Assessment of Benthic Invertebrate Populations in the Muskegon Lake Area of Concern

Prepared By

Richard R. Rediske, Laurie Beth Nederveld, Ying Hong, and Kate Rieger

Annis Water Resources Institute
Grand Valley State University
740 West Shoreline Drive
Muskegon, Michigan 49441

Neil W. MacDonald and James P. Dunn

Grand Valley State University
Department of Biology
1 Campus Drive
Allendale, Michigan 49401

Donald G. Uzarski

Central Michigan University
Department of Biology
Mount Pleasant, Michigan 48859

Prepared For

Michigan Department of Environmental Quality
481047-06

January 2009
MR-2009-1

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

Acknowledgements

We thank the Michigan Department of Environmental Quality for awarding AWRI a Clean Michigan Initiative - Clean Water Fund grant (TC: 481047-06), which funded this research and for the contributions of Matt Wesener as project Officer.

For the part of this project that involved Muskegon Lake, we thank the AWRI staff that assisted with this project. Brian Scull assisted with the sample collection and Jen Gradisher, Catharina Kramer, and Anthony Straley were involved with the preparation of the benthic invertebrates for analysis. We also than the NOAA Lake Michigan Field Station for providing the elutriation system and Dr. Tom Nalepa and Glenn Carter for their assistance with the identification of benthic macroinvertebrates and providing reference slides for the 1999 study for comparison.

For the Ruddiman Creek part of this project, we thank Matthew Cooper, Brent Kasza, Michelle Lelli, Daniel Mays, Katherine Rieger, Brian Scull, and Anthony Straley who assisted with field sampling. For use of their field equipment and other logistical support, we thank the Annis Water Resources Institute and Fishbeck, Thompson, Carr & Huber, Inc. In addition, we are obliged to James Janesak of Earth Tech for providing data on stream remediation activities, Dr. Phyllis Curtiss and Dr. Sango Otieno for assisting with the statistical analyses, and Dr. Robert Hoeksema for assisting with the hydrologic assessment.

Executive Summary

Benthic macroinvertebrate populations in the Muskegon Lake Area of Concern (AOC) were evaluated in support of delisting the Degradation of Benthos Beneficial Use Impairment (BUI) in Muskegon Lake and to evaluate the recovery of Ruddiman Creek after remediation. For Muskegon Lake, benthic macroinvertebrates were collected and analyzed at 15 locations in 2006 and compared to data collected in 1999. A significant increase in chironomids and decrease in oligochaetes was observed in 2006, indicating that the benthic invertebrate community in Muskegon Lake continues to improve from 1999 conditions. Shannon Weaver diversity, total benthic organisms, and the chironomid trophic index were not significantly different between years, suggesting stable benthic conditions. All of the metrics either indicated stable or improving conditions in the benthic macroinvertebrate community.

The Muskegon Lake Public Advisