Michigan at the northwestern end of Cedarville Bay (45° 59' N latitude, 85° 21' W longitude), was selected as the impacted site. This marsh typically receives discharge from the Cedarville domestic wastewater treatment lagoon system via Pearson Creek, two times per year in the spring and the fall. Discharge occurred five times during 1996 due to heavy rainfall causing the

Study Location

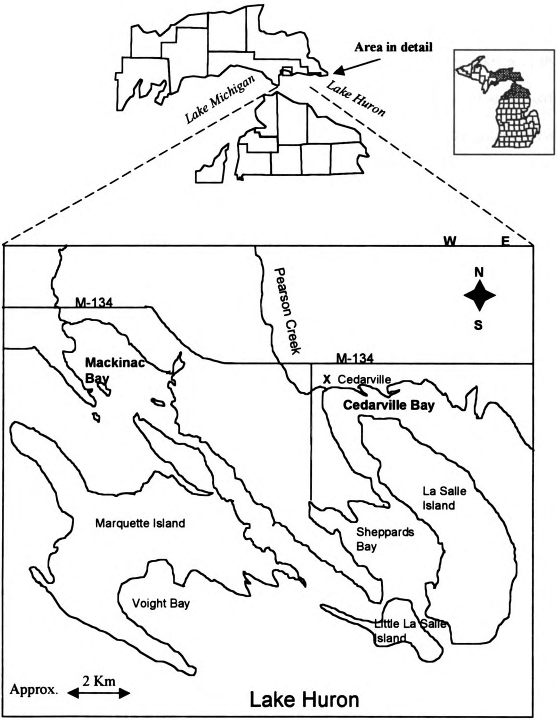

Figure 1. Location of study site in Northern Lake Huron Bioreserve, Cedarville and Mackinac marshes, Lake Huron, Michigan.

wastewater treatment ponds to fill with excess water. Partial discharge occurred May 21-25 and the remainder of the treatment ponds were discharged June 2-5. Fall discharge occurred with an initial partial release September 26-30 and again October 18-24. The final discharge for 1996 occurred November 14-18. The wastewater treatment lagoons consist of three treatment ponds one of which is aerated to promote decomposition and as an aid in settling particles out of solution. Sewage effluent is chemically injected with 3,000 gallons of $FeCl_2$ every six months. $FeCl_2$ was initially added as a means of odor control, however it also reduces phosphorus output from the lagoon through chemical precipitation (Landerville pers. comm.).

Other anthropogenic impacts on Cedarville marsh include the partial filling of the wet-meadow zone and separation of this wet-meadow zone from the deeper emergent marsh, urban stormwater runoff from Cedarville, MI, and impacts associated with marina maintenance and traffic. A portion of the wet meadow zone along Pearson creek was filled beginning in 1911 with the construction of a road separating the wet meadow zone from the deeper portion of the marsh. Filling continued with the addition of a lumber retail store in 1939. The upper end of Cedarville Bay also receives storm-water runoff from Cedarville, Michigan (Population 2000). In addition, a small marina supporting 57 boat slips and a heavily used, public boating access ramp was located within 100 meters of the study sites.

A similar sized marsh at the northwestern end of Mackinac Bay (46° 00' N latitude, 85° 25' W longitude) was selected as the reference site. This coastal marsh was geomorphically similar to Cedarville marsh but received no wastewater discharge, was not immediately adjacent to a town and had no marina or public-access boating ramp near the site. There were several houses around this bay but none were closer than 1 km of the sampling sites.

These two marshes had similar geomorphological characteristics within the marshes in terms of their location in relation to stream discharge, size and shape of their watersheds and underlying geology. The two marshes were located on the Niagara escarpment, which consisted primarily of resistant limestone and dolomite (Albert et al. 1986). The dominant land use in both was forest with a mixture of northern hardwood forest and dense stands of northern white-cedar, balsam fir and spruce along the lakes (Albert et al. 1986, personal observation). The average growing season for this area is 125 days long, with the growing season heat sum relatively low at 1860° C days (Albert et al. 1986). The catchments of both sites were drained by small first order streams, which were relatively similar in length, width, and area drained.

Within the marsh areas of both bays, the study sites were located in the emergent plant zones dominated primarily by *Scirpus acutus*. Macrophytes grew from late May through July, and started senescing in August. Dead stems were partially scoured from the marsh by ice during the winter. Following ice-out, the emergent stand re-grew from rhizomes. The emergent zone at the impacted marsh extended approximately 200-300 meters from the shore at the road into Cedarville Bay. The emergent zone at the reference marsh extended for several hundred meters from the wet meadow zone out into Mackinac Bay. The density of *S. acutus* was fairly consistent between the two marshes, with an overall summer average of 72 stems per 0.25 m^2 at the reference marsh, and an average of 74 steams per 0.25 m^2 at the impacted marsh. However, major differences in the plant communities occurred for the submersed plants in this zone with greater dominance by *Utricularia spp.*, *Myriophyllum spp.*, and *Ceratophyllum spp.* at the impacted site. Water temperatures were usually homogeneous between the two marshes, but when differences occurred, they never exceeded 2°C. Both sites were protected from direct, harsh wave action from Lake Huron by a network of islands.

Substrate at the sampling sites of Mackinac Marsh appeared to be composed primarily of loose organic matter (33%), with lesser fractions of clay and sand. Cedarville's marsh substrate at the sampling site appeared to consist primarily of organic matter (70%), with lesser fraction of clay (Personal observation).

Methods

Land-use/Land-cover Characterization of Sites

Watersheds for Mackinac and Cedarville marsh were delineated, digitized and overlaid on to 1978 Michigan Resource Information System (MIRIS) Land-use/Land-coverages of Mackinac county. These coverages were used to determine percent Land-use/Land-cover in both Mackinac's and Cedarville's watersheds.

Chemical-Physical Data Collection

Environmental data on physical and chemical data were collected at each of the nine sampling sites in each marsh, on each invertebrate sampling date to characterize the water within the marsh. Water depth was measured at each site to the nearest 0.5 cm. Vegetation type and density was determined by counting the number of emergent stems taken at a random distance and direction from the sampling point in a 0.25 m^2 plot. Random direction (0° to 360° in increments of 10°) and random distance (1 m to 5 m in increments of 0.5 m) were obtained from a table of randomly generated numbers. Dissolved oxygen (YSI model 51B oxygen meter), pH (Altec monitor II meter), and conductivity (YSI model 31 conductivity bridge) were determined in situ using appropriate, calibrated probes. Water and air temperatures were measured just above the sediment surface to the nearest $\pm$ 0.1°C using a hand-held mercury thermometer.

For each date at each site, water samples were collected at one-half the depth of the water column in acid washed, deionized water rinsed opaque plastic bottles. All samples were immediately placed on ice and transported to the laboratory within three hours. Alkalinity (APHA 1985) and turbidity (HACH model 2100A turbidimeter) were determined in the field laboratory. Sub-samples were filtered through 0.45μm millipore filters, frozen at -20° C, and later sent to Michigan State University's Soils Testing Laboratory for analysis of Cl, NH_4-N, NO_2-N, NO_3-N and SRP. All samples were tested at a level of detection of 0.01 mg/L except CL which was tested at a level of detection of 1.0 mg/L. Results were then used to examine differences in water quality between the two sites, and to examine if a gradient in littoral water quality existed as a factor of distance from discharge.

Invertebrate Sampling

A fixed transect was established within the *Scirpus acutus* dominated plant communities of each marsh. The transects ran parallel to the shoreline 55 meters into the emergent plant community in each marsh, beginning 10 meters west of the mouths of the streams. Sampling sites were located every 50 meters for a total of nine stations (within each marsh) resulting in the last station being 410 meters west of the stream mouths. This design was used to minimize habitat variability for such factors as depth, water temperature, and vegetation density, while at the same time establishing a gradient moving away from the source of discharge. All transects were placed within a predominantly monotypic *S. acutus* zone to minimize the possible confounding effects of varying vegetation types.

Macroinvertebrates were sampled monthly from June through September in 1996 beginning two weeks after *S. acutus* stems extended above the water surface. Sampling occurred on June 24-25, July 24-25, August 18-19 and September 28-29. On each date, macroinvertebrates were collected using a sediment sample collected with a coring device and a water column/vegetation sample taken with a standard number of sweeps with a D-framed dip net. The sediment and plant associated samples were taken at a random distance and direction from each of the nine fixed sampling station points along the transect. For each sample taken at each point along the transect, a random direction (0° to 360° in increments of 10°) and a random distance (1 m to 5 m in increments of 0.5 m) were obtained from a table of randomly generated numbers. Care was taken never to sample the same direction twice from any particular fixed point, nor was any sample taken within one meter of the transect itself.

Vegetation/water column 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 um mesh net in order to capture smaller macroinvertebrates. Samples consisted of 3 sweeps at the surface of the water column, 3 sweeps in the center of the water column and 3 sweeps along the sediment surface at the bottom of the water column. Each sweep covered 0.15 m^2 of substrate (net width of 0.3 m x 0.5 m length of pass); therefore, the total composite sample was taken from an area of approximately 0.90 m^2.

In the field, all samples were washed through a 250um sieve, and then transferred to 1 liter large mouth bottles. The macroinvertebrates along with all debris left in the sieve after washing, were preserved in 95% ethanol containing 100 $mg.l^{-1}$ of Rose Bengal dye with enough 95% ethanol added to the jar to produce a concentration of about 70% when mixed with the water in the debris (Mason and Yevich 1967). The dye stained the protein in the sample red which helped to increase the accuracy of picking individual specimens from the debris in the laboratory. After returning to the laboratory, sample jars completely filled with macroinvertebrates and debris were drained and rinsed again through a 250um screen. The ethanol was then replaced with 70% ethanol. Invertebrates were picked from the debris and sorted with the use of a 10x dissecting microscope.

Organisms were classified to the lowest Operational Taxonomic Unit (OTU'S) possible throughout the study. The OTU's and the references used for identification, are shown in Table A-1. Every effort was made to sort invertebrates to species level, but some taxa that were taxonomically difficult (e.g. Chironomidae) or that were not properly preserved (e.g. Nematoda) were grouped at higher taxonomic levels. Chironomidae larvae were identified to genus or species group for the June water column samples at the site nearest the source of discharge in both the impacted and reference wetland (Table A-2), otherwise they were sorted to sub-family or tribe. Zooplankton were identified to the species level at the site nearest the source of discharge in both the sediment and water column samples of the impacted marsh and the reference marsh in June and September. Specimens were also sent to taxonomists whenever possible for verification or identification of species present as a means of most accurately reflecting biodiversity for each of the marsh areas.

Benthic invertebrates were sampled using a corer made of a 1.22 m long, 5 cm (internal diameter) Plexiglas tube capped with a 4.5 cm rubber stopper prior to pulling the corer out of 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 the water column from those residing in the sediments. All benthic samples were preserved with the same method used for vegetation/water column samples. In the laboratory, each sediment sample was divided into sixths before it was picked, using a sub-sampling device similar to that described by Waters (1969). To insure that an adequate number of organisms was obtained to characterize the sample, sub-samples were picked until at least 50 organisms were found, when possible. Once picking of a sub-sample was started, it was sorted completely.

The numbers of the invertebrates in each sediment sample were converted to number per m^2 based on the following formula:

$$\text{Area: } \pi r^2 = \pi (0.025\text{m})^2 \times 2 \text{ cores per sample}$$
$$= 3.93 \times 10^{-3} \text{ m}^2$$

Macroinvertebrates were classified into biotic categories describing trophic status (omnivore, detritivore, herbivore, carnivore, scavenger), functional-feeding group, and habitat using information from Merritt and Cummins (1996) to compare macroinvertebrate communities between sites.

Macroinvertebrate communities that demonstrated a response to impact were then selected for determination of whether a gradient existed within the impacted marsh with distance from discharge. This was accomplished by plotting the selected taxa with increasing distance from discharge with a linear regression. If no gradient was detected, the selected taxa from the three sites closest to discharge were pooled, and compared to the same taxa pooled from the three sites furthest from discharge, using a nonparametric Mann-Whitney U test for two independent samples, to determine if a difference was present.

RESULTS

Comparisons of Land-use/Land-cover in watersheds

The reference and impacted watersheds were both dominated by forest cover (Table 1). Although land-use was primarily forested for both the watersheds, Cedarville Bay's (impacted) watershed had a higher percentage of urban and agricultural land than did Mackinac Bay's (reference) watershed, and lower percentage of hardwood forest and wetlands (Table 1).

Table 1. Total area and Percent Land-use/Land-cover in Cedarville and Mackinac bay's watershed (MIRIS 1978).

	MACKINAC	CEDARVILLE
	Percent	
Coniferous forest	39	41
Hardwood forest	39	34
Urban	5	12
Agricultural	1	4
Wetland	7	4
Other	9	5
Total area of watershed	1.31 km^2	1.70 km^2

Comparisons of Water Quality Data

Greatest differences in water quality between the impacted marsh (Cedarville) and the reference marsh (Mackinac) occurred during times of active discharge via Pearson creek from the wastewater treatment lagoon in September (Table A-3). The impacted marsh was characterized by higher levels of Cl, NH_4-N, NO_3-N, NO_2-N, SRP, conductivity, turbidity, alkalinity and lower levels of dissolved oxygen when compared to the reference marsh during June and September but not during July and August (Table A-3). These differences in water quality during the times of active discharge from the wastewater treatment lagoon into the creek in June and September were greatest near the mouth of Pearson creek , and decreased to levels similar to those in the reference marsh between 210 and 260 meters from the creek mouth (Figures 2-6). The relatively limited area of the marsh with detectable impacts was also documented in detail by Grant (1994). The well-mixed nature of the marsh, may contribute to a decreasing impact with distance from discharge. June samples indicated very low chemical impact at the time of sampling, approximately two weeks after the actual discharge event, suggesting that the marsh had returned to levels near background within two weeks after the sewage discharge ceased.

Elevated Cl concentrations occurred during or immediately following discharge from the wastewater treatment lagoon at the impacted marsh in June and September (Figure 2). These elevated Cl concentrations decreased to mean background levels at 260 m from the mouth of the creek, indicating rapid dilution and mixing within the marsh water. However, Cl concentrations were similar in the two marshes on other sampling dates, with concentrations ranging between 12-21 mg Cl/l (Table A-3). Cl concentrations in natural freshwater systems average 8.3 mg Cl/l (Wetzel 1983). The slightly elevated Cl levels of 12-21 mg Cl/l may indicate presence of evaporite deposits within the underlying bedrock of this

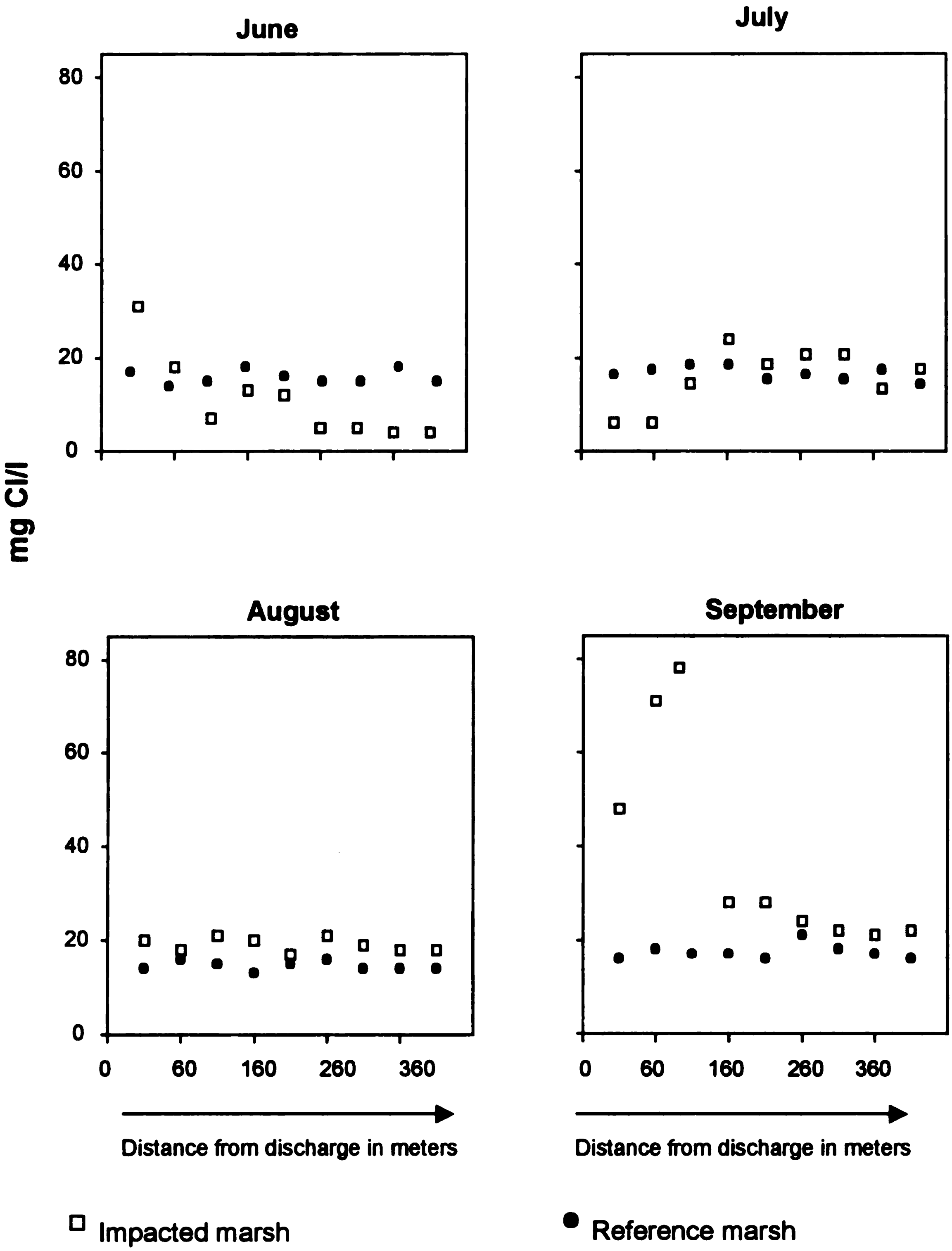

Figure 2. Chloride concentration gradient moving away from the source of discharge at Cedarville (impacted) and Mackinac (reference) marshes, 1996.

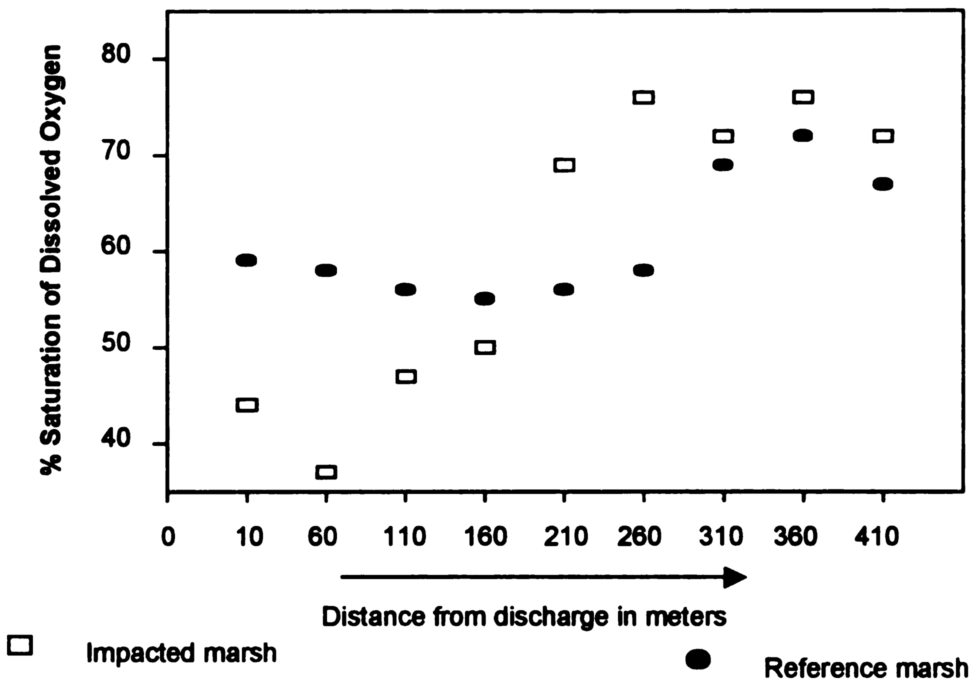

Figure 3. Percent saturation of dissolved oxygen moving away from the source of discharge at Cedarville (impacted) and Mackinac (reference) marshes, September 1996.

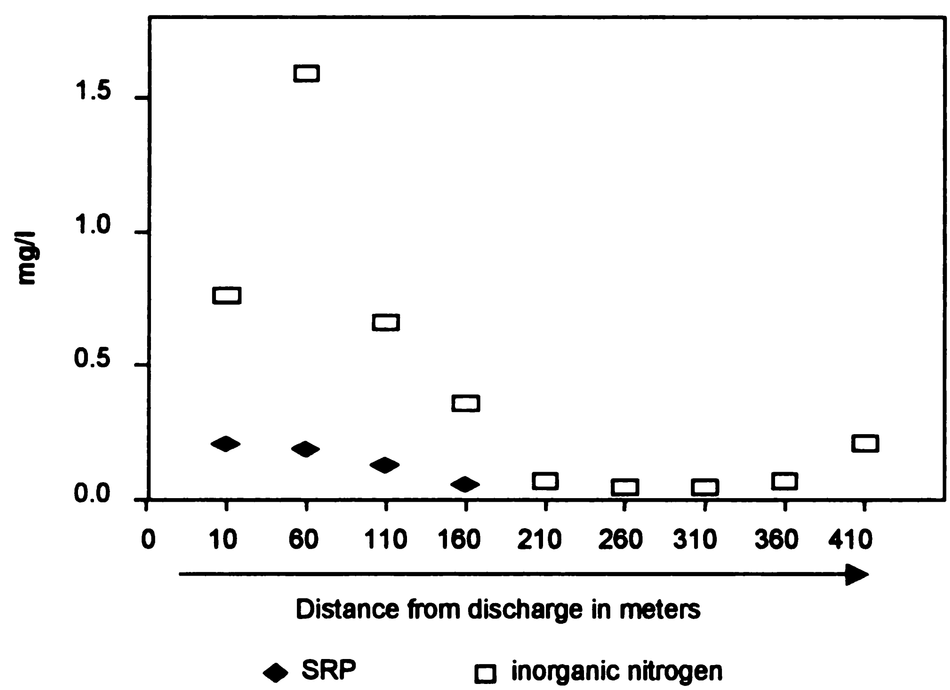

Figure 4. Concentration gradient of soluble reactive phosphorous (SRP) and inorganic nitrogen at Cedarville (impacted) marsh, September 1996.

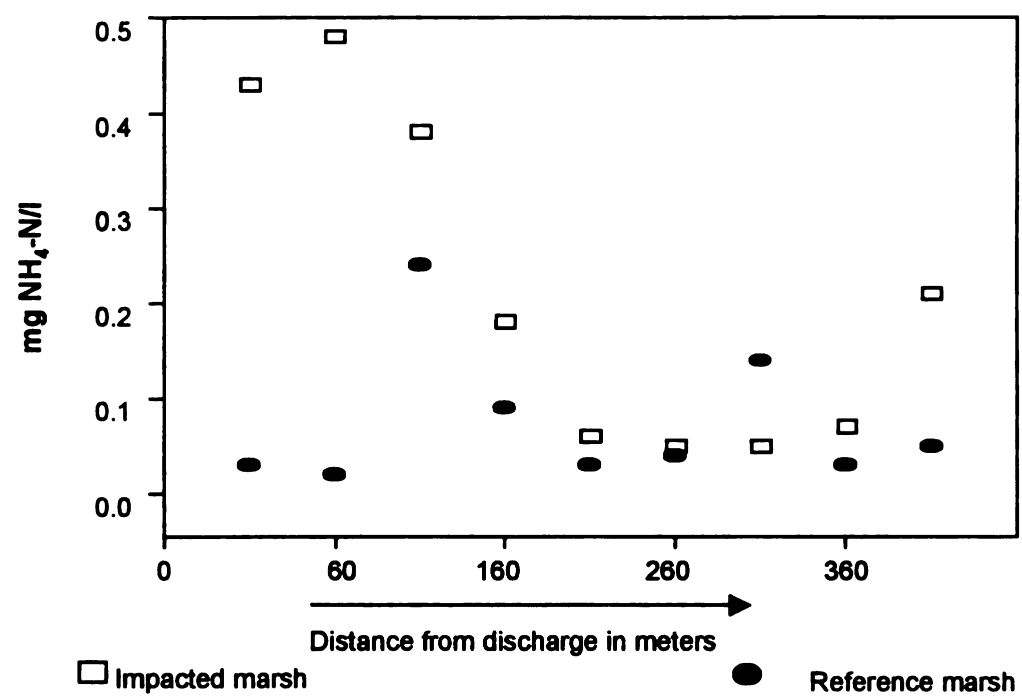

Figure 5. Concentration gradient of NH_4-N moving away form the source of discharge at Cedarville (impacted) and Mackinac (reference) marshes, September 1996.

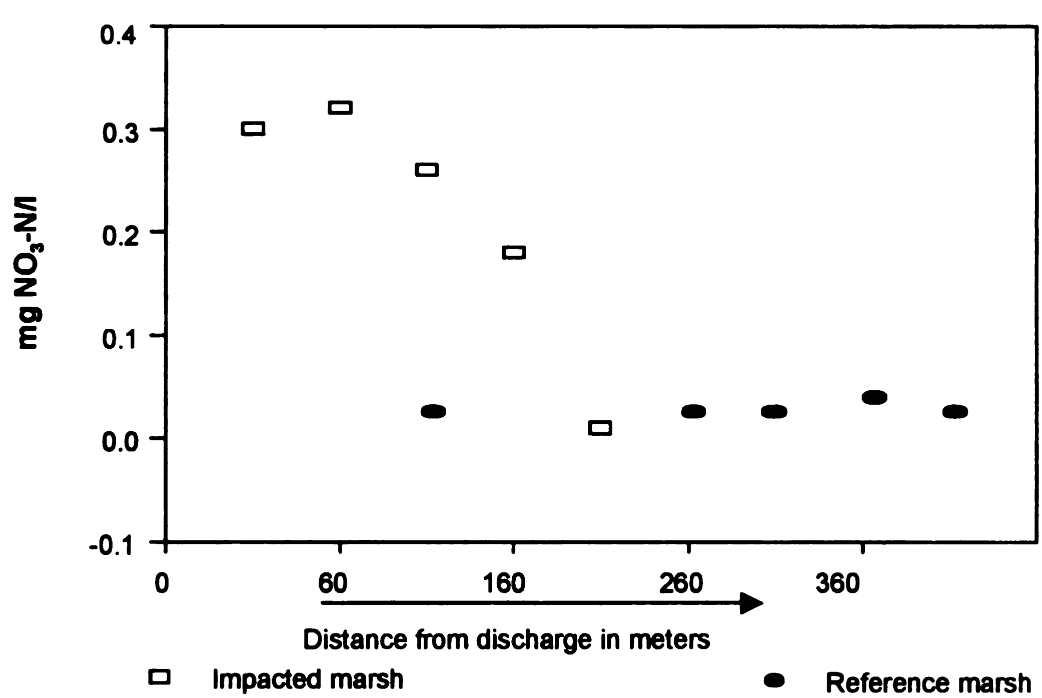

Figure 6. Concentration gradient of NO_3-N at Cedarville (impacted) and Mackinac (reference) marshes, September 1996.

catchment. Road salt from local roads seems unlikely to be the cause of the elevated Cl levels because impacts from road salt would be expected to decline over the course of the summer but did not in this study (Figure 2). Cl concentrations remained relatively constant in concentration and distribution within the reference marsh throughout the sampling season (Figure 2).

Turbidity levels were low at both the impacted and reference marsh never exceeding 5 NTU's (Table A-3). The highest turbidity levels at the impacted marsh occurred during June and September, perhaps indicating slight elevations in turbidity associated with wastewater discharge, but trends were inconclusive for other sampling dates. June turbidity levels at the impacted marsh were highest at the sites nearest discharge and decreased with distance from discharge. Turbidity at neither site was ever high enough to cause major differences in biota (EPA 1986), especially given the variance from <1 to 5 NTU's for both marshes.

Dissolved oxygen ranged from 61% saturation to supersaturated values up to 114% saturation from June through August, at the impacted and the reference marsh with no consistent differences between the sites (Table A-3). Dissolved oxygen concentrations were substantially below saturation during September for both the reference and impacted marshes. Plant senescence leading to a decrease in primary productivity and an increase in respiration is the most likely explanation for the undersaturation at both sites (Table A-3). Dissolved oxygen saturation was significantly lower at the impacted site compared to the reference site during wastewater discharge in September. Dissolved oxygen levels in the impacted marsh fell to levels below 5 mg/l at the sites less than 210 meters from discharge (Figure 3). The U. S. Environmental Protection Agency (1986) suggested dissolved oxygen levels below 5 mg/l are incompatible with maintenance of diverse aerobic aquatic communities. The low dissolved oxygen levels may be an important factor for many species of aquatic life in the areas near the source of discharge at the impacted marsh. If dissolved oxygen levels dropped to low levels following the June discharge event, they had recovered by the time of sampling two weeks after discharge had ceased (Table A-3). Low dissolved oxygen levels may have not occurred in June due to discharge coinciding with periods of high primary productivity. Dissolved oxygen was monitored during mid-day only, so periods of low dissolved oxygen that may have occurred at night would not have been detected.

During wastewater discharge in September, the two limiting nutrients, Soluble reactive phosphorus (SRP) and inorganic N, were elevated for only 160 meters from the source of discharge along the transect (Figure 4). SRP was always below limits of detection (0.01 mg P/l) in the reference marsh. SRP was at or less than limits of detection in June, July and August at the impacted marsh (Table A-3). From September 26 to 30, the Clark township wastewater treatment lagoon discharged sewage effluent containing a daily average of 0.96 mg P/l of total phosphorus approximately 2.39 km upstream from the impacted marsh. SRP was detected at a maximum concentration of 0.21 mg P/l nearest the source of discharge on September 29, 1996 and returned to non-detectable concentrations at 210 m from discharge (Figure 4). Aqua-Terrra Labs performed a water quality survey at the impacted marsh in 1994 and found that only 64% of the P discharged from the wastewater lagoon actually entered the marsh (Grant 1994). This suggests that much of the P loss can be attributed to rapid assimilation by the stream biota, chemical precipitation, and sorption onto sediment particles. Even so, SRP exceeded 10μg P/l from the stream mouth to 160 meters along the marsh transect, and levels exceeding 10μg P/l are known to stimulate plant growth in many lakes (Wetzel 1983).

NH_4-N levels generally ranged from 0.03 to 0.07 mgN/l at the reference and impacted marshes from June through August (Table A-3). The highest concentration of NH_4-N occurred in September at the impacted marsh, near the source of discharge, reaching concentrations of 0.48 mg NH_4-N/l (Figure 5). NH_4-N returned to background levels of 0.06 mg NH_4-N/l at 210 meters from discharge at the impacted marsh in September (Figure 5). NH_4-N concentrations appeared to be relatively constant in both concentration and distribution at the reference marsh (Table A-3).

NO_3-N was only detected at the impacted marsh during June and September and was below the limits of detection of 0.01 mg NO_3-N/l in July and August (Table A-3). June NO_3-N concentrations were low, at levels no greater than 0.03 mg NO_3-N/l. However, during September NO_3-N was detected at levels as high as 0.32 mg NO_3-N/l near the source of discharge and decreased in concentration with distance from discharge until no longer detected at distances greater than 210 meters from discharge (Figure 6). NO_3-N was always below the 0.01 mg NO_3-N/l limit of detection in the reference marsh expect during September. Concentrations in September were never greater than 0.04 mg NO_3-N/l, or an order of magnitude lower than values detected near the mouth of the stream during wastewater discharge.

NO_2-N was only detected (0.01 mg NO_2-N/l = limit of detection) at the impacted marsh during September, at the three sites nearest the source of discharge at levels no greater than 0.03 mg NO_2-N/l (Table A-3). NO_2-N was never detected at the reference marsh.

Both marshes tended to be characterized by pH values from 7.0 to 9.5 and usually in the 8.0 to 9.0 range. There were no consistent trends in pH at the impacted marsh even during wastewater discharge in September, except for somewhat more variable pH values (Table A-3). Conductivity and alkalinity were generally lower at the impacted marsh compared to the reference marsh, except in September when wastewater discharge was actively occurring at the impacted marsh (September) (Table A-3).

Comparison of Macroinvertebrate Community Dynamics

The plant and sediment associated macroinvertebrate communities within the *Scirpus acutus* zone of the reference marsh were dominated numerically by aquatic insects for all sampling periods (Figure 7). Aquatic insects were less dominant at the impacted site than they were at the reference site for both the sediment and plant associated communities (Figure 7), and this was particularly true for the sediment community (Figure 7B). Sediment associated aquatic insects represented 50 to 80% of the macroinvertebrate abundance of the reference marsh, but only represented 20 to 46% of the sediment community at the impacted marsh (Figure 7B). Likewise, aquatic insects represented 50 to 63% of the macroinvertebrate community of the plant associated community of the reference marsh, and only 28 to 50% of the plant associated community of the impacted marsh (Figure 7A).

Macroinvertebrates of both the plant and sediment associated communities tended to reach peak abundances in September with numbers tending to increase as the season progressed from June through September (Figure 8). Macroinvertebrates reached their peak mean density of 35,000$\cdot$m^2 in the sediment of the reference marsh during September (Figure 8B), approximately 80% of which were aquatic insects (Figure 7B). The macroinvertebrate

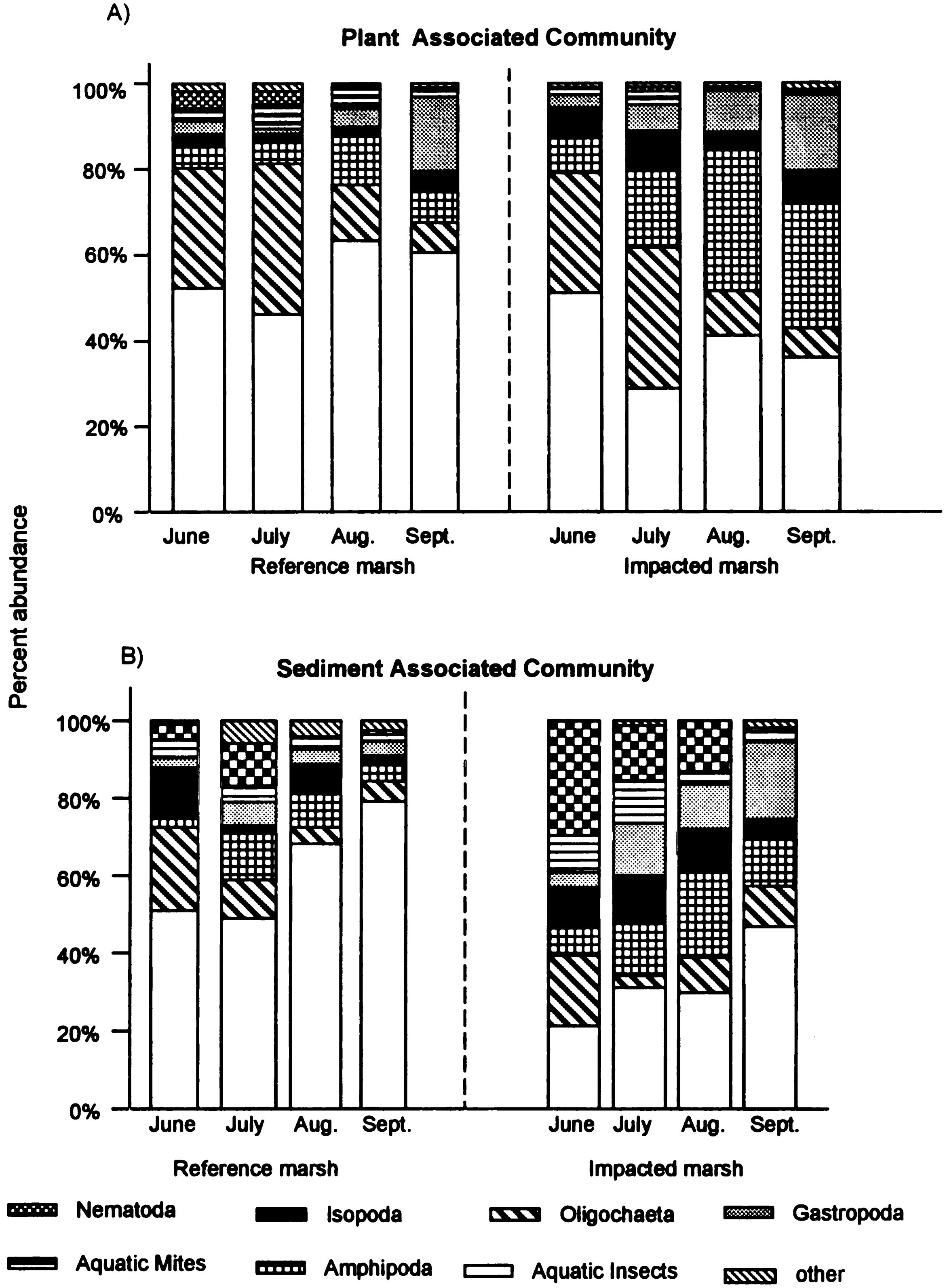

Figure 7. Community composition of the macroinvertebrate community at Cedarville (impacted) and Mackinac (reference) marshes, Lake Huron, 1996. A) Plant associated community (dip-net samples) and B) Sediment associated community (core samples).

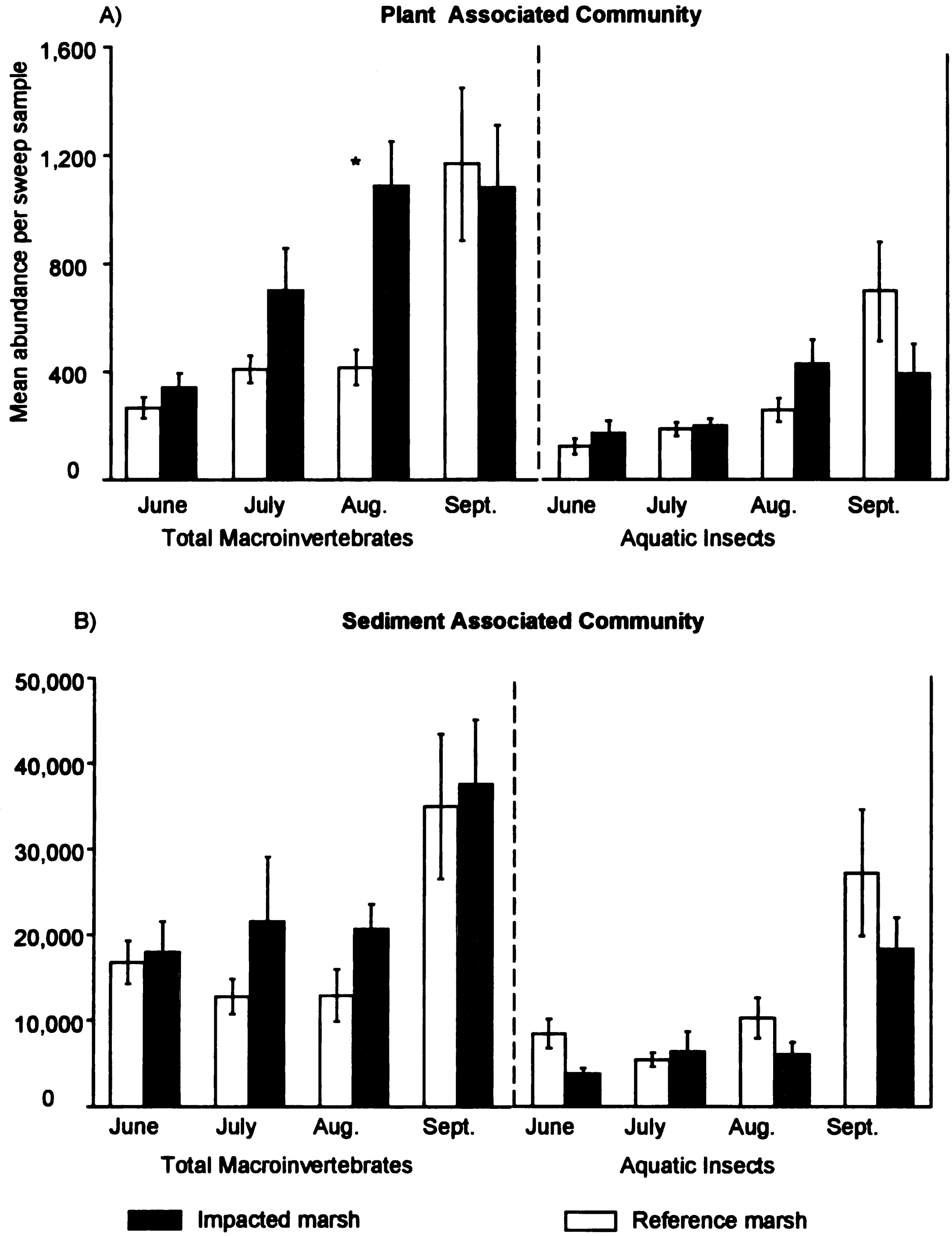

Figure 8. Abundance trends, including standard error, of total macroinvertebrates and insects at Cedarville (impacted) and Mackinac (reference) marshes, Lake Huron, 1996. A) Plant associated community (dip-net samples) and B) Sediment associated community (core samples). *Mann-Whitney U test: Significant at $\alpha = 0.05$.

community in the sediment at the impacted marsh reached a similar peak mean density in September, of 39,836·m^{-2}; but only 35% of these macroinvertebrates were aquatic insects (Figures 8B and 7B). Non-insect macroinvertebrates were more abundant than were insects at the impacted site (Figures 7 and 8). Amphipoda, Oligochaeta, Isopoda, Gastropoda, Nematoda, and Hydracarina were important members of the non-insect, invertebrate fauna. Each major group of macroinvertebrates will be discussed separately below.

Aquatic Insecta:

Plant associated aquatic insects reached their greatest mean abundance of approximately 700 individuals per sweep net sample in September at the reference marsh (Figure 8A). Plant associated aquatic insects at the impacted marsh reached a lower peak mean abundance of 400 individuals per sample in August but these differences between the reference and impacted sites were significant ($p< 0.05$) only during August (Figure 8A). Aquatic insects reached a peak abundance in September in both the impacted and reference marsh sediment (Figure 8). Sediment associated aquatic insects at the reference marsh reached a peak density of 27,000·m^{-2} compared with a peak density of 18,660·m^{-2} at the impacted marsh (Figure 8B). These differences were not statistically significant ($p< 0.05$) for any sampling date (Figure 8B).

Aquatic insects associated with the plant community in the reference marsh were represented by 73 taxa from 37 families (Table 2). Thirty-four taxa of insects, representing 17 families, were collected from the sediment core samples in the reference marsh (Table 2). The Chironomidae, Ephemeroptera, and Trichoptera were dominant insect groups within the plant and sediment associated communities at the reference marsh (Figure 9), and Odonata

Table 2. Aquatic insects collected with standardized dip-net sweep sampling from *Scirpus* dominated zones (mean number/sample±S.E) and with cores from sediments in these zones (mean number·m^{-2}±S.E) from the reference (Mackinac) and Impacted (Cedarville) marshes in 1996. * was not collected in the samples.

TAXON	Mackinac		Cedarville	
	Dip-Net	Core samples	Dip-Net	Core samples
	Mean #/sample±S.E.	Mean #·m^{-2}±S.E.	Mean #/sample±S.E.	Mean #·m^{-2}±S.E.
<u>COLEOPTERA</u>				
Chrysomelidae				
Donacia spp.	.14±.10	21.21±21.21	.14±.10	35.34±21.21
Nehaemonia spp.	.34±.20	*	.06±.06	*
Curculionidae	.03±.03	*	*	*
Dytiscidae				
Agabus spp.	.03±.03	*	*	*
Elmidae				
Duiraphia spp.	.06±.06	*	*	*
Gyrinidae				
Dineutus spp.	.08±.08	*	.03±.03	*
Gyrinus spp.	.14±.08	*	.06±.03	*
Haliplidae				
Halplus spp.	.03±.03	*	.19±.08	*
Peltdytes spp.	*	*	.19±.05	*
Ptilidae	.03±.03	*	*	*
<u>COLLEMBOLA</u>				
Isotomidae				
Isotomurus tricolor	*	*	.03±.03	*
Sminthuridae				
Entomobrya nivalis	*	*	*	.06±.06
Pseudobourletiella spinata	.03±.03	*	.11±.08	*
<u>DIPTERA</u>				
Ceratopogonidae				
Bezzia/Palpomyia spp	.42±.31	148.43±72.43	1.47±.71	21.21±21.21
Culicoides spp.	.17±.13	*	*	21.21±21.21
Probezzia spp.	.06±.06	*	*	*
Serromyia spp.	*	7.07±7.07	*	*
Sphaeromias spp.	*	21.21±21.21	*	*
Chironomidae*				
Chironominae				
Chironomini	75.08±32.75	4134.86±1783.24	118.33±30.99	5605.00±2445.00
Tanytarsini	38.97±14.32	692.68±265.60	27.25±11.47	657±576.82
Orthocladiinae				
Corynoneura spp.	4.08±.96	106.02±63.91	18.14±5.54	98.96±24.48
Others	25.67±13.65	494.77±212.98	55.06±11.87	417.02±155.02
Tanypodinae	38.64±24.66	2325.42±1389.45	21.87±4.11	1329.00±673.81
Empididae	.14±.07	134.30±53.40	.08±.05	*
Ephydridae				
Hydrellia spp.	*	*	.02±.02	*

Table 2 (cont'd)

TAXON	Mackinac		Cedarville	
	Dip-Net	Core samples	Dip-Net	Core samples
	Mean #/sample±S.E.	Mean #·m^{-2}±S.E.	Mean #/sample±S.E.	Mean #·m^{-2}±S.E.
Sciomyzidae	*	*	.03±.03	*
Stratiomyidae				
Odontomyia spp.	.11±.11	*	.03±.03	*
Tabanidae				
Chrysops spp.	*	42.41±42.41	.06±.06	14.14±14.14
Haematopota spp.	.03±.03	*	*	*
Hybomitra spp.	.28±.24	*	.08±.08	*
EPHEMEROPTERA				
Baetidae				
Callibaetis ferrugineus	3.7±2.3	127.23±54.75	2.17±.80	*
Procleon viridocularis	.42±.29	*	*	*
Caenidae				
Caenis amica	24.95±21.84	1229.86±1060.79	.17±.11	42.41±42.41
Caenis latipennis	8.81±5.81	770.43±659.46	.08±.08	*
Caenis youngi	17.03±4.91	1887.19±398.4	*	*
Ephemerellidae				
Eurylophella funeralis	1.03±.63	7.07±7.07	.03±.03	*
Ephemeridae				
Hexagenia limbata	.58±.58	212.05±184.86	*	*
HEMIPTERA				
Belostomatidae				
Lethocerus spp.	.03±.03	21.21±21.21	*	*
Corixidae				
Hesperocorixa kennicotti	*	*	.02±.02	*
Sigara transfiturata	.20±.20	*	*	*
Sigara variabillis	1.00±.82	*	*	*
Palmacorixa buendi	.11±.08	21.21±21.21	.02±.02	*
Trichocorixa sexcincta	*	*	.06±.03	*
Others	.61±.33	7.07±7.07	1.06±.46	21.21±21.21
Hebridridae				
Merragata spp.	.03±.03	*	*	*
Nepidae				
Ranatra spp.	.03±.03	*	.03±.03	*
Mesoveliidae				
Mesovelia mulsanti	.08±.05	*	.58±.45	21.21±21.21
Gerridae				
Gerris comatus	.03±.03	*	.14±.10	*
Trepobates spp.	*	*	.03±.03	*
LEPIDOPTERA				
Noctuidae				
Bellura spp.	.03±.03	*	.06±.06	
Pyralidae				
Acentria spp.	.43±.27	*	2.06±.24	21.21±21.21
Munroessa/Synclita/Neocataclysta spp.	*	*	1.44±1.29	*
Parapoynx spp.	2.10±.94	7.07±7.07	.52±.24	21.21±21.21

Table 2 (cont'd)

TAXON	Mackinac		Cedarville	
	Dip-Net	Core samples	Dip-Net	Core samples
	Mean #/sample±S.E.	Mean #·m^{-2}±S.E.	Mean #/sample±S.E.	Mean #·m^{-2}±S.E.
NEUROPTERA				
Sisyra spp.	*	*	.03±.03	*
ODONATA				
Ashnidae				
Anax junius	.08±.08	*	*	*
Aeshna eremita	*	*	.11±.08	*
Aeshna spp.	*	*	.33±.33	*
Coenagrionidae				
Enallagama boreale	.36±.32	*	*	*
Enallagma ebrium/hageni	4.39±4.17	176.71±91.89	6.42±5.42	204.98±106.02
Enallagma geminatem	.06±.03	*	.11±.04	*
Enallagma vernale	.25±.16	*	*	*
Ischnura verticalis	30.47±15.65	21.21±21.21	19.28±11.10	14.14±14.14
Others	1.70±1.20	*	1.70±1.70	*
Corduliidae				
Epitheca spp.	.42±.25	*	.39±.35	*
Cordulia shurleffi	1.08±.79	*	.25±.43	*
Gomphidae				
Arigomphus cornutus	.03±.03	*	*	*
Libellulidae				
Celithemis spp.	.06±.06	*	*	*
others	.17±.17	*	.56±.35	*
Lestidae				
Lestes spp.	.06±.06	*	.03±.03	*
TRICHOPTERA				
Hydropsychidae				
Ceratopsyche spp.	.03±.03	*	.14±.14	*
Cheumatopsyche campyla	.03±.03	*	*	*
Hydroptilidae				
Agraylea multipuntata	*	21.21±.21	4.44±3.35	35.34±21.21
Hydroptila spp.	*	*	3.67±3.67	*
Orthotrichia spp.	*	7.07±7.07	*	*
Oxyethira spp.	9.09±5.05	28.27±19.99	6.39±3.8	*
Leptoceridae				
Ceraclea spp.	.14±.14	*	*	*
Mystacides interjecta	2.30±1.50	63.62±40.60	.03±.03	*
Mystacides sepulchralis	8.14±3.27	120.16±120.16	1.59±.77	35.34±35.34
Nectopsyche spp.	.22±.12	*	.14±.07	*
Oecetis cinerascens	1.14±1.1	*	1.36±.79	14.14±14.14
Oecetis inconspicua	.03±.03	*	*	*
Oecetis osteni	*	42.40±42.41	*	*
Oecetis persimilis	.03±.03	*	*	*
Oecetis spp.	8.75±2.98	127.23±127.23	1.81±.89	21.21±21.21
Triaenodes aba	.70±.43	*	.17±.11	*

Table 2 (cont'd)

TAXON	Mackinac		Cedarville	
	Dip-Net	Core samples	Dip-Net	Core samples
	Mean #/sample±S.E.	Mean $\#\cdot m^{-2}$±S.E.	Mean #/sample±S.E.	Mean $\#\cdot m^{-2}$±S.E.
Limnephilinae				
Lilmnephilus spp.	.17±.11	*	*	*
Nemotaulius hostilus	*	*	.08±.08	*
Molannidae				
Molanna tryphena	.28±.28	*	*	*
Molanna spp.	.14±.08	56.55±19.99	*	*
Phryaneidae				
Agrypni improba	.56±.37	*	.36±.36	*
Fabria spp.	.03±.03	*	.25±.25	*
Phryganea cinera	14.11±10.87	42.41±42.41	1.45±1.09	21.21±21.21
Polycentropodidae				
Cernotina spp.	*	*	.03±.03	
Polycentropus spp.	1.47±.81	134.30±98.19	2.5±1.33	162.57±63.61
Aphids	*	21.21±21.21	3.67±3.67	*

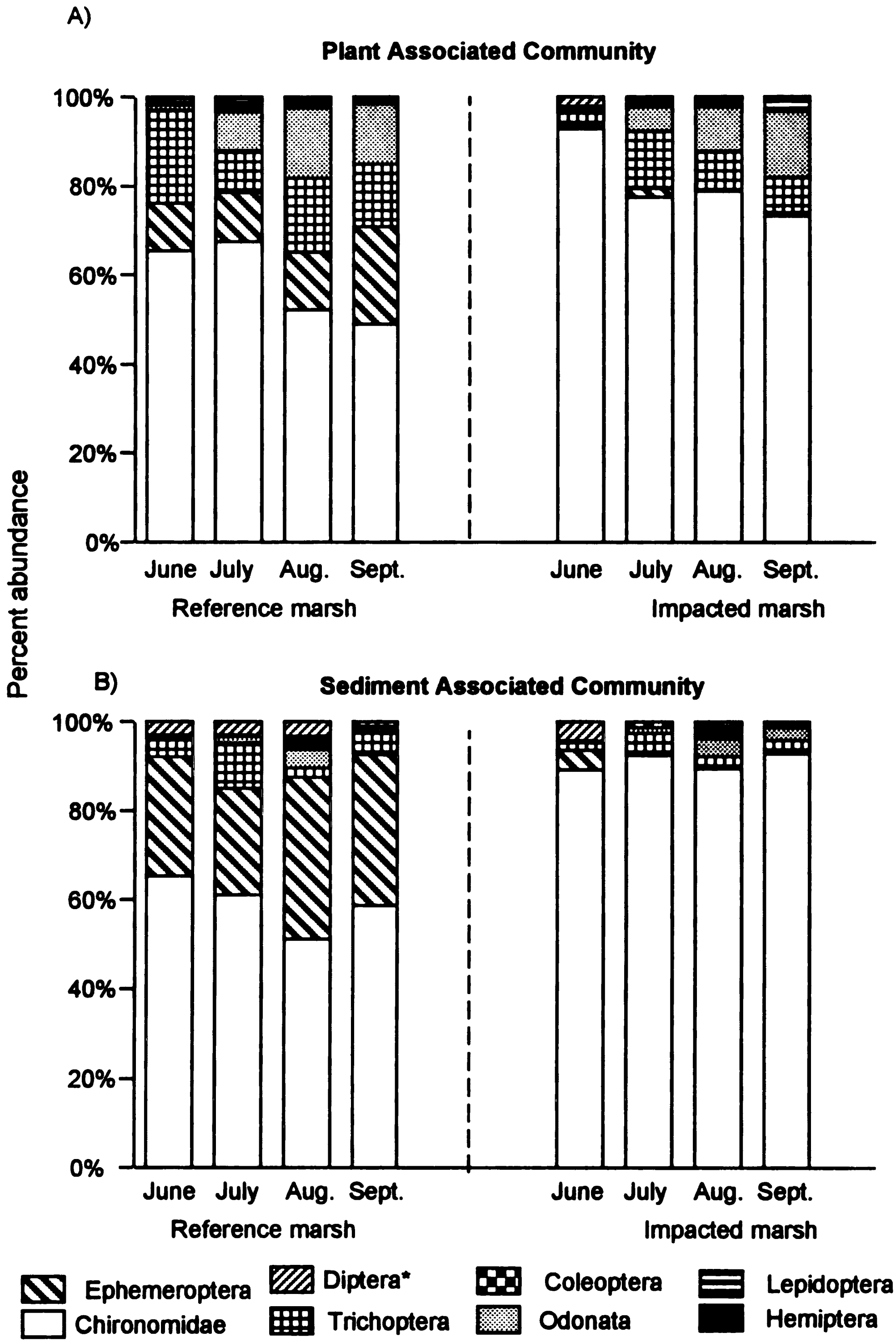

Figure 9. Community composition of the aquatic insect orders at Cedarville (impacted) and Mackinac (reference) marsh, Lake Huron, 1996. A) Plant associated community (dip-net samples) and B) Sediment associated community (core samples). * Diptera group does not include Chironomidae

were also important constituents of the plant associated insects (Figure 9A).

The plant associated community at the impacted marsh was represented by 64 taxa of aquatic insects, from 33 families (Table 2). Chironomidae made up 73-93% of the plant associated aquatic insects in the impacted marsh (Figure 9), as compared to 50-60% of plant associated insects in the reference marsh (Figure 9). The remainder of the plant associated community at the impacted marsh was primarily represented by Trichoptera and Odonata, although neither group ever represented more than 15% of the aquatic insects (Figure 9). Ephemeroptera never comprised more than 2% of the plant associated aquatic insect community in the impacted marsh even though they represented more than 10% of the fauna on every sampling date for the reference marsh (Figure 9A). The impacted marsh sediment community was represented by 22 species of aquatic insects, belonging to 12 families (Table 2). Chironomidae was the only dominant group of aquatic insects in the sediment at the impacted marsh, representing between 70 to 90% of aquatic insects (Figure 9B). Ephemeroptera comprised 20% or more of insects collected from the reference marsh sediments on every sampling date, but were completely absent from impacted marsh sediments in July, August and September and were only present in low numbers in June (Figure 9B).

Chironomidae abundance trends were comparable for the sediment and plant associated communities in the reference and impacted marshes with no consistent trends in abundance and no significant differences ($p<0.05$) occurring between the two marshes (Figures 10 and 11). Thus, differences in percent composition of insect communities for the two sites (Figure 9) were the result of reductions in Ephemeroptera, Trichoptera and other

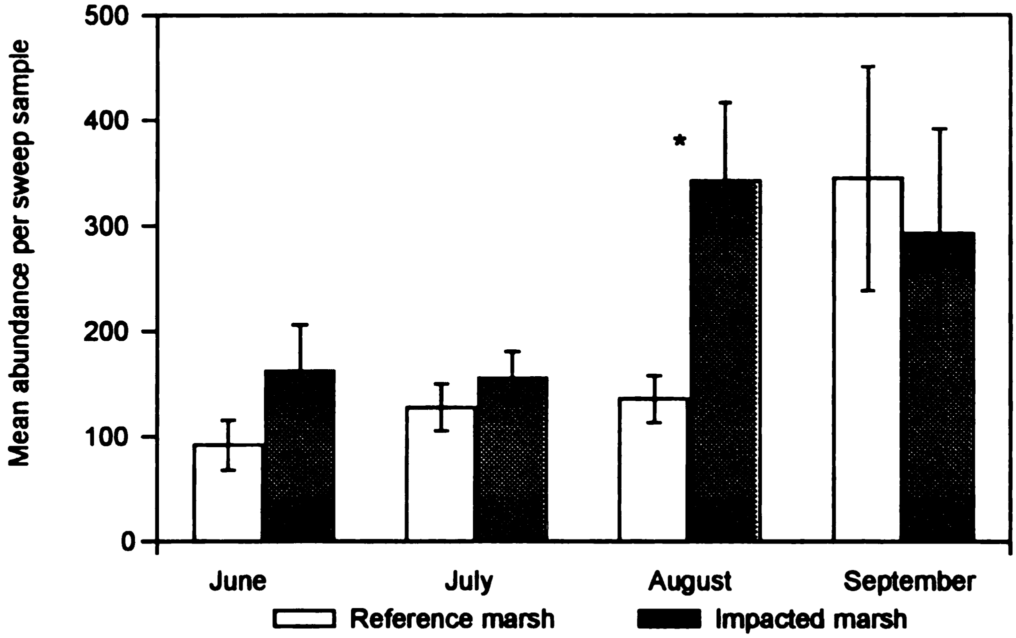

Figure 10. Abundance trends, including standard error, of the Chironomidae in the plant associated community (dip-net samples) of Cedarville (impacted) and Mackinac (reference) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

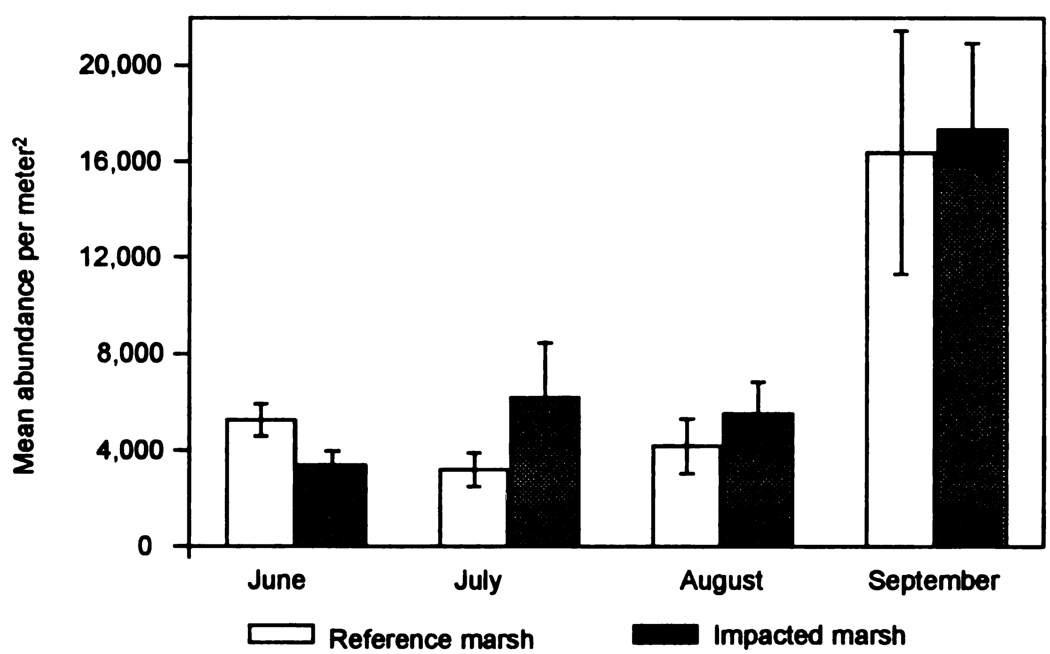

Figure 11. Abundance trends, including standard error, of the Chironomidae in the sediment community (core samples) of Cedarville (impacted) and Mackinac (reference) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

insects rather than from reductions in Chironomidae abundances (Figures 10 and 11).

Chironomidae larvae were often found inside the stems of senescent *S. acutus*, *Utricularia spp.*, and *Nuphar variegata*, with densities inside the plants increasing as decomposition progressed towards late summer. On actively growing *Nuphar variegata*, Chironomidae larvae were found on the outside of the stems or underneath the leaves, with the periphyton community. Some of these larvae were attached to the stems and leaves by fixed cases, while others were free living. Chironomidae larvae were seldom found on actively growing *S. acutus*. Free living Chironomidae included *Corynoneura spp.*, Tanypodinae, and several of the Chironomini and Tanytarsini taxa. Most of the other Orthocladiinae and some of the Chironomini and Tanytarsini, constructed fixed cases.

Even though there were no consistent, significant ($p < 0.05$) differences between total abundance of Chironomidae larvae in the two marshes (Figures 10 and 11), the taxonomic composition of the Chironomidae communities did differ between the two marshes (Tables 3 and 4). Chironomini was consistently the most abundant Chironomidae at the impacted marsh in both the plant and sediment community (Tables 3 and 4). The reference marsh sediment was also dominated by Chironomini, however the plant associated Chironomini was only the dominant Chironomidae in June and September at the reference marsh (Tables 3 and 4). During July, the most abundant Chironomidae in the plant associated community of the reference marsh was the Tanytarsini comprising 40% of that population (Table 3). The other Orthocladiinae dominated the Chironomidae community in August at the reference marsh making up 50% of the Chironomidae community (Table 3). The free-living *Corynoneura spp.* never represented over 5% of the Chironomidae community in the plant associated

Table 3. Total number of Chironomidae collected and percent of the total Chironomidae represented by taxa in the plant associated community in Mackinac (reference) marsh, Lake Huron, 1996.

Taxa	Date			
	June	July	August	September
Chironomini	55 %	30 %	29 %	50 %
Orthocladiinae				
Corynoneura spp.	3 %	5 %	3 %	1 %
others	6 %	13 %	49 %	4 %
Tanypodinae	9 %	12 %	14 %	33 %
Tanytarsini	27 %	40 %	5 %	12%
Total number	**828**	**1,148**	**1,219**	**3,103**

Table 4. Total number of Chironomidae collected and percent of the total Chironomidae represented by taxa in the plant associated community in Cedarville (impacted) marsh, Lake Huron, 1996.

Taxa	Date			
	June	July	August	September
Chironomini	42 %	47%	59 %	44 %
Orthocladiinae				
Corynoneura spp.	21%	6 %	4 %	6 %
others	22%	35 %	12 %	30 %
Tanypodinae	6 %	9 %	8 %	9 %
Tanytarsini	9 %	3 %	17 %	11%
Total number	**1,457**	**199**	**3,076**	**2,634**

community of the reference marsh, but was frequently collected, with an average of four individuals per sample (Table 3). *Corynoneura spp.* comprised 21% of the Chrionomidae community during June at the impacted site and then fell to levels similar to those exhibited at the reference marsh of about 5% or less of the community composition (Table 4). Chironomidae reached a peak mean abundance of 341 individuals per sample in September in the plant associated community of the reference marsh, and a peak abundance of 341 individuals per sample in August at the impacted marsh (Figure 10). The Chironomidae were identified to the lowest taxonomic level possible in June (Table A-2). On this date, 28 taxa were identified. The reference marsh had 19 identifiable taxa compared with 13 at the impacted marsh (Table A-2). Only one Tanypodinae, *Ablabemyia peleenis*, was present in the impacted marsh samples, compared with five species collected at the reference marsh (Table A-2).

Chironomidae was the dominant group of aquatic insects in the sediment community, comprising over 50% of the aquatic insects at the reference marsh, and over 90% at the impacted marsh (Figure 9B). The sediment associated Chironomidae reached a maximum mean density in both marshes in September, with no significant ($p<0.05$) difference between the mean density of $17{,}331 \cdot m^{-2}$ at the impacted marsh, and the $16{,}000 \cdot m^{-2}$ at the reference marsh (Figure 11). Larvae of the tribe Chironomini was the dominant Chironomidae in the sediment community of both the reference and impacted marsh (Tables 5 and 6). The predacious, Tanypodinae was also a major constituent of the Chironomidae community of the reference marsh, representing nearly 40% of the Chrionomidae in September (Table 5). Tanypodinae in the sediment samples of the impacted marsh had similar abundances to the

Table 5. Total number of Chironomidae collected per m^2 and percent of the total Chironomidae represented by taxa in the sediment community in Mackinac (reference) marsh, Lake Huron, 1996.

Taxa	Date			
	June	July	August	September
Chironomini	62 %	54 %	33 %	57 %
Orthocladiinae				
Corynoneura spp.	0 %	7 %	3 %	0 %
others	6 %	12 %	21 %	1 %
Tanypodinae	11 %	27 %	24 %	39 %
Tanytarsini	21 %	0 %	19 %	3 %
Total number	**50,382/m^2**	**32,569/m^2**	**47,582/m^2**	**148,601/m^2**

Table 6. Total number of Chironomidae collected per m^2 and percent of the total Chironomidae represented by taxa in the sediment community in Cedarville (impacted) marsh, Lake Huron, 1996.

Taxa	Data			
	June	July	August	September
Chironomini	60 %	63 %	66 %	75 %
Orthocladiinae				
Corynoneura spp.	3 %	1 %	3 %	0 %
others	22 %	3 %	11%	1 %
Tanypodinae	12 %	29 %	2 %	17 %
Tanytarsini	3 %	4 %	18 %	7 %
Total number	**30,535/m^2**	**55,726/m^2**	**49,618/m^2**	**155,980/m^2**

reference marsh in June and July (Tables 5 and 6). However, abundance of Tanypodinae dropped at the impacted marsh from 29% of Chironomidae in July, to 2% of the Chironomidae population in August, and remained at abundances much less than exhibited at the reference marsh in September (Tables 5 and 6). The Orthocladiinae, *Corynoneura spp.*, was the least abundant Chironomidae larvae in the sediment samples at both the reference and impacted marshes, representing less than 10% of the total Chironomidae (Tables 5 and 6).

The only other dipteran families present in the plant associated and sediment samples of the marshes were uncommon to rare especially in the plant associated samples (Figure 9 and Table 2). Empididae and Ceratopogonidae were collected in the sediment community of the reference marsh (Table 2). The Ceratopogonidae was represented by three genera; *Bezzia/Palpomyia spp.*, *Serromyia spp.*, and *Sphaermias spp.* in the sediment community of the reference marsh. Ceratopogonidae occurred in much larger abundances in the sediment community of the reference marsh compared with the plant associated community (Table 2). The most abundant of these species was *Bezzia/Palpomyia spp.* which had an average density of 148 larvae$\cdot$ m^{-2} in the sediment community of the reference marsh and 21 larvae$\cdot$m^{-2} in the sediment of the impacted marsh (Table 2). The only Diptera, besides Chironomidae, collected in the sediment of the impacted marsh was *Bezzia/Palpomyia spp.* and *Culicoides spp.* members of the family Ceratopogonidae, and *Chrysops spp.* (Tabanidae) (Table 2).

There were 7 species of mayflies in four families collected from the reference marsh, but only 4 species from 3 families were collected from the impacted marsh (Tables 7 and 8). Ephemeroptera was a major component of the sediment and plant associated macroinvertebrate communities of the reference marsh, comprising from 10 to more than 30%

Table 7. Total number of Ephemeroptera collected and percent of the total Ephemeroptera represented by species in the plant associated community in Mackinac (reference) marsh, Lake Huron, 1996.

Taxa	Date			
	June	July	August	September
Baetidae				
Callibaetis ferrugineus	10 %	7 %	4 %	7 %
Procleon viridocularis	3 %	6 %	0 %	0 %
Caenidae				
Caenis amica	30 %	13 %	6 %	58 %
Caenis latipennis	21 %	14 %	8 %	17 %
Caenis youngi	35 %	60 %	71 %	16 %
Ephemerellidae				
Eurylohella funeralis	<1 %	0 %	4 %	2 %
Ephemeridae				
Hexagenia limbata	0 %	0 %	7 %	0 %
Total number/dip-net sample	**15/sample**	**21/sample**	**35/sample**	**155/sample**

Table 8. Total number of Ephemeroptera collected per m^2 and percent of the total Ephemeroptera represented by species in the sediment of Mackinac (reference) marsh, Lake Huron 1996.

Taxa	Date			
	June	July	August	September
Baetidae				
Callibaetis ferrugineus	4 %	12 %	0 %	3 %
Caenidae				
Caenis amica	11 %	6 %	5 %	47 %
Caenis latipennis	4 %	18 %	0 %	29 %
Caenis youngi	81 %	64 %	75 %	20 %
Ephemerellidae				
Eurylohella funeralis	0 %	0 %	0 %	<1 %
Ephemeridae				
Hexagenia limbata	0 %	0 %	20 %	<1 %
Total number	**20,610/m^2**	**12,723/m^2**	**34,097/m^2**	**84,988/m^2**

percent of the aquatic insect communities (Figure 9). In contrast, mayflies were only minor components of the impacted marsh fauna and never represented more than 4% of the plant or sediment associated aquatic insect communities (Figure 9). Ephemeroptera abundance was significantly higher ($p<0.05$) in the sediment and plant associated communities at the reference marsh than was Ephemeroptera abundance in the impacted marsh throughout the sampling season (Figures 12 and 13). Ephemeroptera in the plant and sediment associated communities reached peak abundances in September with numbers generally increasing as the season progressed from June through September (Figures 12 and 13). Ephemeroptera increased from 15 per dip-net sample in June to their peak density of 155 per dip-net sample in the plant associated community in September. Density also increased in the sediment associated community at the reference marsh to 84,988$\cdot m^{-2}$ in September from a low of 12,723$\cdot m^{-2}$ in July (Tables 7 and 8). In contrast, no Ephemeroptera were collected in July, August, and September in the sediment of the impacted marsh. Ephemeroptera reached a mean peak abundance of only 4 individuals per sample in the plant associated community of the impacted marsh during July (Figure 12). The predominant Ephemeroptera family in the plant and sediment associated communities of the reference marsh was the family Caenidae with 3 species present comprising from 80-96% of total mayfly numbers collected (Tables 7 and 8). An average of 51 Caenidae per dip-net sample was collected over the entire season from the reference site as compared to 0.25 Caenidae per sample for the impacted site (Table 2). Likewise, a seasonal average of 3,887 Caenidae$\cdot m^{-2}$ were collected from sediments of the reference site as compared to 42 for the impacted site.

The taxonomic composition between the sediment and the plant associated community

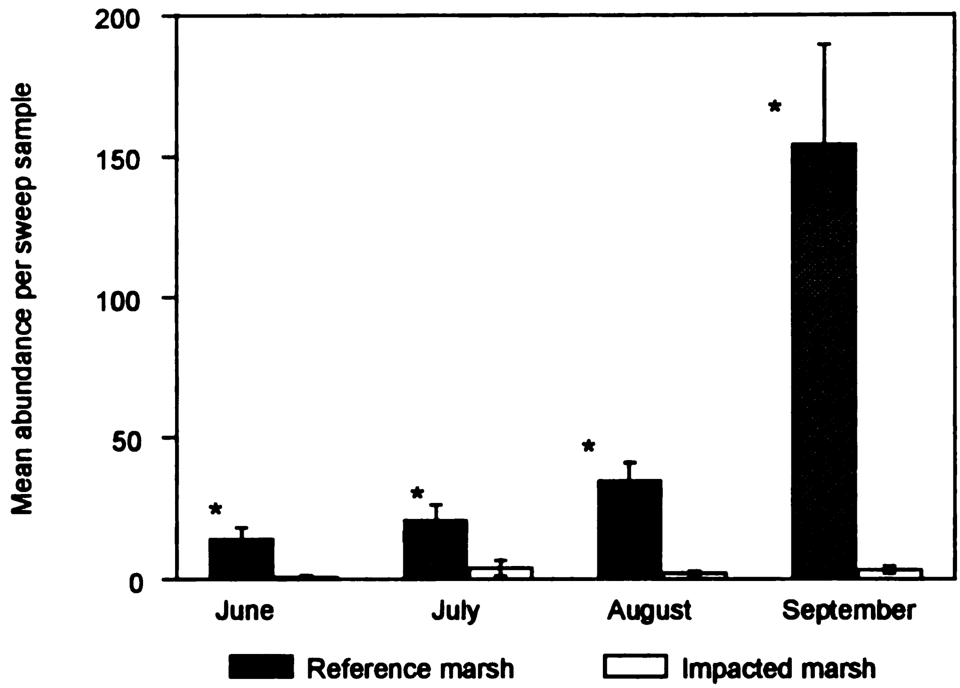

Figure 12. Abundance trends, including standard error, of Ephemeroptera in the plant associated community of Mackinac (reference) and Cedarville (impacted) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

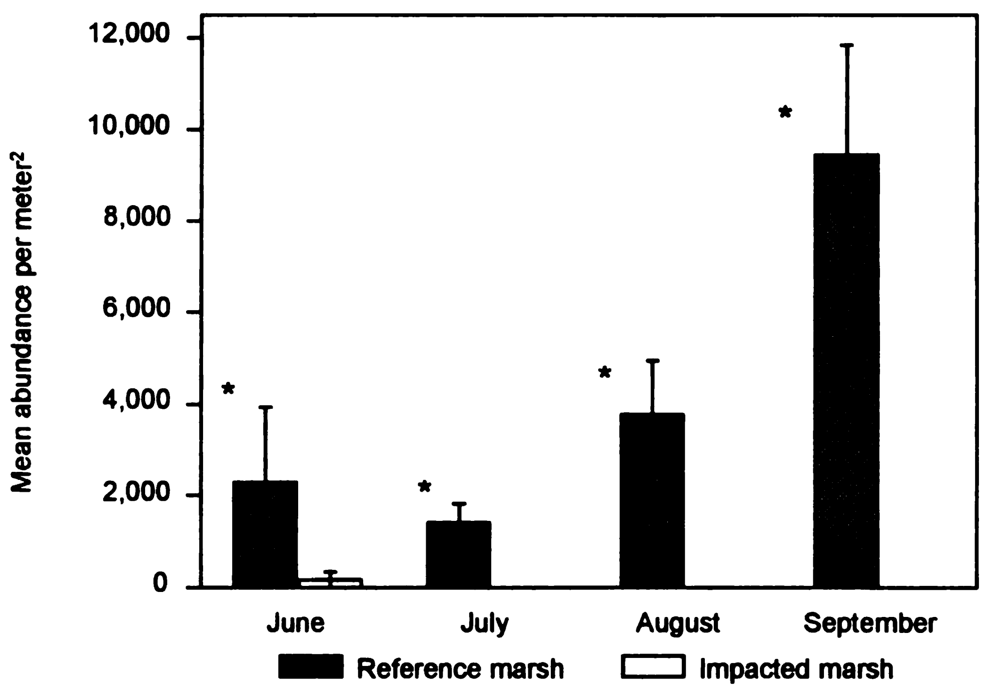

Figure 13. Abundance trends, including standard error, of Ephemeroptera in the sediment associated community of Mackinac (reference) and Cedarville (impacted) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

of the reference marsh differed only in the occurrence of one additional Baetidae species in the plant associated community, *Procleon viridocularis*. The remainder of the plant and sediment associated taxa in the reference marsh communities were similar with three species of Caenidae, *Caenis amica*, *C. latipennis*, and *C. youngi*, and one species each in the family Ephemerellidae, Ephemeridae and Baetidae (Table 2). The most commonly observed specimens in the plant and sediment associated communities of the reference marsh were *Caenis amica* and *C. youngi* (Tables 7 and 8). *C. youngi* was the dominant Ephemeroptera species in the sediment and plant associated communities of the reference marsh June through August but was never collected from the impacted marsh (Table 2). *C. amica* was the most abundant Ephemeroptera in September in both the sediments and plant associated samples (Tables 7 and 8). The burrowing mayfly *Hexagenia limbata* was only collected in the plant associated community in August and was collected in both August and September in the sediment associated community of the reference marsh (Tables 7 and 8). However, no *H. limbata* were collected in either the sediment or the plant associated communities of the impacted marsh (Table 2).

The only Ephemeroptera species collected in the plant community of the impacted marsh were two species of Caenidae, *C. amica* and *C. latipennis,* and two species of Baetidae, *Callibaetis ferrugineus* and *Eurylophella funeralis* (Table 2). *C. ferrugineus* was the most abundant Ephemeroptera at the impacted marsh, yet only an average of 2 individuals per sample were collected (Table 2). Only one species of Ephemeroptera was collected in the sediment samples at the impacted marsh, *C. amica,* and it was only present in June (Table 2).

The Trichoptera was the most diverse group of insects in the plant and sediment associated communities in both marshes with a total of 25 species in 7 families collected from the two marshes (Table 2). The Trichoptera in the plant associated community of the reference marsh was represented by a total of 19 species; two species each in the families Molannidae and Hydropsychidae; one species each in the families Hydroptilidae, Limnephilinae and Polycentropodidae; three species of Phryganeinae and nine species of Leptoceridae (Table 2). Sixteen species representing 6 families of Trichoptera were collected from the plant associated community of the impacted marsh with the family Molannidae being the only family present in the reference marsh that was not also represented in the impacted marsh (Table 2). Ten species of Trichoptera were present in the sediment core samples of the reference marsh; three species of Hydroptilidae, four species of Leptoceridae and one species each of Molannidae, Phryganeidae and Polycentropodinae (Table 2). Six species of Trichoptera were present in the sediment community of the impacted marsh, but only *Polycentropus spp.* and *Agraylea multipuntata* occurred in more than one month of sampling (Table 2).

Two weeks following discharge in June and in September during discharge from the wastewater treatment lagoon, the plant associated Trichoptera in the impacted marsh were significantly ($P<0.05$) less abundant than Trichoptera at the reference marsh (Figure 14). Although Trichoptera were consistently more abundant in the sediment community of the impacted marsh, the only month a significant difference ($P<0.05$) occurred was September when discharge was actively occurring from the wastewater treatment lagoon (Figure 15). Trichoptera of both the plant and sediment associated communities reached peak abundances in September at the reference marsh (Figures 14 and 15). Trichoptera reached a maximum

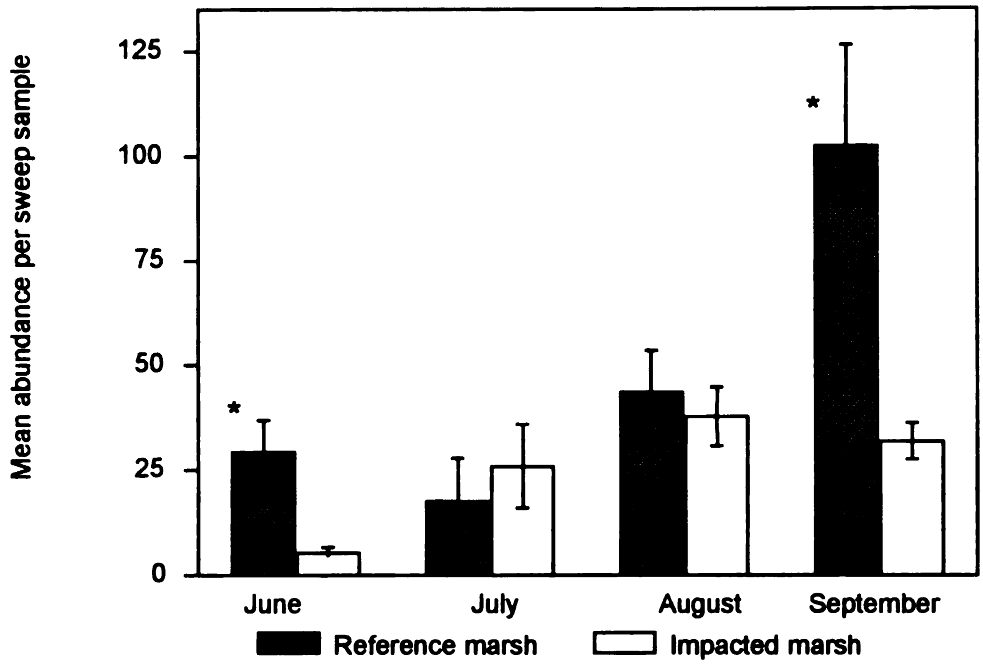

Figure 14. Abundance trends, including standard error, of Trichoptera in the plant associated community of Mackinac (reference) and Cedarville (impacted) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

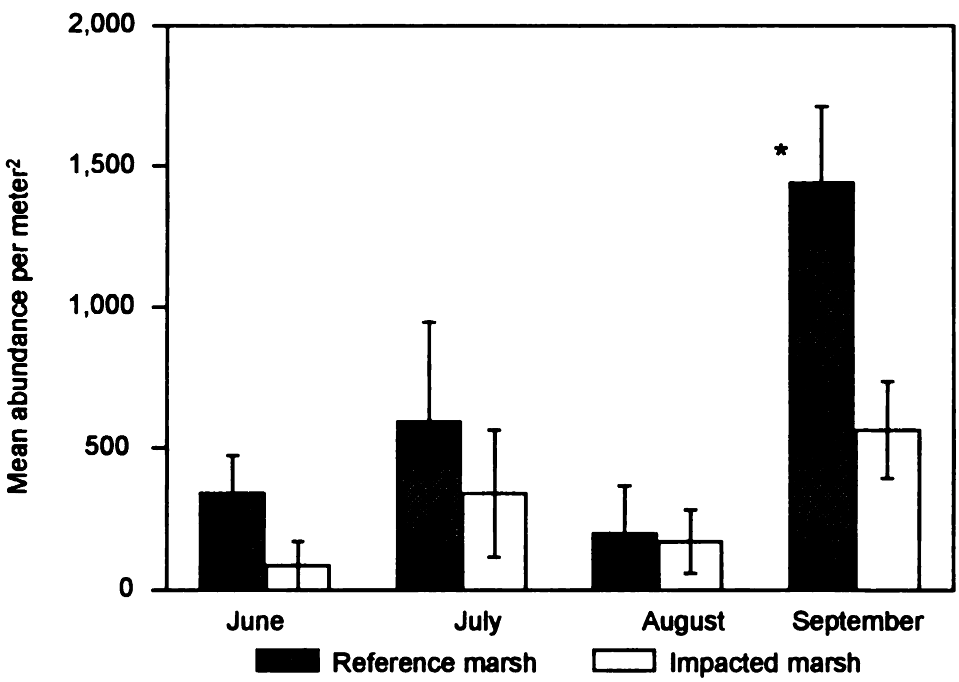

Figure 15. Abundance trends, including standard error, of Trichoptera in the sediment associated community of Mackinac (reference) and Cedarville (impacted) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

mean abundance of 102 individuals in the plant associated community, and a maximum mean density of 1,441$\cdot$m^{-2} in the sediment of the reference marsh (Figures 14 and 15). Trichoptera also reached a peak mean abundance in September in the sediment community of the impacted marsh, however only at 565 individuals$\cdot$m^{-2} (Figure 15). Plant associated Trichoptera at the impacted marsh reached a peak mean abundance a month earlier in August, with a mean abundance of only 31 individuals per sample (Figure 14).

Few similarities existed in the dominant taxa between marshes and between sampling regimes (Table 2). *Phryganea cinera, Oxyethira spp.*, *Oecetis spp.*, and *Mystacides sepulchralis* were the most abundant Trichoptera species in the plant associated community at the reference marsh and were also among the most common species collected from sediment samples (Table 2). Although *P. cinera* represented the majority of the plant associated Trichoptera population in July and September, it was not collected in June and was scarce in August sampling (Figure 16). During wastewater discharge in September, *P. cinera* were significantly ($p<0.05$) more abundant in the plant associated samples at the reference marsh than at the impacted marsh (Figure 16). The sediment community of the reference marsh was not consistently dominated by any one species of Trichoptera (Figure 17). During June, only two species were collected in the sediment community of the reference marsh, *Mystacides interjecta* and *Oecetis osteni*, with these species occurring in equal proportions (Table 2). During July, the Polycentropodidae, *Polycentropus spp.,* was the dominant Trichoptera species in the sediment community of the reference marsh reaching a peak mean density at approximately 400$\cdot$m^{-2} (Figure 17).

The most common Trichoptera in the plant associated community of the impacted

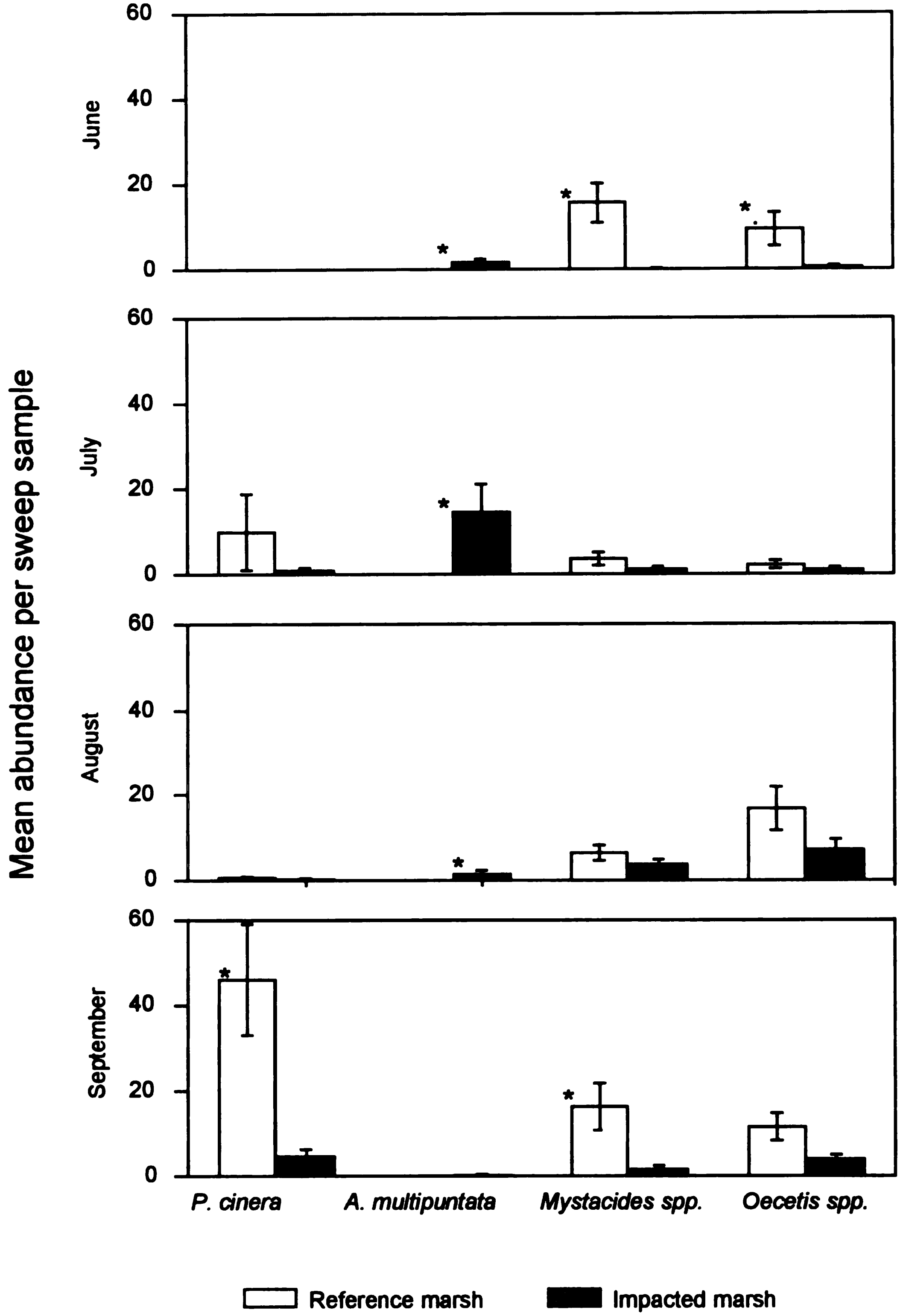

Figure 16. Abundance trends including standard error of the dominant Trichoptera species in the plant associated community of Cedarville (impacted) and Mackinac (reference) marshes, Lake Huron 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

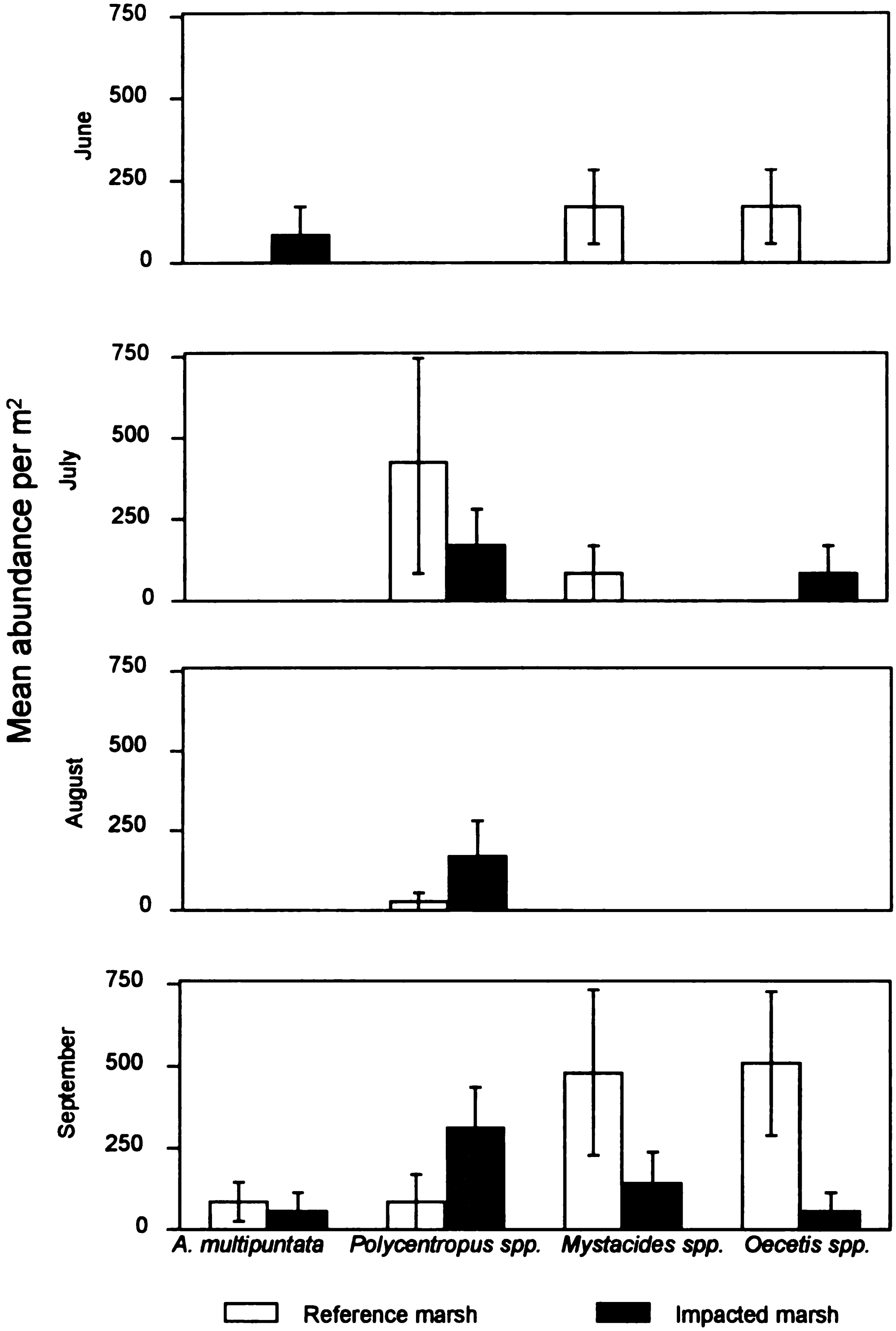

Figure 17. Abundance trends including standard error of the dominant Trichoptera species in the sediment associated community of Cedarville (impacted) and Mackinac (reference) marshes, Lake Huron 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

marsh were members of the Hydroptilidae family, but a large number of other species in several families were also present (Table 2). *Agraylea multipunctata* was the dominant Trichoptera during June and July, and was significantly ($p<0.05$) more abundant in those months at the impacted marsh than compared with the reference marsh (Figure 16). Yet during September, *A. multipunctata* was uncommon in the plant community of the impacted marsh (Figure 16). The most abundant Trichoptera in the plant associated community of the impacted marsh in August was *Oxyethira spp.* *Oxyethira spp.* comprised nearly 50% of the Trichoptera community in the plant associated samples during August and dropped to comprise less than 10% in September at the impacted marsh. September plant associated samples at the impacted marsh were dominated by *Hydroptila spp.* which represented nearly 50% of the Trichoptera population that month (Table 2). *Hydroptila spp.* had not been collected in previous samples all summer (Table 2). The remainder of the Trichoptera community in the plant associated community of the impacted marsh tended to be uncommon, seldom representing more than 5% of the Trichoptera community composition (Table 2). Trichoptera occurred in very low numbers in the sediment of the impacted marsh (Table 2). *Polycentropus spp.* was the most abundant Trichoptera in the sediment of the impacted marsh averaging $162 \cdot m^{-2}$ (Table 2). The remainder of the sediment associated Trichoptera at the impacted marsh, *Mystacides sepulchralis*, *Oecetis cinerascens*, *Oecetis spp.*, and *Phryganea cinera*, occurred in very low abundances (Table 2). No significant difference were detected between the dominant Trichoptera species between the sediment communities of the impacted and reference marsh (Figure 17).

Odonata represented approximately 15% of the plant associated macroinvertebrates in both marshes with the majority being Coenagrionidae damselflies (Table 2). Odonata abundance trends were comparable for the sediment and plant associated communities in the reference and impacted marshes no significant ($p<0.05$) differences occurred for any sampling date (Figures 18 and 19). Peak abundances occurred in September with numbers generally increasing as the season progressed from June through September (Figures 18 and 19). Odonata were uncommon in the sediment samples of both marshes (Table 2). Only two species of Odonata, *Ischnura verticalis* and *Enallagma spp.* were collected in the sediment samples of these marshes (Table 2). *Enallagma spp.* was lacking the characteristics necessary to identify them to species, either due to the collection of early instars or physically damaged specimens.

Plant associated Odonata reached their highest abundance in September with 813 individuals collected at the reference marsh, and 548 individuals collected at the impacted marsh (Tables 9 and 10). The Odonata associated with the plant community of the reference marsh was represented by 13 taxa compared to 10 taxa for the impacted marsh (Table 2). The Coenagrionidae, *Ischnura verticalis,* was the most abundant Odonata in the plant associated community of the both the reference and impacted marshes, representing up to 80% of the Odonata in the reference marsh and up to 89% in the impacted marsh (Table 9). Also present but generally uncommon in the plant associated community at the reference marsh were five other Coenagrionidae species, one Lestidae species, and six Anisoptera species (Table 2). While occurrence of uncommon species differed between the reference and impacted marshes, no significant differences ($p<0.05$)occurred for any of the common Odonate taxa (Table 2). Sediment associated Odonata were dominated by *Enallagma spp.*,

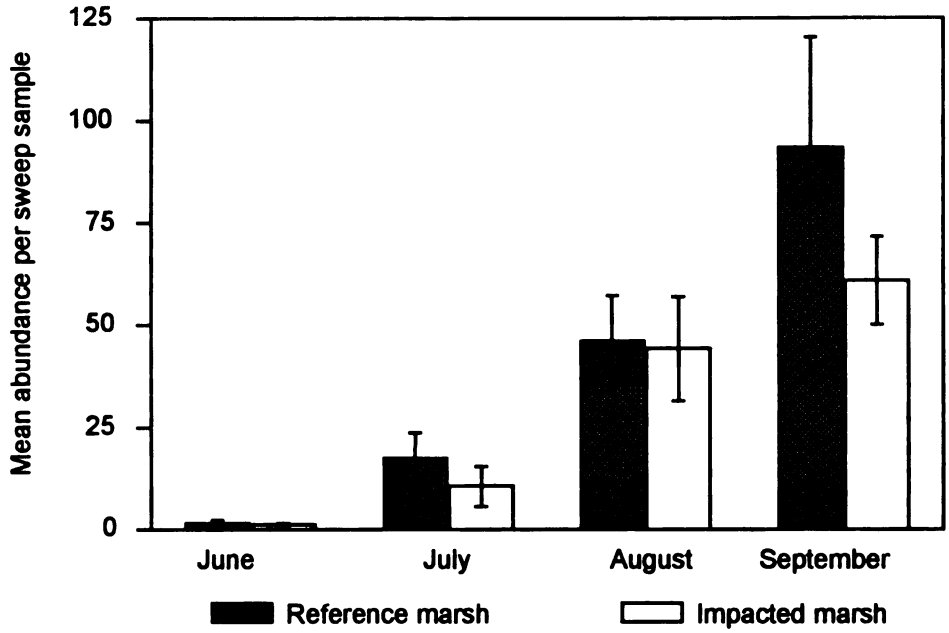

Figure 18. Abundance trends, including standard error, of Odonata in the plant associated community of Mackinac (reference) and Cedarville (impacted) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

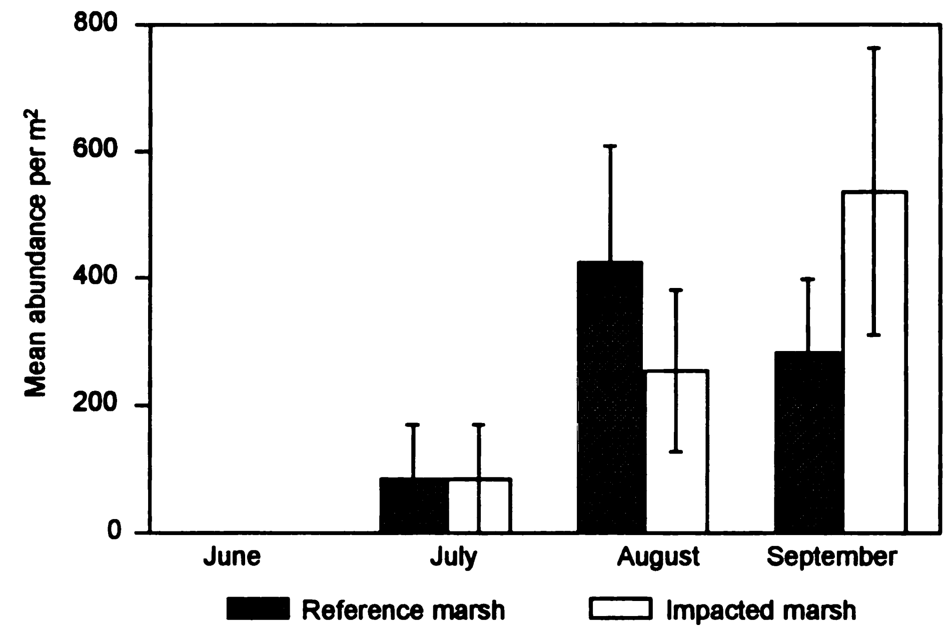

Figure 19. Abundance trends, including standard error, of Odonata in the sediment associated community of Mackinac (reference) and Cedarville (impacted) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

Table 9. Total number of Odonata and percent of the total Odonata represented by taxa in the plant associated community in Mackinac (reference) marsh, Lake Huron, 1996.

Taxa	June	July	August	September
Aeshnidae				
Anax junius	0 %	0 %	0 %	<1 %
Coenagrionidae				
Enallagama boreale	0 %	0 %	<1 %	1 %
Enallagama ebrium/hageni	31%	<1 %	<1 %	18 %
Enallagma geminatem	8 %	0 %	0 %	<1 %
Enallagma vernale	0 %	0 %	1 %	1%
Ischnura verticalis	46 %	80 %	78 %	78 %
Others	0 %	10 %	11 %	0 %
Corduliidae				
Epitheca spp.	0 %	0 %	1 %	1 %
Cordulia shurleffi	0 %	6 %	7 %	0 %
Gomphidae				
Arigomphus cornutus	0 %	<1 %	0 %	0 %
Libellulidae				
Celithemis spp.	0 %	0 %	<1 %	0 %
Others	0 %	4 %	0 %	0 %
Lestidae				
Lestes spp.	15 %	0 %	0 %	0 %
Total number	**13**	**151**	**413**	**813**

Table 10. Total number of Odonata and percent of the total Odonata represent by taxa in the plant associated community of Cedarville (impacted) marsh, Lake Huron, 1996.

Taxa	June	July	August	September
Aeshnidae				
Aeshna eremita	0 %	13 %	<1 %	1%
Coenagrionidae				
Enallagama ebrium/hageni	50 %	7 %	4 %	37 %
Enallagma geminatem	20 %	0 %	<1%	<1 %
Ischnura verticalis	10 %	0 %	89 %	62 %
Others	0 %	64 %	0 %	0 %
Corduliidae				
Epitheca spp.	0 %	0 %	3 %	<1%
Cordulia shurleffi	10 %	8 %	0 %	0 %
Libellulidae	0 %	8 %	3 %	0 %
Lestidae				
Lestes spp.	10 %	0 %	0 %	0 %
Total number	**10**	**95**	**397**	**548**

which was collected with a mean density of 177 nymphs·m^{-2} at the reference marsh and 205 nymphs·m^{-2} at the impacted marsh (Table 2). *I. verticalis* was the only other species collected from the sediment samples in either the impacted or the reference marsh (Table 2), and it only occurred during September. No Odonata were collected in June sediment samples. No significant differences ($p<0.05$) occurred between the impacted and reference marshes for either species of sediment occurring Odonata (Table 2).

I. verticalis was collected every month in the plant associated samples of the impacted marsh with the exception of July, where on this date 64% of the Odonata population was represented by an early instar Coenagrionidae that were lacking characteristics necessary for proper identification to the generic level (Table 10). The unidentified Coenagrionidae dominated the Odonata plant associated community at the impacted marsh during July (Table 2). It appeared that *I. verticalis* hatched sometime between the June and July sampling date, and the unidentified Coenagrionidae may have been early instar *I. verticalis*. If so *I. verticalis* abundance in the impacted marsh would have been similar to abundance in the reference marsh on this date as it was in August and September (Tables 9 and 10). Too few Odonata were collected in June from either site for inter-site comparisons to be meaningful (Tables 9 and 10).

The remainder of the aquatic insect community in the plant and sediment associated samples occurred in very low numbers and infrequently in the reference and impacted marsh (Table 2). Aquatic Coleoptera were represented by nine taxa at the reference marsh, and six taxa at the impacted marsh (Table 2). Sediment associated aquatic Coleoptera was represented by just one larval species, *Donacia spp.*, and it occurred in both the impacted and reference marsh (Table 2).

Hemiptera was represented by nine species belonging to six families in the plant associated community of the reference marsh and 8 species belonging to four families in the impacted marsh (Table 2). The most abundant Hemipteran family, the Corixidae, included three species and a group of nymphs lacking the characteristics necessary to identify them below the family level in the reference marsh and four species in the impacted marsh (Table 2). *Sigara transfigurata* and *Sigara variablilis* were two Corixidae collected in the plant associated samples of the reference marsh that had not previously been recorded from this area. Sediment associated aquatic Hemiptera at the reference marsh consisted of the family Corixidae and the Belostomatidae, *Lethocerus spp.* (Table 2). The sediment associated Hemiptera of the impacted marsh was represented by unidentified Corixidae nymphs, and *Mesovelia mulsanti* (Table 2).

The same three species of Lepidoptera were collected in the plant associated samples at both the impacted and reference marsh in low numbers; *Bellura spp.; Acentria spp.*; and *Parapoynx spp.* (Table 2). Sediment samples from both marshes contained low numbers of Lepidoptera larvae (Table 2).

Three species of Collembola, *Isotomurus tricolor*, *Entomobrya nivalis* and *Pseudobourletiella spinata* were occasionally collected in the plant associated samples at the impacted marsh (Table 2). Only one species of Collembola, *Pseudobourletiella spinata,* was collected from the plant associated community of the reference marsh (Table 2). No Collembola were collected in the sediment samples of either marsh.

The least diverse aquatic insect group in the plant associated community of the impacted marsh was a sponge predator Neuroptera, *Sisyra spp.* (Table 2). *Sisyra spp.* was uncommon in the plant associated samples at the impacted marsh, although many sponges were observed along the transect (Table 2).

Non-Insect Macroinvertebrates:

A total of 8,619 non-insect macroinvertebrates were collected in the plant associated community of the reference marsh, distributed among 23 identified taxa (Table 11). There were 10,985 non-insect invertebrates in 30 taxa collected from the impacted marsh. Thus, the impacted marsh supported more non-insect taxa than did the reference marsh (Table 11). The sediment community at the reference marsh was represented by 17 identified taxa, only one of which, a Lymnaeid snail, *Acella haldemani,* was not collected in the plant associated community of the reference marsh (Table 11).

The sediment community of the impacted marsh consisted of 18 taxa with only two taxa present that were not also collected from sediments of the reference marsh (Table 11). The Oligochaeta, Amphipoda, Gastropoda and Isopoda were significant components of the plant and sediment associated non-insect macroinvertebrate communities in both the reference and impacted marshes.

Non-insect macroinvertebrates made up more than 50% of the invertebrate community in both the sediments and plant associated communities of the impacted marsh, while aquatic insects made up more than 50% of the macroinvertebrate communities in the reference marsh (Figure 7).

Nematoda were over 5 times more common in the sediments of the impacted marsh than they were in sediments of the reference marsh (Table 11). Nematoda dominated the

Table 11. Non-insect macroinvertebrates collected with standardized dip-net sweep sampling from *Scirpus* dominated zones (mean number/sample±S.E) and with cores from sediments in these zones (mean number·m^{-2} ±S.E) from the reference (Mackinac) and Impacted (Cedarville) marshes in 1996. * was not collected in the samples.

TAXON	Mackinac		Cedarville	
	Dip-Net	Core samples	Dip-Net	Core samples
	Mean #/sample±S.E.	Mean #·m^{-2}±S.E.	Mean #/sample±S.E.	Mean #·m^{-2}±S.E.
ANNELIDA				
OLIGOCHAETA				
Naididae				
Stylaria spp.	67.55±23.24	353.41±144.85	65.58±30.55	551.32±467.78
Others	19.53±8.51	982.46±265.56	59.36±6.03	1547.92±744.13
Tubificidae	1.58±.86	453.61±233.95	5.17±2.98	374.61±128.20
POLYCHAETA				
Manayunkia speciosa	*	*	.08±.08	*
HIRUDINAE	*	*	*	*
Erpobdellidae	.36±.36	*	*	*
Mooreobdella spp.	*	*	.11±.08	*
Glossiphoniidae				
Alboglossiphonia heteroclita	*	*	.31±.21	*
Batrachobdella phalera	*	*	.22±.22	*
Helobdella stagnalis	*	*	.11±.04	*
Theromyzon spp.	.03±.03	*	*	*
CRUSTACEA				
AMPHIPODA				
Crangnox spp.	*	*	.14±.14	*
Gammarus spp.	3.06±.71	247.39±45.26	136.45±51.93	1936.67±799.04
Hyallella azteca	36.78±14.51	883.52±212.63	72.22±27.20	1448.97±542.41
DECAPODA				
Orconectes propinquus	*	*	.03±.03	*
ISOPODA				
Lirceus lineatus	11.64±10.01	155.50±106.73	43.56±12.41	720.98±230.99
Racovitzai racovitzai	8.64±2.5	911.79±395.38	12.17±2.74	1533.79±316.17
MOLLUSCA				
GASTROPODA				
Pulmonata				
Ancylidae				
Ferrissa parallela	.22±.10	*	10.28±7.89	254.45±227.06
Lymnaeidae				
Acella haldemani	.50±.30	42.41±42.41	.83±.83	63.62±40.6
Fossaria spp.	.17±.17	*		
Physidae				
Aplexa elongata	*	*	.08±.05	*
Physa gyrina	5.06±2.56	63.62±40.6	6.98±2.46	134.30±107.89
Planorbidae				
Gyraulus deflectus	2.45±2.41	*	.55±.38	*
Gyralus parvus	40.97±35.05	445.29±111.53	41.61±19.55	1957.87±523.52
Promenetus exacuous	1.08±.97	42.41±42.41	6.03±2.97	269.86±127.23

Table 11 (cont'd)

TAXON	Mackinac		Cedarville	
	Dip-Net	Core samples	Dip-Net	Core samples
	Mean #/sample±S.E.	Mean #·m^{-2}±S.E.	Mean #/sample±S.E.	Mean #·m^{-2}±S.E.
Prosobranchia				
Bithyniidae				
Bithynia tentaculata	*	*	.55±.34	56.54±32.24
Hydrobiidae				
Amnicola limosa	7.72±6.43	183.78±46.88	23.11±8.61	728.02±213.57
Valvata bicarinata	*	*	.22±16	*
OTHERS				
NEMATODA	8.92±2.07	586.66±291.85	2.59±.55	3011.03±871.53
PELECYPODA				
Sphaeriidae	5.06±2.02	523.05±131.35	1.97±1.46	77.75±60.39
ARACHNOIDEA (MITES)				
Oribatei	4.72±1.95	424.09±94.48	3.22±.69	1095.56±359.37
Hydrocarina	13.11±2.48	184.30±29.14	10.42±4.6	346.34±71.5
Spider				
Tetragnatha laboriosa	.11±.22	*	*	*
TUBELLARIA				
Tricladida	.14±.10	21.21±21.21	.50±.20	*
Dugesia tigrina	*	*	1.36±1.25	*

sediment community of the impacted marsh during June, representing 30% of the macroinvertebrate community, and was a significant component of this community in July and August. Nematoda were a less significant component of the reference marsh sediment community (Figure 7B and Table 11).

Oligochaeta was represented by two families, the Naididae and the Tubificidae (Table 11). There were no significant differences ($p<0.05$) in Oligochaeta abundance in the two marshes on any sampling date (Figures 20 and 21). Most of the Oligochaeta collected from the reference and the impacted marsh were members of the family Naididae (Tables 11-15). The genus *Stylaria spp.* was easily identified and was separated from other unidentified Naididae. *Stylaria spp.* and other Naididae were abundant in the plant associated community in both marshes, comprising from 88 to almost 100% of the Oligochaeta for all sampling dates (Tables 11-13). Tubificidae made up more than 5% of the plant associated Oligochaeta only during August in the impacted marsh (Table 13). Oligochaeta reached a peak mean density in the plant associated community in July at both marshes (Tables 12 and 13, Figure 20). While abundance in the sediments were more variable with peaks in June and September (Tables 14 and 15, Figure 21). During July's peak density for the plant associated community, 1,286 Oligochaeta were collected, at the reference marsh, compared with 2,105 at the impacted marsh (Tables 12 and 13). *Stylaria* numbers were comparable for both marshes (Tables 12 and 13). Thus, differences in overall numbers in July and August occurred through increases in other species of Naididae in the impacted marsh (Tables 12 and 13).

Sediment associated Oligochaeta abundance at the impacted marsh, was higher in September during discharge from the wastewater treatment lagoon than it was for the

Table 12. Total number of Oligochaeta and percent of the total Oligochaeta represented by taxa in the plant associated community at Mackinac (reference) marsh, Lake Huron, 1996.

Taxa	Date			
	June	July	August	September
Naididae				
Stylaria spp.	38 %	94 %	83 %	77%
others	59 %	6 %	17 %	18 %
Tubificidae	3 %	<1 %	<1%	5 %
Total number	**676**	**1,286**	**488**	**742**

Table 13. Total number of Oligochaeta collected and percent of the total Oligochaeta represented by taxa in the plant associated community of Cedarville (impacted) marsh, Lake Huron, 1996.

Taxa	Date			
	June	July	August	September
		Percent		
Naididae				
Stylaria spp.	31 %	66 %	41 %	37 %
others	68 %	32 %	47 %	61 %
Tubificidae	1 %	2 %	12 %	2 %
Total number	**849**	**2,105**	**1,016**	**678**

Table 14. Total number of Oligochaeta per m² and percent of the total Oligochaeta represented by taxa in the sediment in Mackinac (reference) marsh, Lake Huron, 1996.

Taxa	Date			
	June	July	August	September
Naididae				
Stylaria spp.	21 %	26 %	14 %	15 %
others	48 %	51 %	86 %	61 %
Tubificidae	31 %	23 %	0 %	24 %
Total number	**32,061/m²**	**10,942/m²**	**5,598/m²**	**15,821/m²**

Table 15. Total number of Oligochaeta per m² and percent of the total Oligochaeta represented by taxa in the sediment in Cedarville (impacted) marsh, Lake Huron, 1996.

Taxa	Date			
	June	July	August	September
Naididae				
Stylaria spp.	60 %	13 %	9 %	0 %
others	24 %	87 %	60 %	90 %
Tubificidae	16 %	0 %	31 %	10 %
Total number	**29,008/m²**	**6,107/m²**	**16,285/m²**	**37,659/m²**

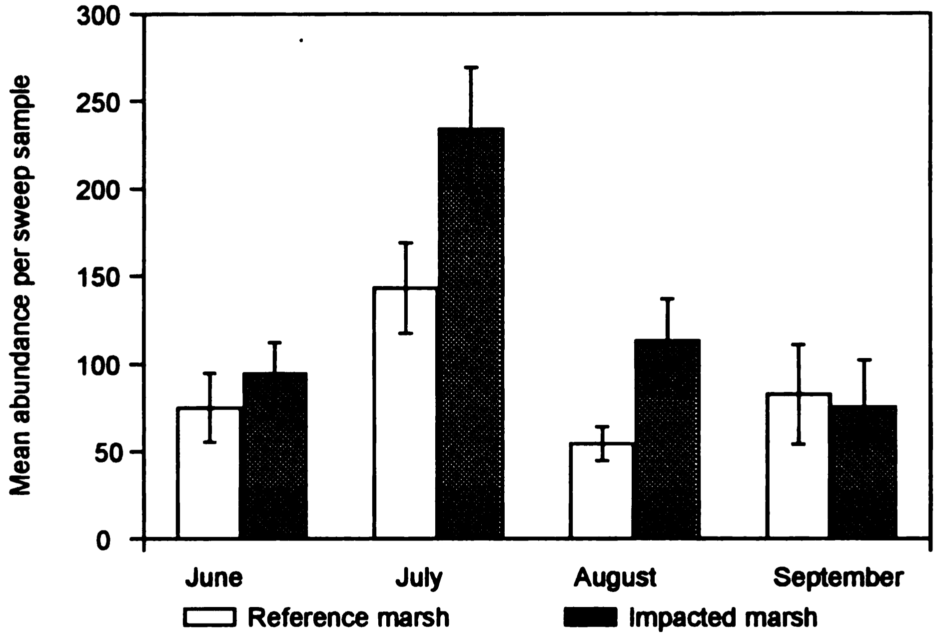

Figure 20. Abundance trends, including standard error, of Oligochaeta in the plant associated community of Mackinac (reference) and Cedarville (impacted) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

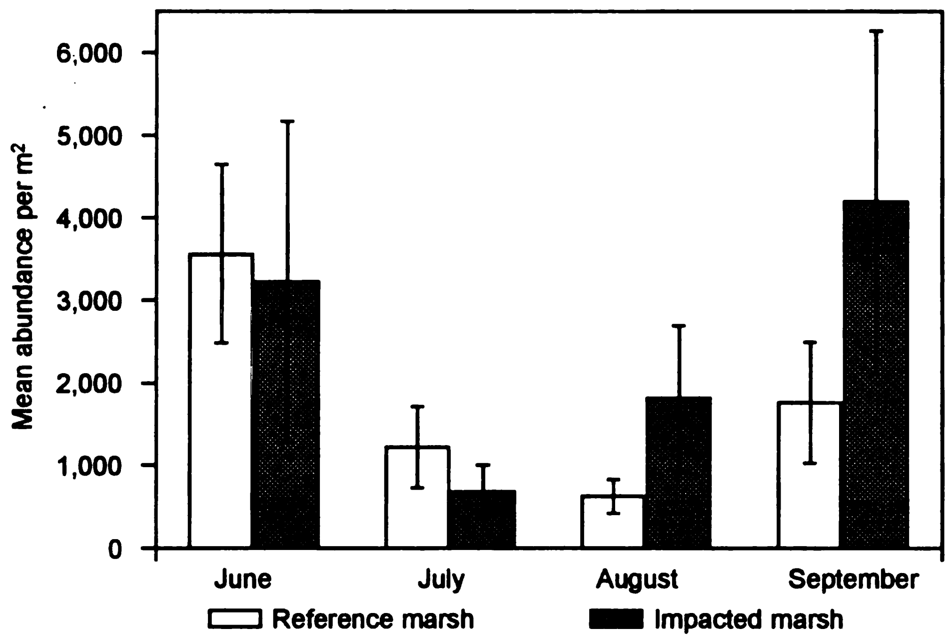

Figure 21. Abundance trends, including standard error, of Oligochaeta in the sediment community of Mackinac (reference) and Cedarville (impacted) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

reference marsh (Figure 21 and Table 15), but this difference was not significant ($p=0.86$). Oligochaeta densities in the sediment community of the reference marsh were never significantly different from Oligochaeta densities in the impacted marsh (Figure 21). The peak mean density in the reference marsh occurred in June at $3{,}562 \cdot m^{-2}$, while the mean peak for the impacted marsh occurred in September at $4{,}184 \cdot m^{-2}$.

The Tubificidae were never abundant within the plant associated samples at the reference marsh reaching a peak of 4 per dip-net sample or 5% of Oligochaeta in September (Table 12), Tubificidae peaked at 13.5 per dip-net sample or 12% of the plant associated Oligochaeta in the impacted marsh in August (Table 13). Tubificidae were more important components of the sediment associated communities at both sites comprising 31% of the Oligochaeta sediment community at the reference marsh in June (Table 14) and up to 31% of the Oligochaeta plant associated community in the impacted marsh in August (Table 15).

Tubificidae are often classified as very pollution tolerant but no consistent differences occurred between the impacted and reference sites (Tables 14 and 15). During wastewater discharge in September into the impacted site, for example, Tubificidae densities reached $3{,}797 \cdot m^{-2}$ at the reference site and $3{,}766 \cdot m^{-2}$ at the impacted site (Tables 14 and 15).

Gastropoda was the most diverse group of non-insects in the sediment and plant associated communities at both the impacted and reference marsh (Table 11). Gastropoda associated with the plant and sediment communities reached peak abundances in September with numbers generally increasing as the season progressed from June through September (Figures 22 and 23). Gastropoda was dominated by the Planorbidae snail, *Gyraulus parvus*, in the plant and sediment community at both the impacted and the reference marsh (Figure

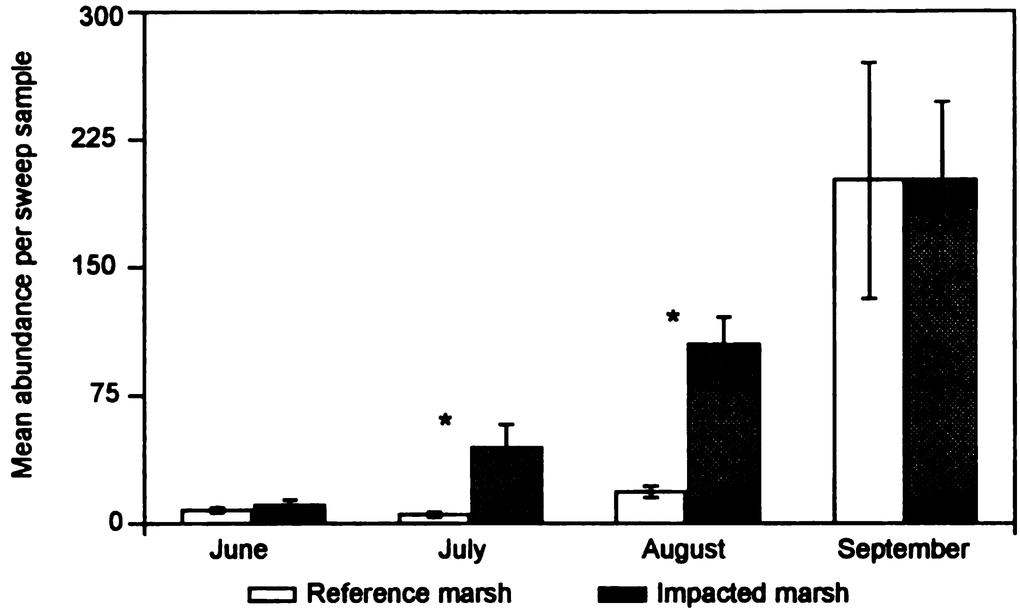

Figure 22. Abundance trends, including standard error, of Gastropoda in the plant associated community of Cedarville (impacted) and Mackinac (reference) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

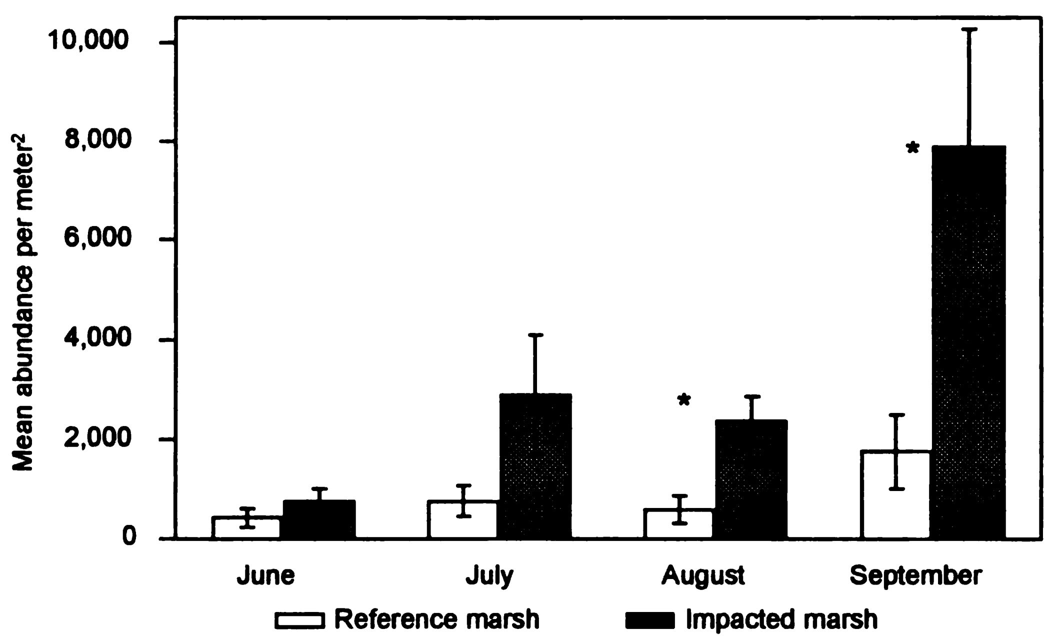

Figure 23. Abundance trends, including standard error, of Gastropoda in the sediment associated community of Cedarville (impacted) and Mackinac (reference) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

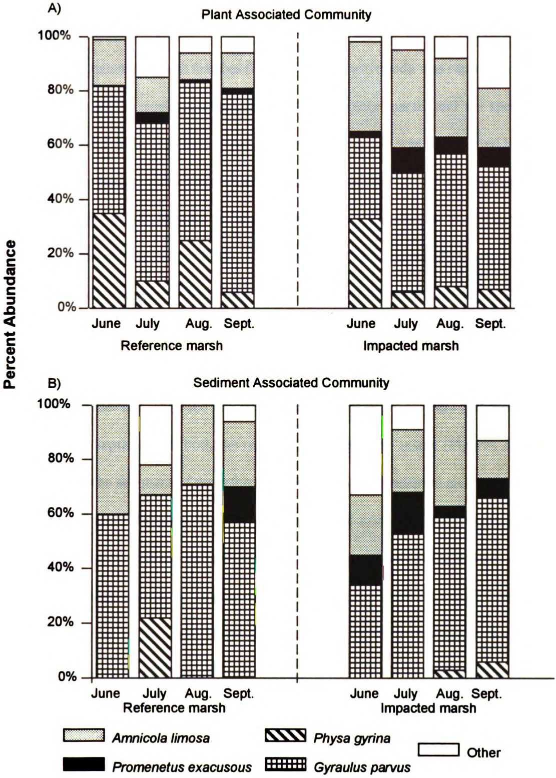

Figure 24. Community composition of the Gastropoda community based on abundance at Cedarville (impacted) and Mackinac (reference) marshes, Lake Huron, 1996. A) Plant associated community and B) Sediment associated community.

24). *Physa gyrina*, and *Amnicola limosa* were also dominant Gastropoda in the plant associated communities at both marshes (Figure 24A). Gastropoda was represented by eight species in the plant associated community at the reference marsh, and ten species in the impacted marsh (Table 11). Sediment associated Gastropoda was represented by five taxa at the reference marsh, and seven taxa at the impacted marsh (Table 11). Gastropoda were more abundant in both sediment and plant associated communities at the impacted marsh, 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 (Figures 22 and 23).

Gastropoda were present in relatively low numbers in the plant associated community early in the summer and increased to a peak abundance of approximately 200 snails per dip-net sample in September at both the reference and impacted marsh (Figures 22 and 23). Gastropoda in the sediment of the reference marsh, never represented more than 6% of the macroinvertebrate community but were more abundant and represented up to 20% of the sediment macroinvertebrate community at the impacted marsh (Figure 7B). *G. parvus* dominated the plant associated Gastropoda community of the impacted marsh from July until September (Figure 24A). Gastropoda collected in the June plant associated samples of the impacted marsh was more evenly distributed, *P. gyrina*, *A. limosa* and *G. parvus* all represented around 30% of the Gastropoda community composition (Table 11 and Figure 24A).

Amphipoda were more abundant in the sediment and plant associated communities at the impacted marsh compared with the reference marsh (Table 11, Figures 25 and 26).

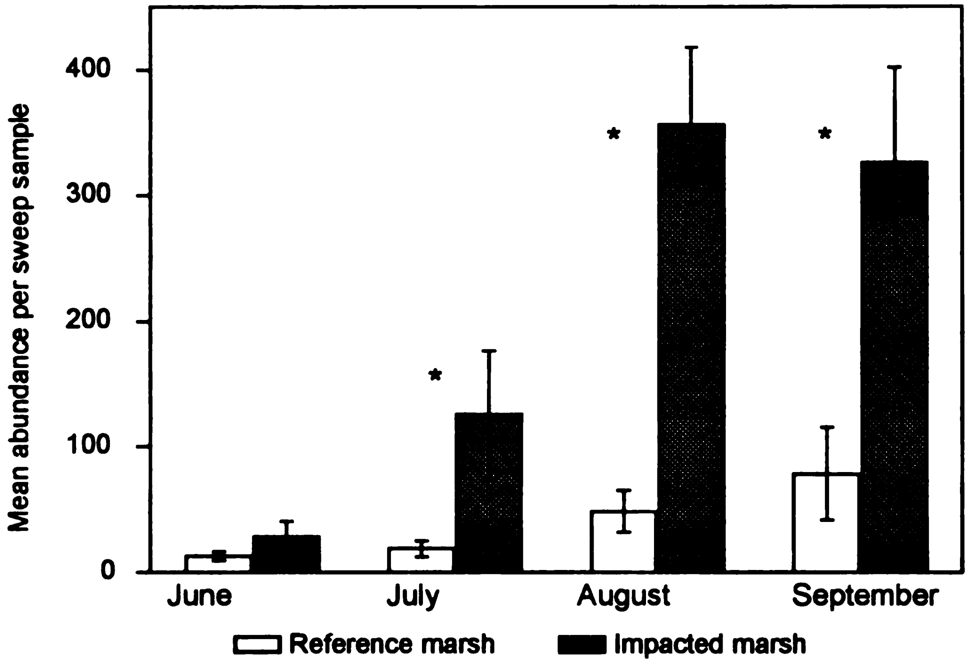

Figure 25. Abundance trends, including standard error, of Amphipoda in the plant associated community of Cedarville (impacted) and Mackinac (reference) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

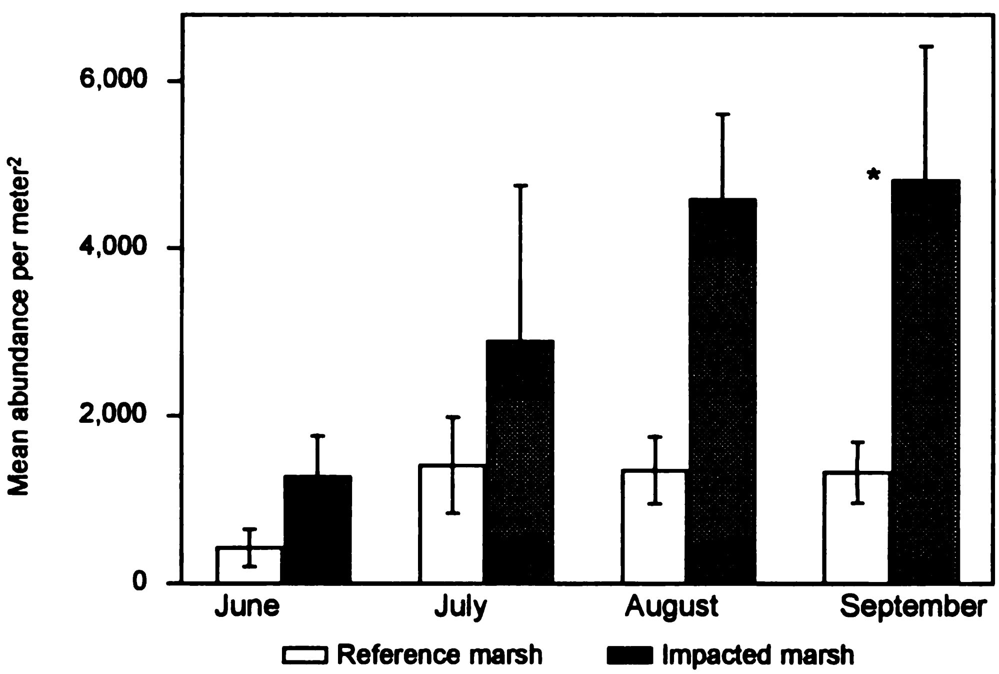

Figure 26. Abundance trends, including standard error, of Amphipoda in the sediment associated community of Cedarville (impacted) and Mackinac (reference) marshes, Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

Significant differences ($p<0.05$) between the two marshes occurred in July, August and September in the plant associated community, and in September in the sediment associated community (Figures 25 and 26). Amphipoda abundance was relatively low in the plant associated community of the reference marsh in June and July, and increased in August and September reaching a maximum mean abundance in September of 78 individuals per sample (Figure 25). Amphipoda abundance in the plant associated community of the impacted marsh also reached high abundances in August and September at levels nearly three times greater than those exhibited at the reference marsh (Figure 25). Amphipoda density in the sediment community remained relatively stable averaging around 1,400 individuals $\cdot m^{-2}$ at the reference marsh, except during June where they were at their lowest mean density of $424 \cdot m^{-2}$ (Figure 26). Sediment associated Amphipoda density at the impacted marsh was greater than at the reference marsh on every sampling date and reached a maximum mean density of approximately $4{,}500 \cdot m^{-2}$ in September, the only date where differences were significant at the $p=0.05$ level (Figure 26). The overall dominant Amphipoda in both the sediment and plant associated community at the impacted marsh was *Gammarus spp.*, while *Hyalella azteca* was dominant in the reference marsh (Table 11). *Gammarus spp.* abundance was 45 fold greater at the impacted marsh in plant associated samples and 8 fold greater at the impacted marsh sediment samples compared to the reference marsh (Table 11) and these differences were significant at the $p=0.05$ level. *H. azteca* abundance was also greater at the impacted marsh compared to the reference marsh but differences were less dramatic (Table 11) and were not significant.

Isopoda were much larger components of the plant associated community at the impacted marsh than at the reference marsh with over 2,000 individuals collected, compared with only 730 individuals collected in the plant associated community of the reference marsh (Table 11). Even so, differences between the two sites were only significantly different during August (Figure 27) because of high variance. Isopoda abundance was also higher in the sediment community of the impacted marsh compared to the reference marsh with an average density of $1{,}534 \cdot m^{-2}$ at the impacted marsh, compared with $912 \cdot m^{-2}$ in the sediment community of the reference marsh but these differences were not significant ($p=0.05$) (Table 11 and Figures 27 and 28). Two species of Isopoda were collected in both marshes, *Lirceus lineatus* and *Racovitzai racovitzai* (Table 11). *L. lineatus* was the dominant Isopoda in the plant associated community at both marshes (Table 11), while *R. racovitzai* was the dominate species in the sediment community of both marshes (Table 11).

The remainder of the taxa occurred in low abundances in the plant and sediment communities in both marshes (Table 11). Four species of Leeches (Hirudinea) were collected in the plant associated samples at the impacted site; *Mooreobdella spp.*, *Alboglossiphonia heteroclita*, *Batrachobdella phalera* and *Helobdella stagnalis* (Table 11). Hirudinea were represented by two families with only one identifiable genera, *Theromyyzon spp.*, in the plant associated community of the reference marsh (Table 11). Orb-weaving spiders *Tetragnatha laboriosa* commonly found near water, were collected at the reference marsh (Table 11). Other less common members of the sediment macroinvertebrate community included the mites (Oribatei and Hydracarina), Sphaeriidae and Tricladida (Table 11).

Zooplankton Community:

While sampling gear was not designed specifically for sampling microcrustaceans,

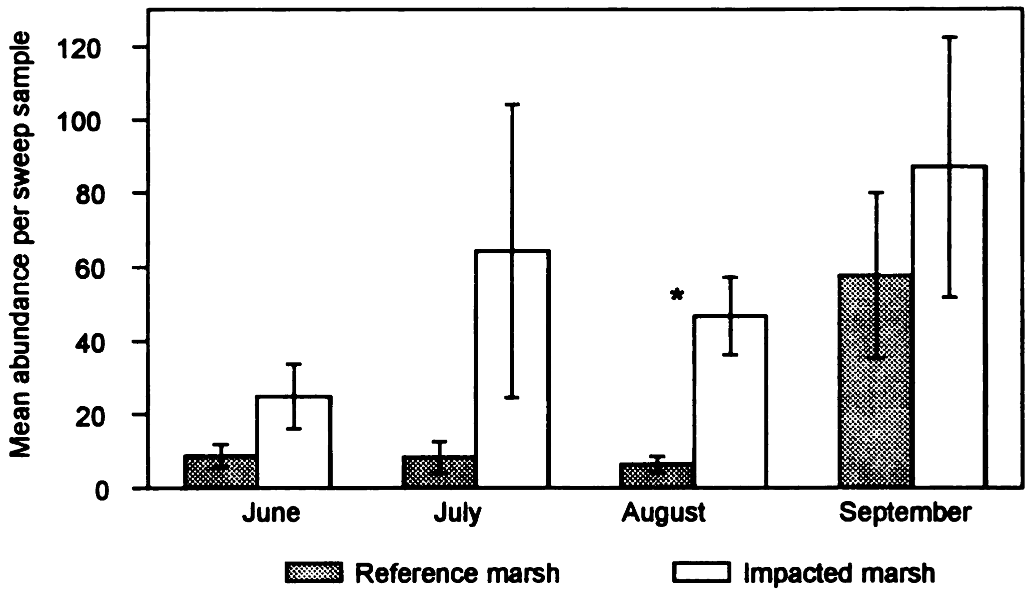

Figure 27. Abundance trends, including standard error, of Isopoda in the plant associated community of Cedarville (impacted) and Mackinac (reference) marshes Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

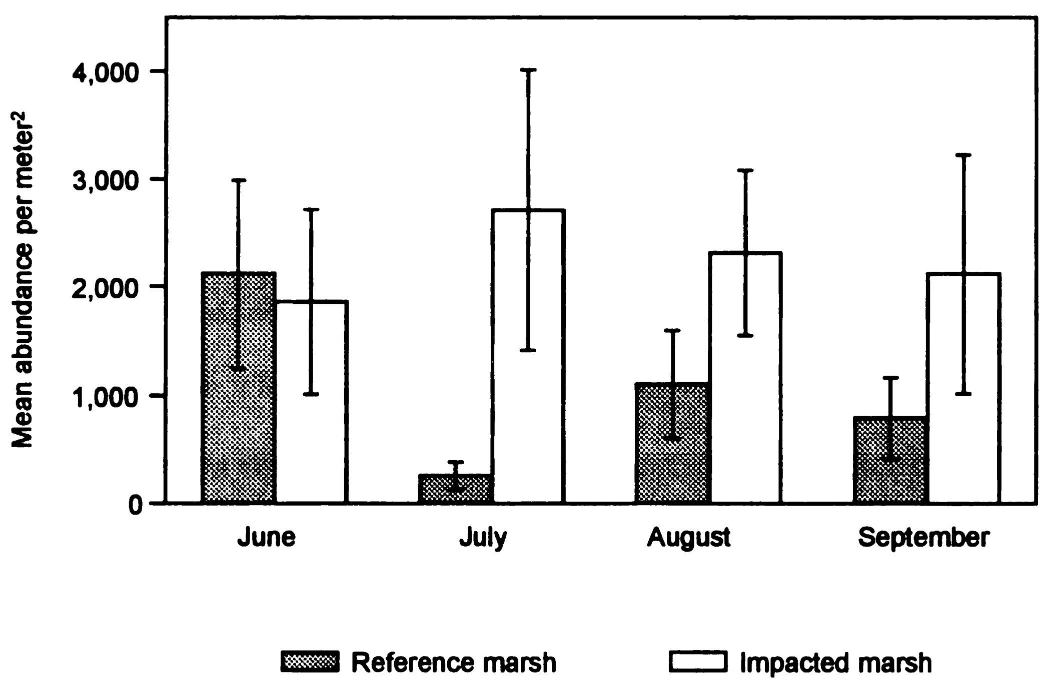

Figure 28. Abundance trends, including standard error, of Isopoda in the sediment community of Cedarville (impacted) and Mackinac (reference) marshes Lake Huron, 1996. *Mann-Whitney U test: Significant at $\alpha = 0.05$.

many were collected in the dip-net and core samples. Cladocera and Copepoda were identified to the species level at the site nearest the source of discharge in both the dip-net and core samples of the impacted and reference marsh in June and September and are included here (Table 16), but results should be viewed as biased by the sampling gear used.

Cladocera was consistently the most abundant group of zooplankters in the plant associated community in both marshes (Table 16) with *Bosmina longirostris* being the most common species collected. A major portion of the sediment associated zooplankton community at both marshes was composed of Ostracoda and Copepoda. The Cyclopoida Copepoda *Diacyclops thomasi* was the most common species collected.

Macroinvertebrate Trends Along a Pollution Gradient:

Ephemeroptera and Trichoptera were selected as pollution sensitive taxa that may occur in reduced abundances nearest the source of discharge. The abundance of these taxa at each site was plotted independently along the transect representing a gradient moving away from the source of discharge. A linear regression showed no consistent pattern in abundance in response to distance from discharge. Isopoda, Amphipoda, Gastropoda, Oligochaeta, and Chironomidae were selected as tolerant taxa that may occur in greater abundances nearest the source of discharge. Once again, the abundance of these taxa were plotted at the sites along the transect moving away from the source of discharge. No consistent patterns in response to discharge were apparent. The three sites nearest discharge were then grouped and treated as replicates and compared to the three sites furthest from discharge. None of the groups selected demonstrated a significant difference ($p<0.05$) in abundance between the sites nearest discharge compared with the sites furthest from discharge.

Table 16. Relative abundance of Zooplankton collected in dip-net and core samples, identified near the source of discharge, at Mackinac (reference) and Cedarville (impacted) marshes, Lake Huron, September 1996.

Taxa	Dip-net Samples				Core Samples			
	Reference marsh		Impacted marsh		Reference marsh		Impacted marsh	
	June	Sept.	June	Sept.	June	Sept.	June	Sept.
CLADOCERA								
Bosminidae								
Bosmina longirostris (O.F. Muller)	57%	--	64%	20%	--	--	--	--
Chydoridae								
Alona guttata (Sars)	2%	15%	--	--	--	--	7%	--
Camptocercus rectirostris (Schodler)	8%	30%	4%	19%	67%	17%	5%	--
Daphnia pulex (Leydig)	5%	--	--	6%	--	--	--	2%
Diaphanosoma birgei (Fisher)	15%	20%	4%	17%	--	17%	12%	--
COPEPODA								
Cyclopoida								
Diacyclops nanus (Sars)	--	--	2%	--	--	--	4%	--
Diacyclops thomasi (S. A. Forbes)	10%	25%	15%	28%	33%	49%	53%	86%
Tropocyclops prasinus	1%	--	--	10%	--	17%	--	5%
Mesocyclops edax (S. A. Forbes)	1%	--	2%	--	--	--	2%	--
Harpactacoida	1%	--	5%	--	--	--	--	2%
Leptodiaptomus minutus (Lilljeborg)	--	--	1%	--	--	--	--	--
Copepodite nauplii	--	10%	3%	--	--	--	17%	5%

Proximity to discharge does not appear to impact abundance of sensitive or tolerant macroinvertebrates. Impacts on the macroinvertebrate community in response to sewage effluent affected the overall marsh community and were not restricted to areas near the source of discharge.

Discussion

The macroinvertebrate community composition was dominated at both the reference and the impacted marsh by aquatic insects, Oligochaeta and Gastropoda. The impacted marsh showed signs of moderate degradation in community structure and composition in both the plant and sediment associated communities compared to the reference marsh. The aquatic insect community was a more taxa rich community at the reference marsh than it was at the impacted site in the plant and sediment associated communities. Aquatic insects represented a smaller fraction of the macroinvertebrate community in the plant and sediment community of the impacted marsh compared to the reference marsh (Figure 7). The smaller proportion of aquatic insects at the impacted marsh compared with the reference marsh was the result of reduced abundance of aquatic insects, and higher abundances of non-insect groups such as Amphipoda, Oligochaeta, and Isopoda (Figure 7). Sediment associated Gastropoda ($p=0.002$) and Nematoda ($p=0.005$) represented a greater portion of the macroinvertebrate samples at the impacted site than they did at the reference site, with Nematoda dominating the macroinvertebrate fauna in June (Figure 7B).

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 plant and sediment communities of the impacted marsh compared with the reference marsh (Figures 12 and 13, Table 2). *Caenis spp.*, a relatively tolerant Ephemeropteran (Hilsenhoff 1977), dominated the Ephemeropteran sediment and plant community at both marshes; however, it was present in significantly ($p<0.05$) reduced numbers in the impacted marsh (Table 2). The pollution sensitive, *Hexagenia limbata* (Britt 1955), comprised 7% of the Ephemeroptera plant associated community, and 20% of the sediment community at the reference marsh in August, and was never collected at the impacted marsh (Table 2). The sediment associated Ephemeroptera community at the impacted marsh reflected the greatest signs of degradation. Ephemeroptera were only collected in the June sediment samples at the impacted marsh, however they were present in relatively large numbers in every month of sampling at the reference marsh. Several taxa were present throughout the summer in the reference marsh, but *Caenis amica* was the only species of Ephemeroptera collected from the sediments at the impacted marsh (Table 2).

Trichoptera, also a group sensitive to disturbance in lotic systems, had lower taxa richness and were less abundant in both the sediment ($p=0.04$) and plant associated ($p=0.02$) samples from the impacted marsh than they were in the reference marsh (Figure 9 and Table 2). The dominant Trichoptera species at the impacted marsh was always a member of the family Hydroptilidae (Table 2); however, the reference marsh was dominated at different times by species of Leptoceridae and Phryganeidae (Figure 16). All three of these Trichoptera groups were assigned the same tolerance values for lotic systems by Plafkin et al. (1989). However, the change in species dominance for these marshes may demonstrate a more diverse and possibly more dynamic system at the reference marsh. Although

Trichoptera was uncommon or scarce in the sediment samples of both the impacted and reference wetland, Trichoptera taxa richness was significantly ($p<0.05$) greater at the reference marsh compared to the impacted marsh. Sediment associated Trichoptera reached a peak density in both marshes in September, but the density of Trichoptera was almost five fold greater ($p=0.04$) at the reference marsh than it was at the impacted marsh with a maximum density of 3,053 individuals$\cdot m^{-2}$, compared with a maximum density of 763 individuals$\cdot m^{-2}$ at the impacted marsh (Figure 15).

Several groups of taxa that are characterized as tolerant to disturbance in lotic systems exhibited similar trends of higher relative abundances and total numbers in response to disturbance in the plant associated community of these coastal marshes. Chironomidae made up a larger percentage of the insect community, and were more abundant at the impacted marsh than at the reference marsh (Figures 9A). The tribe *Chironomini,* which includes such pollution tolerant genera as blood worms (*Chironomus*), was consistently the dominant group of Chironomidae in the plant associated community of the impacted marsh, but was the dominant Chironomidae only in June and September at the reference marsh (Tables 3 and 4). Amphipoda and Isopoda, which are commonly associated with organic pollution loading in lotic systems (Plafkin et al. 1989), were significantly more ($p=0.005$) prominent in the plant associated community at the impacted site than at the reference site (Figure 7A). Amphipoda abundance was more than five times greater ($p=0.004$) in the plant community of the impacted marsh as compared to the reference marsh (Figure 7A).

Although aquatic insects usually dominated the macroinvertebrate community at both the impacted and reference marsh, the impacted marsh was dominated during July by

Oligochaeta (Figure 7A). Oligochaeta were more abundant in the plant associated community at the impacted marsh than at the reference marsh for every sampling period. Oligochaeta derive most of their nutrition from bacteria and are able to withstand considerable oxygen depletion and are frequently associated with organic pollution loading in streams (Brinkhurst and Cook 1974).

Significant differences in the abundance of tolerant taxa were also detected ($p<0.05$) in the sediment community between the impacted and reference marshes. Stream communities with 90% of the aquatic insect community represented by Chironomidae are generally impaired systems (Hilsenhoff pers. comm.). Chironomidae represented nearly 90% of the aquatic insect community at the impacted marsh compared to approximately 60% at the reference marsh (Figure 9B). Isopoda, which are assigned a very high pollution tolerance value in Hilsenhoff's (1988) Family Biotic Index, were also a larger component of the macroinvertebrate community of the impacted marsh compared with the reference marsh (Figures 26 and 27). Over the course of the sampling season, nearly twice as many Isopoda were collected from the impacted marsh as were collected from the reference marsh. Amphipoda was a significantly ($p<0.05$) larger component of the macroinvertebrate community within the sediment of the impacted marsh compared to the reference marsh (Figure 7B). Oligochaeta or "sludge worms" were a major part of the sediment community at both the impacted and reference marshes. Oligochaeta were more abundant in the sediment community at the impacted marsh than they were at the reference marsh every month of sampling with the exception of June (Figure 20). Oligochaeta reached a maximum mean density of $3{,}652 \cdot m^{-2}$ in the sediment of the reference marsh during June (Figure 20). Kairesalo and Koskimies (1987) found a similar peak in Oligochaeta densities in early June. However, Oligochaeta in the impacted marsh sediment reached a maximum mean density of $4{,}184 \cdot m^{-2}$ in September, the month in which sewage effluent from the wastewater treatment lagoon was discharged, and these differences were not significant ($p=0.30$).

Gastropoda were more diverse and abundant in the plant and sediment community of the impacted marsh than they were in the reference marsh (Table 11). Mason et al. (1970) found *Physa spp.*, *Ferrissia spp.*, and *Promenetus spp.* were often found in moderately to grossly polluted situations. In this study *Promenetus exacuous* was very common in the sediment of the impacted marsh, yet was only found at the reference marsh in September and in low numbers (Figure 24B). *Physa spp.* and *Ferrissia spp.* were also more abundant in the sediment and plant associated community at the impacted marsh compared to the reference marsh (Table 11). Berg and Ockelmann (1959) have shown that many Gastropoda can maintain oxygen consumption, despite a lowering of dissolved oxygen, until a critical threshold is reached, suggesting that many Gastropoda may be able to withstand brief periods of low dissolved oxygen.

Smock and Stoneburner (1980), found that macroinvertebrate densities increased with the onset and progressive senescence of vegetation. The plant associated macroinvertebrate data from both the reference and impacted marsh supported their suggestion (Figure 8). Total macroinvertebrate abundances increased throughout the sampling season reaching a peak abundance in September corresponding with the onset and senescence of *S. acutus* at the reference marsh (Figure 8A). However, peak abundance of aquatic insects corresponded with the onset and senescence of *S. acutus* only at the reference marsh (Figure 8A). At the impacted marsh, aquatic insects reached peak abundance in August, a month earlier than at the reference marsh. In September, the reference marsh had nearly twice the number of aquatic insects ($p=0.29$) when compared to the plant community of the impacted marsh. Many of aquatic insects at the reference marsh were Ephemeroptera and Trichoptera, and these taxa were significantly ($p<0.05$) less abundant at the impacted site (Figure 9A). The aquatic insects at the impacted marsh reached maximum abundance before discharge from the wastewater treatment lagoons occurred in August, whereas they reached maximum abundance in the reference marsh in September.

CHAPTER TWO

IMPLICATIONS FOR THE DEVELOPMENT OF A MULTIMETRIC INDEX OF ECOLOGICAL INTEGRITY

Introduction

Early biological monitoring programs consisted of biotic indexes that were primarily sensitive to organic effluent and sedimentation (Kolkwitz and Marsson 1908 as cited in Kemp et al. (1967)). The most common approach in early biological monitoring of aquatic systems involved the ranking of taxa, typically family, genera or species of limited numbers of major groups of organisms such as invertebrates or fish, where taxa were assigned numerical values on a scale ranging from pollution intolerant to pollution tolerant (Karr and Chu 1997). An average pollution tolerance level was then assessed for each sample site (Hilsenhoff 1977, 1982 and 1987, Armitage et al. 1983). These biological indices were limited in their ability to detect degradation from toxins pollution or from altered physical habitat or flow (Karr and Chu 1997). This led to the use of toxicity bioassay approaches, which typically examined the tolerances of only a few species, usually the most tolerant taxa, to a known toxicant. This bioassay, toxicity approach underestimates the effects of contaminants (Karr and Chu 1997).

The need for more comprehensive monitoring systems led to the development of the multimetric index based on several major groups of organisms and/or physical habitat characteristics (Karr and Chu 1997).

The development of multimetric indices of biological condition gained favor in 1981 with Karr's (1981) fish based, Index of Biological Integrity (IBI). This first multimetric index was designed to include a range of attributes (a measurable component of a biological system), of fish assemblages for use in monitoring the health of stream systems (Karr 1981 and 1986). Shortly thereafter, Ohio EPA (1988) adopted the concepts involved in the IBI to develop an invertebrate community index (ICI) evaluation system using benthic invertebrates. A multimetric index has also been developed to use invertebrates to assess the water quality for rivers of the Tennessee Valley (Kerans and Karr 1994). Multimetric biological indexes are currently used in monitoring the health of streams in 42 states, and 6 additional states are in the process of developing them (Karr and Chu 1997). Multimetric indices 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 that includes a variety of such metrics integrates information from ecosystem, community, population and individual organism levels (Gerritsen 1995, Karr 1991, Barbour et al. 1995), and it can be expressed in numbers and words.

Changes in the physical, chemical, and biological environment resulting from human activities alter assemblages. These changes may be changes in species composition, species richness, or trophic structure, such as a decrease in top carnivores or an increase in omnivores; or shifts from specialist to generalist in food or reproductive habitats, reflecting shifts in food web organization, including energy flow and nutrient cycling (Karr and Chu 1997). Multimetric indices incorporate this information by including metrics such as the percentage of predators, omnivores, or trophic feeding groups (Karr and Chu 1997). A multimetric index should be able to demonstrate that an attribute, has a consistent quantitative change across a range, or gradient of human influence (Karr and Chu 1997). They should increase or decrease as human influences increases. They should be sensitive to a range of biological stress. Multimetric biological indices are now well documented as effective in the assessment of ecological condition in a variety of management settings, with many taxa, and in diverse geographic regions (Karr and Chu 1997). Multimetric indices are predominantly used in the assessment of lotic systems, yet may be a useful tool in the assessment of lentic or wetland systems. The objectives of this study are to select and test various metrics for use in the future development of a multimetric index of ecological integrity for use in northern Lake Huron coastal marshes.

Methods

1. Metric selection

I considered 38 structural or functional measurements of the macroinvertebrate assemblages that had been used or suggested by other authors as having potential use as biological metrics (Table 17) (Resh et al. 1995, Kerans et al. 1992, Barbour et al. 1996, Resh et al. 1995, Kerans and Karr 1994). Potential metrics included elements of macroinvertebrate community structure and composition, including measures of the trophic and functional composition of the assemblage considered to be indicative of ecological processes (Karr and Chu 1997).

Table 17. Definitions of potential metrics and expected direction of metric response to increasing perturbation in Northern Lake Huron coastal marshes (selected and modified from Karr and Kerans 1992).

Category	Potential Metrics	Hypothesized effect of impact
Community structure and composition		
Richness measures		
	Ephemeroptera plus Trichoptera	Decrease
	Number of Crustacea plus Mollusca	Decrease
	Number of Diptera	Decrease
	Number of Ephemeroptera	Decrease
	Number of families	Decrease
	Number of Trichoptera	Decrease
	Total abundance	Decrease
	Total taxa richness	Decrease
Enumerations		
	Proportion of individuals as Amphipoda	Decrease
	Proportion of individuals as Chironomidae	Increase
	Proportion of individuals as Chrionomini	Increase
	Proportion of individuals as Crustacea plus Mollusca	Decrease
	Proportion of individuals as Diptera	Increase
	Proportion of the dominant taxon	Increase
	Proportion of individuals as Ephemeroptera	Decrease
	Proportion of individuals as Gastropoda	Decrease
	Proportion of individuals as Isopoda	Increase
	Proportion of individuals as Odonata	Increase
	Proportion of individuals as Oligochaeta	Increase
	Proportion of individuals as Orthocladiinae	Decrease
	Proportion of individuals as Tanytarsini	Decrease
	Proportion of individuals as Trichoptera	Decrease
	Proportion of individuals as Tubificidae	Increase
	Proportion of individuals as Sphaeriidae	Decrease
	Proportion of individuals as *Stylaria spp.*	Increase
Trophic and Functional composition		
	Number of scraper plus piercer taxa	Decrease
	Proportion of individuals as collector-gatherers	Increase
	Proportion of individuals as filterers	Decrease
	Proportion of individuals as predators	Decrease
	Proportion of individuals as scrapers	Decrease
	Proportion of individuals as shredders	Decrease
	Ratio of scrapers/collector-filterers	Increase
Community diversity and similarity indices		
	Coefficient of community loss	Decrease
	Evenness	Decrease
	Jaccard coefficient	Decrease
	Margalef diversity	Decrease
	Shannon diversity	Decrease
	Simpson diversity	Decrease

The first group included elements on macroinvertebrate community structure and composition consisting of richness measures and enumerations (taken from Kerans et al. 1992) (Table 17). Metrics based on measures of community structure and composition were selected to provide information on the make-up of the assemblage and the relative contribution of the population to the total fauna. Richness measures included metrics based on macroinvertebrate taxa that are generally regarded as sensitive to anthropogenic disturbances including total abundance. Loss of taxa in sensitive groups is an indication of perturbation (Wallace et al. 1996, Barbour et al. 1996). Metrics of sensitive macroinvertebrate taxa included the total number of Odonata taxa and the total number of Crustacea plus mollusca taxa (Karr and Chu 1997). Richness metrics considered were also based on the presence of EPT (Ephemeroptera, Trichoptera and Plecoptera) taxa which have been found to be universal and applicable in monitoring the health of many stream systems (Plafkin et al. 1989, Barbour et al. 1996). However this metric was modified to exclude Plecoptera due to lack-of or low number of the predominantly lotic Plecoptera, within the northern lake Huron coastal marshes. Resulting metrics were measurements of Ephemeroptera plus Trichoptera taxa richness, and the number of Trichoptera taxa and Ephemeroptera taxa independently. Richness metrics were also examined which have been shown to decrease as human disturbances increases and included measures of total taxa richness and total number of families (Fore and Karr 1996, Barbour et al. 1996, Resh et al. 1995) (Table 17).

Community structure and composition metrics of relative abundance (enumerations) are based on the premise that a healthy and stable assemblage will be relatively consistent in the proportional representation of major taxa, though individual abundances may vary in magnitude. Metrics of relative abundances of taxa that have been shown to decrease in the presence of perturbation, include percent Ephemeroptera, Trichoptera, Amphipoda, Crustacea plus Mollusca, Gastropoda, Sphaeriidae, and Tanytarsini (Fore and Karr 1996, Barbour et al. 1996, and Resh et al. 1995). Metrics considered based on the relative abundances of taxa that would most likely increase in the presence of perturbation were percent composition of Oligochaeta, Tubificidae, *Stylaria spp.*, Odonata, Diptera, Chironomidae, Orthocladiinae, Chironomidae, and Isopoda (Barbour et al. 1996). Although, certain individual taxa within each group may be relatively sensitive to pollution they typically should have no effect on the overall result of the metric (Barbour et al. 1996). Percentage of the dominant taxon was also measured which is a simple measure of redundancy. A high level of redundancy corresponds with a pollution tolerant organism dominating a community resulting in a lower overall diversity (Plafkin et al. 1989, Barbour et al. 1996).

Metrics on the trophic and functional composition of the assemblage were selected based on the premises that trophic metrics are surrogates of complex processes such as trophic interaction, production, and food source availability (Karr et al. 1986, Cummins et al. 1989, Plafkin et al. 1989, Barbour et al. 1996) (Table 17). The majority of trophic metrics were evaluated as relative abundance (percentage). Metrics considered included relative abundances of sensitive organisms such as specialized feeders including scrapers, piercers, filter feeders and shredders which are expected to decrease with increasing disturbance (Barbour et al. 1996, Wallace et al. 1996). The number of scraper plus piercer taxa was also included. This metric includes taxa that feed primarily on diatoms (scrapers) and living macrophytes (piercers) and is expected to decrease with increasing perturbation (Barbour et al. 1996). Generalist consumers (collector-gatherers) have a broader range of acceptable food materials than do specialist consumers and are, therefore, more likely to be tolerant to pollution that might alter the availability of certain foods. Therefore, a metric of the relative abundance of collector-gatherers is expected to increase in the presence of perturbation (Cummins et al. 1989). The ratio of scrapers to collector-filterers was also considered as a potential metric to reflect available food resources. The dominance of collector-filterers may reflect organic enrichment (Resh and Jackson 1993). In addition, four functional feeding group metrics suggested by Merritt et al. (1996) were used to evaluate several ecosystem attributes (Table 18).

Community diversity and similarity indices were examined as potential metrics, to reflect the diversity of the aquatic assemblages (Resh et al. 1995). Three diversity indices, one evenness index, the Coefficient of Community Loss, and the Jaccard Coefficient were applied.

(1) Shannon Index	$H' = -\sum Pi \log_2 Pi$.
(2) Simpson's Index	$1-D = 1-\sum N_i(N_i-1)/N(N-1)$
(3) Margalef's Index	$D = S-1/Ln(N)$
(4) Evenness Index	$J' = H'/Log_2S$
(6) Coefficient of Community Loss	d-a/e
(7) Jaccard Coefficient	a/ a+b+c

where

N_i=	number of individuals of species i
N=	the total number of individuals in a sample
P_i=	N_i/N
S=	total species number
a=	number of taxa common to both samples
b=	number of taxa present in sample B but not A
c=	number of taxa present in sample A but not b.
d=	total number of taxa present in sample a
e=	total number of taxa present in sample b

Table 18. 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).

The Shannon Index is a measure of information content, and incorporates both richness and evenness in its formula (Merritt et al. 1996). Simpson's Index takes into account both the abundance patterns and the species richness (Cao et al. 1996). The Margalef formula differs from the other two in that is does not contain an evenness component (Merritt et al. 1996). Jaccard Coefficient of Community similarity measures the degree of similarity in taxonomic composition between two sites in terms of taxon presence or absence. The Jaccard Coefficient discriminates between highly similar sites. Coefficient values, ranging from 0 to 1.0, increase as the degree of similarity with the reference station increase (Jaccard 1912, Ohio EPA 1987). The Coefficient of Community Loss Index measures the loss of benthic taxa between a reference site and the site of comparison, it is an index of compositional dissimilarity, communities are expected to become more dissimilar as stress increases with values increasing as the degree of dissimilarity with the reference site increases (Courtemanch and Davies 1987).

2. Metric testing

Macroinvertebrate data from dip-net samples (plant associated communities, chapter one) were used to test the various metrics and indices for reliability as water quality indicators between the reference and the impacted marsh. A metric should be capable of distinguishing differences in anthropogenic impacts among marshes. Therefore, the initial step was to select metrics based on low within site variability, followed by a determination of the metric's ability to detect the hypothesized effect of impact, and finally to examine the sensitivity of the potential metrics. Potential metrics were calculated for two months, August and September. Months for metric testing were selected based on a greater abundance and diversity of macroinvertebrates, and a higher frequency of late instar larvae, which aid in a more accurate identification of most taxa. Macrophytes also contained the largest amount of periphyton at this time, which is a major food source for many macroinvertebrates, including the scrapers. August samples were used to test whether long-term effects would be detectable (no wastewater discharge had occurred since June 5). September samples should reflect both short and long term impacts of wastewater discharge, since wastewater was being discharged when samples were taken.

The first step was to select metrics that had minimal variability from within-marsh factors so that differences resulting from anthropogenic impacts could be detected. A highly variable metric within the reference site would not useful because it would have low discriminatory power between the impaired and the unimpaired sites (Barbour et al. 1996). Conversely, a metric with a narrow variance within the reference marsh is potentially useful for detecting biological change in response to disturbance. This was determined by measuring within-site variability of the potential metrics. Coefficients of variation for each potential metric were calculated among the nine sites within each marsh (Karr and Chu 1997). Metrics were considered to have high variability if their coefficients of variations were greater than 50%, indicating high within-site variability. Chemical analysis showed that impact from sewage effluent to be greatest at sites less than 260 meters from discharge (at the impacted marsh). This type of point source disturbance could potentially cause high within site variability at the impacted marsh. Therefore, only coefficient of variations for the reference marsh were used as a method of screening potential metrics, because the reference marsh was not influenced by point source pollution. Metrics were not considered for further consideration as candidate metrics if a high coefficient of variation was detected in either August or September at the reference marsh.

Following this initial screening and selection, potential metrics were evaluated to determine if the hypothesized effect of impact was detected by the metric. Metrics were calculated for each marsh by pooling the macroinvertebrates from all nine sites. Results from both marshes were than plotted next to each other with box-and-whisker plots (e.g. Figure 29). The positioning of the box-and whisker plots from each metric were compared between marshes to determine if the metric either agreed with the predictions of the hypothesized effect of impact, measured no effect of impact, or measured the opposite effect from that predicted based on impacts known to occur in streams. Metrics which detected the hypothesized effect of impact in August and September were selected as metrics likely to pick up long term impacts on the system. Metrics which detected the hypothesized effect of impact during wastewater discharge in September, but measured no effect of impact in August were selected as potential metrics of short term impacts on these systems. Metrics which measured the opposite effect of impact in either month were not considered as candidate metrics but recommended for further analysis.

The sensitivity of each metric (its ability to discriminate between the reference and the impacted site) was then judged according to the degree of interquartile overlap in box-and-whisker plots (Barbour et al. 1996). Metrics were judged to have one of four sensitivity values: a sensitivity value of 3 if no overlap existed in the interquartile range; a sensitivity of 2 if there was some overlap that did not extend to the medians; a sensitivity of 1 if there was a moderate overlap of interquartile ranges but at least 1 median was outside the range; and

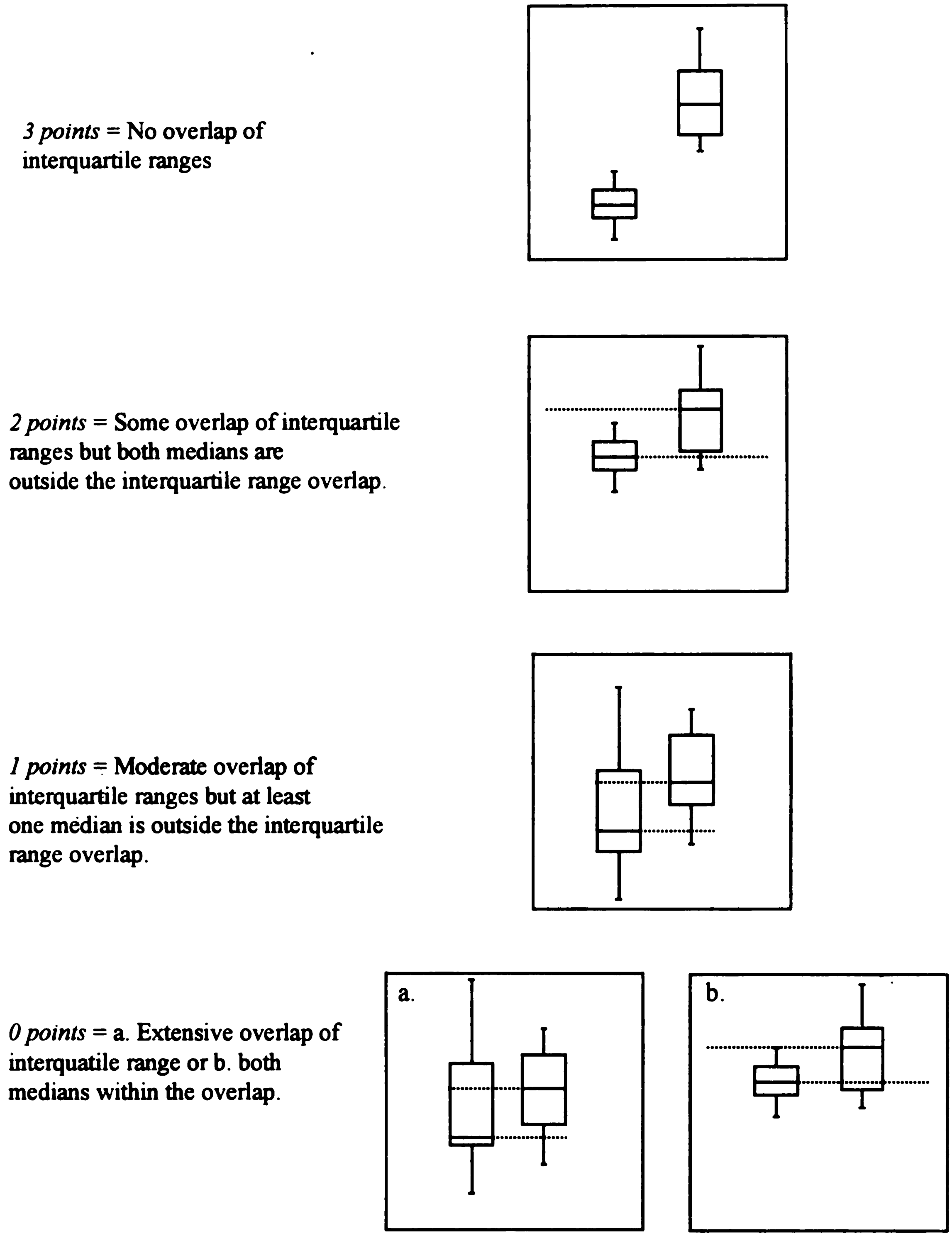

Figure 29. Evaluation of sensitivity of the metrics. Range bars show maximum and minimum of non-outliers; Solid lines inside the box are medians; boxes are interquartile ranges (25%ile to 75%ile), (Taken from Barbour et al. 1996).

a sensitivity of zero if interquartile overlap was considerable, with no discrimination between reference and impaired sites (Barbour et al. 1996) (Figure 29).

In addition, potential metrics were statistically compared between the two marshes in order to determine their ability to detect a significant difference between the two sites. Because metrics were often proportions, they tended not to be normally distributed, so the normality of the distribution of each metric was tested before using a parametric statistical test (Kerans and Karr 1994). If distributions were non-normal ($p<0.05$), richness data were log-transformed, and proportions were arc-sine transformed and the test repeated. If transformation was unsuccessful, a nonparametric Mann-Whitney U test was used for statistical comparisons between sites on a monthly basis.

Metrics were considered candidate metrics based on their ability to consistently detect the hypothesized effect of impact between marshes, and consistently have a low coefficient of variation indicating low within site variability. Statistical comparisons (Mann-Whitney U test) were not used as a means of metric elimination, but were instead used to support previously determined sensitivity values. Final selection of metrics was based on the graphical analysis (Figure 29), because it provided more insight into biology than a simple p-value could.

Results

From an original suite of 38 metrics, six (16%) metrics met the strict criteria of low within site variability in the reference marsh, and no overlap of interquartile ranges between the two marshes for both months (Tables 19 and 20). Thus, these metrics were able to unambiguously separate the impacted from the reference marsh and are excellent candidate

Table 19. Potential metrics and their sensitivity values base on Barbour et al. 1996, Coefficient of variation based on Mackinac values; low (CV<.50), high (CV>.50). From Cedarville and Mackinac marshes, August 1996. Metrics underlined indicate candidate metrics.

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

*Significant value based on Mann-Whitney U test ($P<0.05$)

Table 20. Potential metrics and their sensitivity values base on Barbour et al. 1996, Coefficient of variation based on Mackinac values; low = (CV<.50), high = (CV>.50). From Cedarville and Mackinac marshes, September 1996. Metrics underlined indicate candidate metrics.

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

*Significant value based on Mann-Whitney U test (P<0.05).

metrics for use as indicators of long term impacts on water quality in Northern Lake Huron coastal marshes (Figure 30). Candidate metrics included: (1) proportions of individuals as Ephemeroptera, (2) proportions of individuals as Isopoda, (3) proportions of individuals as Trichoptera (4) proportion of individuals as predators (5) proportion of individuals as filter-feeders, and (5) a ratio of herbivores (shredders of live plants plus scrapers) to detritivores (shredders of detritus plus total collectors) as a surrogate for the production to respiration (P/R) ratio (Tables 19 and 20, Figure 30).

Ephemeroptera and Trichoptera were, as expected (Table 17), sensitive indicators of water quality with proportions decreasing in the impacted marsh (Figure 30). Isopoda, as expected (Table 17), increased with impacts reflecting their known tolerance to pollution (Figure 30). Three functional feeding group based metrics also changed in the direction expected (Tables 19 and 20, Figure 30). The proportion of fauna as predators decreased (Figure 30), perhaps reflecting loss of sensitive top predators. The increase in the herbivore/detritivore ratio at the impacted marsh compared to the reference marsh (Figure 30) is consistent with an expected increase in primary production due to nutrient enrichment in the impacted marsh (Table 18). The increase in filtering collectors in the impacted marsh (Figure 30) is also consistent with the increased suspended particulates expected in the impacted marsh from either increased plankton production and/or particulate organic input into the marsh from the wastewater lagoons.

Another potentially useful metric was the number of Trichoptera taxa (Tables 19 and 20, Figure 31). This metric received a sensitivity value of zero and was classified as showing no effect based on examination of the box-and-whisker plots and p-values in August.

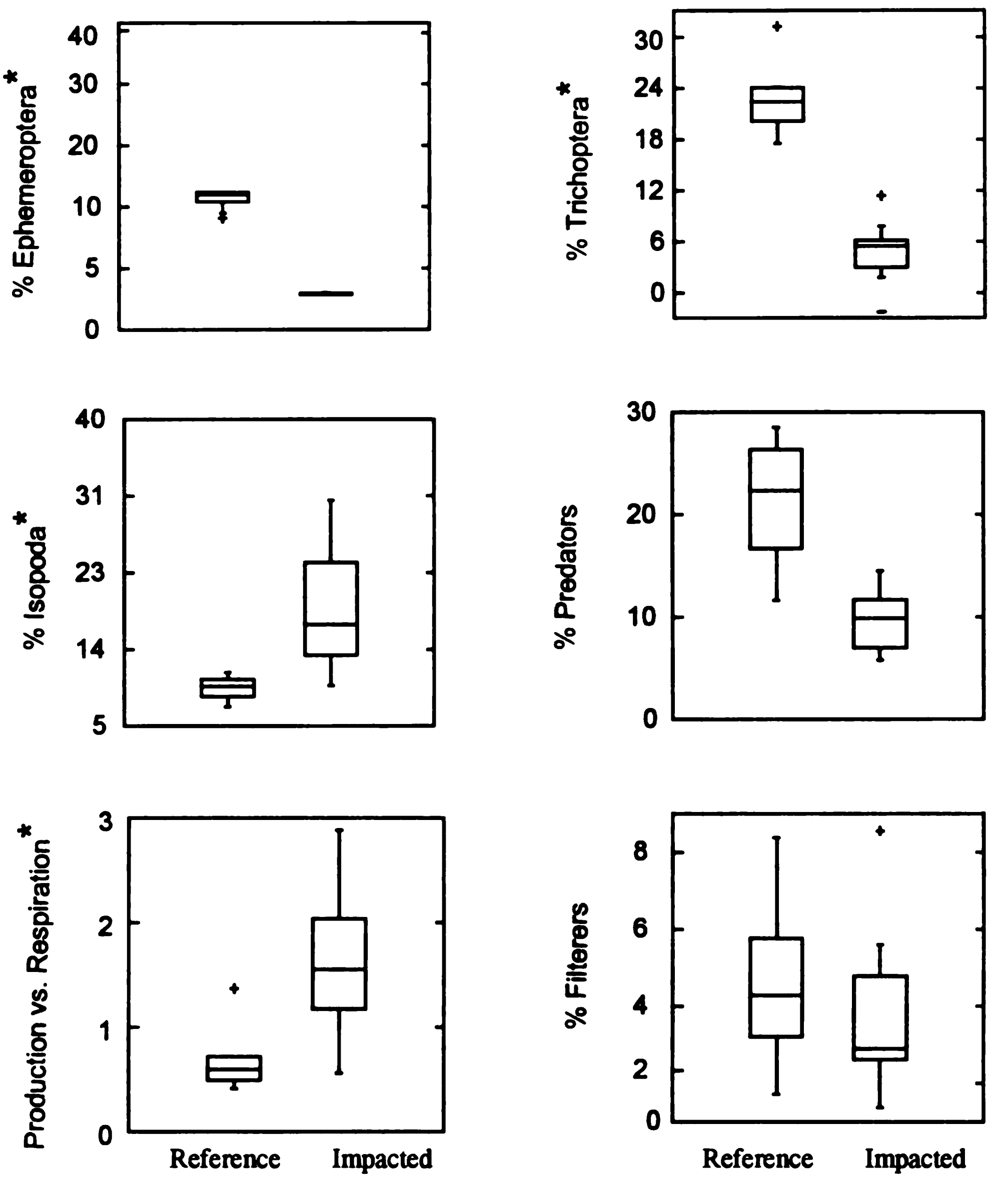

Figure 30. A comparison of candidate metrics of long term impact, for use in Northern Lake Huron Coastal marshes. Metrics determined from Dip net samples at Cedarville (impacted) and Mackinac (reference) marshes, Lake Huron, September 1996. Range bars show maximum and minimum of non-outliers; Solid lines inside the box are medians; boxes are interquartile ranges (25%ile to 75%ile); dots are outliers. (Taken from Barbour et al. 1996). *Mann-Whitney U test: Significant at $\alpha = 0.05$.

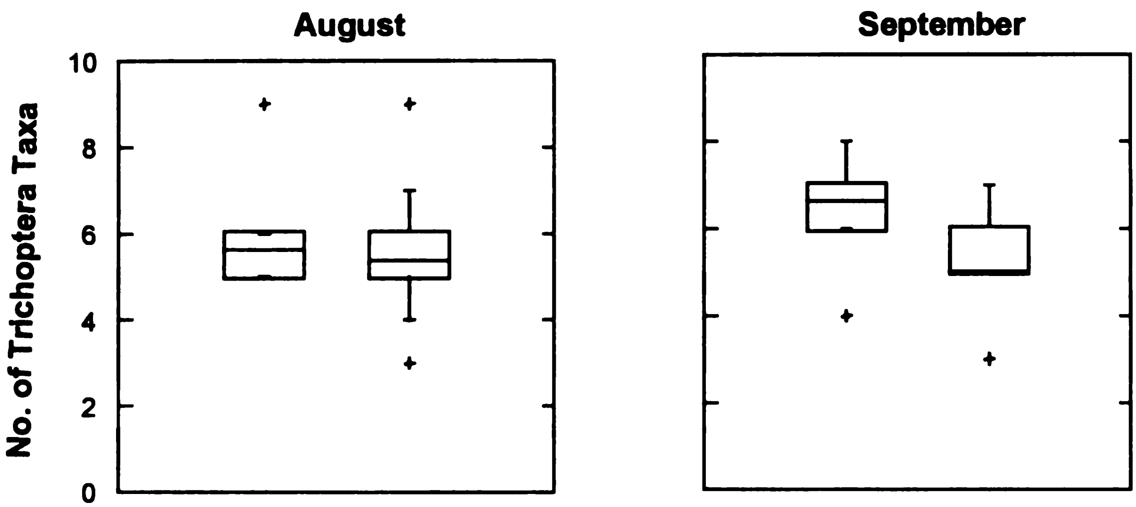

Figure 31. A comparison of candidate metrics of short term impacts, for use in Northern Lake Huron Coastal marshes. Metrics determined from Dip net samples at Cedarville (impacted) and Mackinac (reference) marshes, Lake Huron, 1996. Range bars show maximum and minimum of non-outliers; Solid lines inside the box are medians; boxes are interquartile ranges (25%ile to 75%ile); dots are outliers. (Taken from Barbour et al. 1996). *Mann-Whitney U test: Significant at $\alpha = 0.05$.

However, in September this metric received a sensitivity value of 2 (Tables 19 and 20, Figure 31). Although this metric was inconsistent between months, it did determine differences in these particular coastal marshes during the period of wastewater discharge. Therefore, it may be a valuable metric for detection of short term impacts in these Northern Lake Huron coastal marshes.

Three additional metrics demonstrated potential for use as indicators of water quality in northern Lake Huron coastal marshes (Table 21). However, these metrics consistently showed the opposite effects of impact than predicted (Tables 17-20). These included the proportions of individuals as scrapers, and the ecosystem attribute metrics of Habitat Stability, and Top Down Control. These metrics may prove useful when investigated further but were not recommended as candidate metrics based on the 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 Coefficient of Community similarity measures the degree of similarity between the reference site and the impacted site. A value close to 1 indicates very similar assemblage in terms of taxon present or absent. The Coefficient of Community loss index measures the loss of benthic taxa between the reference site and the site of comparison. 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 compared with August values.

Discussion

Water chemistry (Table A-3, Figures 2-6) and Land use (Table 1) differences between the reference (Mackinac Bay) and the impacted (Cedarville Bay) marshes indicated that

Table 21. 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
Herbivore/Detritivore as surrogate for P/R

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

impacts on the Cedarville Bay marsh were likely to be moderate. Thus, metrics for detecting water quality differences in the wetland had to be quite sensitive to detect differences. Of the 38 potential metrics tested, six metrics (16%) consistently demonstrated that differences did exist in August, two months after the previous release of treated domestic wastewater from the treatment lagoons, and in September, during wastewater release from these lagoons (Tables 19 and 20, Figures 30). An additional metric (Figure 31) was potentially useful but differences were not as great as they were for the first six. Three metrics gave consistent results that were opposite the differences predicted on the basis of metrics designed for streams (Karr and Kerans 1992) (Table 21). Thus, ten or 26% of metrics tested have potential as indicators of coastal wetland integrity (Table 21). These metrics will need to be tested for a variety of coastal wetlands before being widely accepted. The remaining 28 potential metrics derived from indices developed for wadable streams and rivers (Karr and Kerans 1992, Plafkin et al. 1989, Ohio EPA 1988) did not appear to be useful for Great Lakes coastal wetlands (Tables 19 and 20). However, some of these metrics may prove to be useful after testing over a wider array of wetlands. Additional work will be required before any of these 28 can be definitely removed from consideration. Some of the more promising of these potential metrics are discussed below.

Total taxa richness is considered one of the most useful indicators of water quality, in lotic systems (Ohio EPA 1987, Kerans and Karr 1995, DeShon 1995, Fore et al. 1996). However in this study, total taxa richness was not considered a useful indicator of water quality because it demonstrated high within site variability, and indicated that the impacted marsh had higher taxa richness than the reference marsh in August (Table 19). Taxa richness was predicted to decline at the impacted marsh. Taxa richness was higher at the impacted marsh in August due primarily to an increase in the number of Gastropoda and Lepidoptera species and the occurrence of four Hirudinea species. No leeches were collected at the reference site (Table 11). The metric of family taxa richness was also higher at the impacted marsh in August. During September, both the taxa and family richness metrics measured no effect between the two marshes. Based on these results, metrics based on total and family taxa richness of all macroinvertebrates do not appear to be good metrics for use in these northern Lake Huron coastal marshes. The moderate inputs of organic pollution in these marshes may be sufficient to increase the food source for some of the moderately tolerant macroinvertebrates without causing negative impacts, and may result in greater diversity. However, measures of taxa richness in systems with higher degrees of degradation, or modification of these metrics to include just taxa richness of insects and insect families may prove to be valuable metrics in these coastal marshes.

Kerans and Karr (1994), suggested that Ephemeroptera and Trichoptera taxa richness measures were excellent attributes for detecting water quality impairments in streams of the Tennessee Valley. Ephemeroptera taxa richness at the reference marsh was significantly higher ($p<0.05$) in September and August than it was at the impacted marsh, with a sensitivity rating of 3 (Tables 19 and 20). However, it was not selected for a candidate metric because of high within site variability in August (Table 19).

Fore and Karr (1996) suggested that the number of Crustacea plus Mollusca taxa was a useful measure of calcium-dependent taxa, and it is predicted to decrease with the increase in disturbance, due to the presence of sensitive Crustacea and Mollusca taxa (Karr and Kerans

1992). However in the Northern Lake Huron marshes, Crustacea and Mollusca taxa richness was significantly higher in the impacted marsh both months than in the reference marsh (Tables 19 and 20). Very few sensitive Crustacea or Mollusca were collected in either the reference or the impacted marshes, and pollution tolerant snails increased in the impacted marsh. The inability of the metric to detect the hypothesized impact may be due to the lack of sensitive stream Crustaceans in these coastal marshes and the presence of pollution tolerant snails. This metric may be a useful indicator of disturbance in these coastal marshes if additional research confirms that the proportion of Crustacea plus Mollusca increases with an increase in disturbance because of the pollution tolerant snail response.

In fact, all metrics which included Gastropoda indicated the opposite effect of impact from those that were predicted (Tables 19 and 20). The increase in abundance and diversity of Gastropoda in the impacted marsh may have been the result of the increased nutrients stimulating periphyton growth which is a major food source for the snails.

Karr and Chu (1997), suggested that, in an index of biotic integrity, metrics should include measures of sensitive and tolerant organisms. Candidate metrics from this study include the proportion of individuals of Ephemeroptera and Trichoptera as measures of sensitive groups, and Isopoda as a measure of tolerant groups (Karr and Chu 1997).

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 whose pollution tolerances are known may make them of limited use as indicators of water quality. Metrics based on total abundances and metrics based on

Chironomidae abundance of tribes including Orthocladiinae and Tanytarsini generally showed no effect in this study (Tables 19 and 20). These results and the difficulties associated with taxonomic identification agree with Kerans and Karr (1994) suggestion that Chironomidae may not be practical indicators of water quality.

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 the 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 marsh as compared to the reference marsh indicated that they also are sensitive in marshes and may be a useful indicator of water quality in northern Lake Huron coastal marshes. The metric based on the herbivore/detritivore ratio is used as a surrogate measure for gross primary production to community respiration ratio (P/R) and is used to evaluate the balance between autotrophy and heterotrophy (Merritt et al. 1996). The herbivore/detritivore ratio indicated that the impacted marsh was an autotrophic system while the reference marsh was a heterotrophic system. The autotrophic nature of the impacted marsh may be due to increased nutrients levels associated with the sewage effluent.

Metrics based on proportions of scrapers were predicted to decrease based on stream literature (Karr and Kerans 1992). However, numbers of scrapers plus piercers and proportion of individuals as scrapers consistently increased (Tables 19 and 20) rather than decreasing as predicted. In streams, these metrics are expected to decrease because of the preponderance of sensitive Ephemeroptera and Trichoptera grazers (scrapers) in streams. In northern Lake Huron marshes, the increases in scrapers were primarily the result of consistently greater Gastropoda abundances and taxa at the impacted marsh. These grazers are more tolerant of pollution than are the grazers expected to dominate streams. If these results prove to be consistent across an array of coastal wetlands, this metric might prove to be an excellent predictor of water quality.

The metric of the relative abundance of predators indicated that predators would decrease with increased disturbance, and they did (Figure 30). However, one of the measures of ecosystem attributes considered, predicted that normal top down predator control as the ratio of predators to all other functional feeding groups should be <0.15 (Merritt et al. 1996). This metric consistently showed the reference marsh to be >0.15, and the impacted marsh to be <0.15, indicating that the impacted marsh had normal top down predator control while the reference marsh did not. This metric was not selected as a candidate metric as a result. It may be that the ratio threshold calculated to be 0.15 for South Florida marshes (Merritt et al. 1996) needs to be adjusted for Great Lakes Coastal marshes. Otherwise, the use of the two predator metrics together could lead to confusion, because different conclusions can be drawn from the two closely related metrics.

The metric of suspended particulate organic matter (SPOM) as a proportion of deposited particulate organic matter (BPOM) was unable to differentiate between these two marshes, receiving a sensitivity rating of zero, however information can still be drawn from this metric. The ratio of SPOM/BPOM indicates that both these marshes are enriched in suspended particulate organic matter. This metric was originally developed for lotic systems. It may not be appropriate for use as a metric in these northern Lake Huron coastal marshes, which are expected to be net accumulators of particulate organic matter.

Karr and Chu (1997), suggested that diversity indices are often inconsistent because they respond erratically to changes in assemblages (Davis 1995, Simon and Lyons 1995). Different diversity indices may, therefore, produce a different rank order of the same series of sites, making it impossible to compare the sites' biological condition (Karr and Chu 1997). This study lends credence to these suggestions. Several problems arose with the use of diversity indices in this study. The various diversity indices demonstrated conflicting results within the same site and month (Tables 19 and 20). For example, in September, Shannon diversity indicated higher diversity at the reference marsh, while Margalef diversity indicated higher diversity at the impacted marsh (Tables 19 and 20). Different diversity indices are influenced by both number of taxa and their relative abundances, and some are more sensitive to rare taxa, others to abundant taxa. Therefore, different indices measure slightly different aspects of assemblages, and can lead to different results. Another potential problem was that the same indices were inconsistent from month to month (Tables 19 and 20). For example, evenness indicated that the invertebrate community was more evenly distributed at the reference marsh in September during wastewater discharge even though it was more evenly distributed at the impacted marsh in August. These results are opposite from predictions and explanations of why this might be so will be needed if these observations hold up across an array of marshes.

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 northern Lake Huron coastal marshes, but values obtained from just a two marsh comparison 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 data from a larger array of sites.

SUMMARY AND CONCLUSIONS

The purpose of this study was to evaluate the environmental quality of two northern Lake Huron coastal marshes and recommend metrics for use in a multimetric index of ecological integrity for these and similar marshes. The macroinvertebrate fauna and basic limnological characteristics were examined to determine if impairment existed. A comparison of the environmental quality as reflected by the macroinvertebrate fauna suggested that Cedarville (impacted) marsh had been moderately degraded, by sewage lagoon effluent, stormwater runoff from Cedarville, and habitat changes related to marina activities and road construction.

Chemical-physical data showed that water quality in the impacted marsh was significantly different from water quality in the reference marsh only when discharge from the wastewater treatment lagoon was actively occurring. Water quality impacts of Cedarville Bay marsh included moderate increases in chloride, NH_4-N, NO_3-N, NO_2-N, SRP, conductivity, turbidity, alkalinity and moderate decreases in levels of dissolved oxygen. These changes in water quality were greatest near the source of discharge during times of active discharge from the wastewater treatment lagoons and decreased to background levels by 260 m away from the mouth of the discharge stream. Smaller changes in water quality also occurred in June, approximately two weeks after a discharge event, suggesting that water quality in the marsh returns to background levels within two or three weeks after sewage discharge.

Although chemical analysis failed to detect any significant impairment in water quality for the months between discharge events, the macroinvertebrate fauna demonstrated moderate impairment at the impacted marsh throughout the season. Sensitive taxa such as Ephemeroptera and Trichoptera had lower abundances and lower taxa richness in both the sediment and plant community at the impacted marsh than at the reference marsh. For example, only one species of Ephemeroptera was collected in the sediment at the impacted marsh, and it was present only in June, while up to six species comprised 10-20% of the sediment macroinvertebrate fauna at the reference marsh in June, July, August, and September. Trichoptera exhibited a similar trend with nearly twice as many individuals and taxa collected from the sediments at the reference marsh over the course of the sampling season as compared to the impacted marsh. The plant associated community of the reference marsh was dominated by a diverse array of aquatic insects including the sensitive taxa (Ephemeroptera and Trichoptera). At the impacted marsh, the aquatic insect community was predominantly Chironomidae (75-90%) with a less diverse array and number of sensitive taxa being present. The plant and sediment community at the impacted marsh also had higher abundance of pollution tolerant organisms such as Gastropoda, Isopoda, Amphipoda, Chironomidae and Oligochaeta and fewer insects than were present in the reference marsh. Impairment at the impacted marsh appeared to be moderate because it still maintained a fairly diverse and abundant community, which included sensitive taxa even though these taxa were reduced in abundance. Differences in the macroinvertebrates community were not detected in response to a gradient from discharge at the impacted marsh. However, further degradation could potentially result in the elimination of the already reduced sensitive taxa, and the reduction in abundance and diversity of some of the more moderately pollution tolerant organisms.

Several potential macroinvertebrate metrics derived from the stream literature were tested to select metrics that were sensitive enough to detect the moderate impacts on the invertebrate community impacts. The purpose of this metric selection and testing was to recommend metrics that would be useful in development of an index of ecological integrity to monitor water quality in these and similar marshes.

Thirty-eight potential metrics tested included those that incorporated elements of the macroinvertebrate community structure and composition. Six metrics consistently detected predicted changes in the invertebrate community and are strongly recommended for use in development of a multimetric index of ecological integrity. An additional four metrics were identified that show some promise.

The six consistent, strongly recommended candidate metrics of long term impacts included: (1) the proportion of individuals as Ephemeroptera, (2) the proportion of individuals as Trichoptera, (3) the proportion of individuals as Isopoda, (4) the proportion of individuals as filterers, (5) the proportion of individuals as predators, and (6) the ratio of herbivores/detritivores as a surrogate for P/R. An additional metric, number of Trichoptera taxa was less consistent in detecting impacts but is recommended. Three additional metrics gave consistent results that were opposite from those predicted for streams. If this opposite effect proves to be a consistent effect, across a wide array of coastal marshes, these three metrics will also be useful in development of an index of ecological integrity. These three metrics are: (1) the proportion of individuals as scrapers, (2) Habitat stability, and (3) Top Down.

Development of an index of ecological integrity for biomonitoring through the use of multimetric indices based on invertebrates appears to be feasible for northern Lake Huron coastal marshes. Ten potential metrics are suggested from this study. These ten metrics need to be tested and validated across a range, or gradient, of human influence for Great Lake coastal wetlands before they can be widely adopted or discarded. The proposed metrics may need to be modified for particular ecoregions. The incorporation of additional metrics based on attributes of several other species groups (e.g. fish, algae, macrophytes) offers an additional means of strengthening this proposed index of ecological integrity for these wetlands. Thus, testing and validation of the recommended ten metrics and addition of metrics based on other taxonomic groups are recommended next steps in development of an Index of Ecological Integrity for Great Lakes coastal wetlands.

APPENDICES

Table A-1. Operational taxonomic unit (OTU) and identification references for invertebrates collected from Cedarville and Mackinac marsh, Lake Huron, Michigan.

INVERTEBRATE	OTU's	REFERENCES
INSECTA		
Coleoptera	Genus	White & Brigham 1996
Collembola*	Species	Snider 1967; Christiansen 1996
Diptera	Family	Teskey 1984
Chironomidae*	Tribe	Coffman & Ferrington 1996
Ephemeroptera*	Species	Edmunds & Waltz 1996; Provonsha 1990
Hemiptera*	Genus/species	Polhemus 1996
Lepidoptera	Genus	Lange 1996
Neuroptera	Species	Evans & Neunzig 1996
Odonata	Genus/species	Westfall and Tennessen 1996
Trichoptera*	Genus/species	Morse & Holzenthal 1996; Wiggins 1995
ANNELIDA		
Hirudinea*	Genus/species	Pennak 1978
Oligochaeta	Family/genus	Pennak 1978
Polychaeta	Genus	Pennak 1978
Arachnoidea (Mites)	------	-----
CRUSTACEA		
Amphipoda	Species	Pennak 1978
Isopoda	Species	Pennak 1978
Cladocera*	Order/Species	Balcer et al. 1984
Copepoda*		Balcer et al. 1984
Calanoida/Cyclopoida*	Species	Balcer et al. 1984
Harpactacoida*	Species	Balcer et al. 1984
Ostracoda	Subclass	Pennak 1978
Decapoda	Species	
PELECYPODA	Genus	Pennak 1978
MOLLUSCA		
GASTROPODA*	Species	Pennak 1978
NEMATODA	Phylum	Pennak 1978

Table A-1 (cont'd).

INVERTEBRATE	OTU's	REFERENCES
TURBELLARIA	Species	Pennak 1978

*Dr. Brian Armitage of Ohio Biological Survey, confirmed the identification of the Trichoptera. Identification of the Ephemeroptera, *Caenis amica*, *C. youngi* and *C. latipennis*, was based on a reference collection identified by Robert Waltz, Indiana Department of Natural Resources. Chironomidae larvae were identified to genus or species groups by Patrick Hudson of the Great Lakes Science Center, Biological Resources Division, U.S. Geological Survey. Dr . Richard Snider of the Department of Zoology, Michigan State University confirmed the identification of the Collembola. Ethan Nedeau, Department of Entomology at Michigan State University identified the Hirudinea. Edward Rosemen, Department of Fisheries and Wildlife identified the zooplankton species. Brain Keas of the Department of Zoology at Michigan State University confirmed the identification of the Gastropoda.

Table A-2. Chironomidae present in the plant associated Dip-net samples at Mackinac (reference) and Cedarville (impacted) marsh, Lake Huron, Michigan June 1996.

CHIRONOMIDAE*	MACKINAC	CEDARVILLE
<u>Chironominae</u>		
Cladotanytarsus spp.	4	--
Dicrotendipes spp.	--	5
Hyporhygma spp.	--	1
Microchironomus spp.	--	1
Microtendipes spp.	11	--
Pagastiella spp.	1	--
Paratanytarsus spp.	33	13
Paratendipes spp.	19	--
Polypedilum simulans group	8	--
Polypedilum tritum	--	1
Tanytarsus spp.	16	--
<u>Orthocladiinae</u>		
Acricotopus spp.	--	2
Corynoneura spp.	9	4
Cricotopus bicinctus group.	--	2
Cricotopus laricomalis group	1	--
Cricotopus spp.	1	--
Cricotopus sylvestris group	--	53
Cricotopus trifascia	11	--
Doncricotopus spp.	12	--
Nanocladius spp.	--	1
Paracricotopus spp.	--	12
Psectrocladius spp.	3	4
Thienemanniella similis	3	--
<u>Tanypodinae</u>		
Ablabesmyia peleensis		5
Ablabesmyia spp.	1	--
Conchapelopia spp.	1	--
Procladius spp.	2	--
Psectrotanypus spp.	1	--

* Identification references: Simpson and Bode 1980, and Wiederholm 1983.

Table A-3. Water quality data for Cedarville and Mackinac marsh, Lake Huron, 1996. DFD = distance from discharge in meters. *Instrument failure or samples were lost.

Alkalinity (mg $CaCO_3$/L)

	Mackinac				Cedarville			
DFD	20-June	24-July	19-Aug.	29-Sept.	21-June	25-July	18-Aug.	28-Sept.
10	184	157	200	139	125	110	109	136
60	184	169	206	148	85	100	109	143
110	181	167	204	145	80	90	104	138
160	180	160	205	146	77	93	79	132
210	134	143	200	149	78	81	59	100
260	127	136	179	150	79	77	61	97
310	123	131	150	132	*	76	89	105
360	125	119	120	121	82	74	83	87
410	127	121	115	114	86	73	51	85

NH_4-N (mg N/L)

	Mackinac				Cedarville			
DFD	20-June	24-July	19-Aug.	29-Sept.	21-June	25-July	18-Aug.	28-Sept.
10	.02	.05	.03	.03	.03	.09	.04	.43
60	.01	.04	.02	.02	.06	.08	.07	.48
110	.02	.05	.02	.24	.05	.08	.06	.38
160	.04	.06	<.01	.09	.09	.06	.06	.18
210	.01	.03	.04	.03	.09	.07	.05	.06
260	.04	.04	.06	.04	.06	.05	.05	.05
310	.04	.03	.04	.14	.15	.05	.05	.05
360	.03	.04	.02	.03	.06	.05	.04	.07
410	.05	.03	.02	.05	.23	.09	.07	.21

Chloride (mg Cl/L)

	Mackinac				Cedarville			
DFD	20-June	24-July	19-Aug.	29-Sept.	21-June	25-July	18-Aug.	28-Sept.
10	17	14	14	16	31	4	20	48
60	14	15	16	18	18	4	18	71
110	15	16	15	17	7	12	21	81
160	18	16	13	17	13	21	20	28
210	16	13	15	16	12	16	17	28
260	15	14	16	21	5	18	21	24
310	15	13	14	18	5	18	19	22
360	18	15	14	17	4	11	19	21
410	15	12	14	16	4	15	18	22

Table A-3 (cont'd)

Conductivity (uS/cm)

	Mackinac				Cedarville			
DFD	20-June	24-July	19-Aug.	29-Sept.	21-June	25-July	18-Aug.	28-Sept.
10	386	333	414	300	288	342	271	399
60	409	351	431	302	225	259	251	411
110	390	348	439	299	168	235	250	425
160	386	341	430	300	179	242	237	394
210	305	320	365	305	187	213	531	251
260	305	296	404	310	193	222	622	272
310	306	283	404	822	214	214	510	260
360	302	274	380	270	190	193	215	214
410	286	268	321	259	220	210	212	215

% Dissolved Oxygen

	Mackinac				Cedarville			
DFD	20-June	24-July	19-Aug.	29-Sept.	21-June	25-July	18-Aug.	28-Sept.
10	106	92	100	59	106	78	90	44
60	94	76	104	58	110	93	102	37
110	95	82	84	56	101	95	91	47
160	107	90	104	55	112	82	104	50
210	101	91	114	56	112	93	74	69
260	91	90	103	58	103	83	73	76
310	88	103	96	69	104	80	82	72
360	86	102	95	72	105	87	84	76
410	81	102	103	67	97	82	61	72

NO_3-N (mg N/L)

	Mackinac				Cedarville			
DFD	20-June	24-July	19-Aug.	29-Sept.	21-June	25-July	18-Aug.	28-Sept.
10	<.01	<.01	<.01	<.01	<.01	<.01	<.01	.30
60	<.01	<.01	<.01	<.01	<.01	<.01	<.01	.32
110	<.01	<.01	<.01	.03	.02	<.01	<.01	.26
160	<.01	<.01	<.01	<.01	.03	<.01	<.01	.18
210	<.01	<.01	<.01	<.01	.03	<.01	<.01	.01
260	<.01	<.01	<.01	<.01	<.01	<.01	<.01	<.01
310	<.01	<.01	<.01	.02	<.01	<.01	<.01	<.01
360	<.01	<.01	<.01	.02	.02	<.01	<.01	<.01
410	<.01	<.01	<.01	.04	.01	<.01	<.01	<.01

Table A-3 (cont'd)

pH

DFD	Mackinac 20-June	24-July	19-Aug.	29-Sept.	Cedarville 21-June	25-July	18-Aug.	28-Sept.
10	*	7.43	7.69	8.46	*	8.33	7.85	8.00
60	*	7.44	7.69	8.74	*	8.73	*	9.00
110	*	7.47	7.48	8.49	*	9.29	7.92	6.60
160	*	7.66	7.65	9.04	*	8.55	7.9	8.00
210	*	7.58	7.84	8.82	*	8.86	8.77	8.60
260	*	7.58	7.73	8.90	*	8.37	6.10	6.60
310	*	7.62	7.73	8.97	*	7.30	7.47	7.60
360	*	7.60	7.73	8.86	*	7.80	7.13	5.80
410	*	6.95	8.03	8.75	*	8.08	7.42	6.70

SRP (mg P/L)

DFD	Mackinac 20-June	24-July	19-Aug.	29-Sept.	Cedarville 21-June	25-July	18-Aug.	28-Sept.
10	<.01	<.01	<.01	<.01	.01	<.01	<.01	.21
60	<.01	<.01	<.01	<.01	<.01	<.01	<.01	.19
110	<.01	<.01	<.01	<.01	<.01	<.01	<.01	.13
160	<.01	<.01	<.01	<.01	<.01	<.01	<.01	.06
210	<.01	<.01	<.01	<.01	<.01	<.01	<.01	<.01
260	<.01	<.01	<.01	<.01	<.01	<.01	<.01	<.01
310	<.01	<.01	<.01	<.01	.01	<.01	<.01	<.01
360	<.01	<.01	<.01	<.01	<.01	<.01	<.01	<.01
410	<.01	<.01	<.01	<.01	<.01	<.01	<.01	<.01

Turbidity (NTU's)

DFD	Mackinac 20-June	24-July	19-Aug.	29-Sept.	Cedarville 21-June	25-July	18-Aug.	28-Sept.
10	1.4	2.4	.64	1.2	4.5	1.8	1.1	4.5
60	1.2	1.0	.56	1.0	5.0	1.6	1.1	1.3
110	1.5	3.1	.64	1.0	4.3	1.0	1.0	1.5
160	2.0	0.9	.65	1.0	4.0	2.2	2.0	1.8
210	3.5	1.0	.70	1.0	3.8	1.2	1.0	1.0
260	2.57	3.9	.46	3.7	3.6	1.1	.95	1.0
310	3.2	2.0	1.0	4.5	4.3	1.7	.76	1.0
360	2.2	2.4	1.0	5.2	3.5	1.0	.83	1.0
410	2.9	3.0	1.0	1.0	3.8	1.0	1.0	1.0

Water temperature (° Celsius)

DFD	Mackinac 20-June	24-July	19-Aug.	29-Sept.	Cedarville 21-June	25-July	18-Aug.	28-Sept.
10	22	19	23	11	19	12	20	12
60	23	19	22	12	23	22	22	12
110	23	20	22	12	20	22	22	13
160	23	21	22	12	20	22	29	13
210	23	21	23	12	21	22	26	14
260	21	21	23	13	20	23	24	13
310	21	21	23	13	21	22	24	13
360	21	20	23	13	20	22	24	14
410	21	21	23	13	20	23	23	14

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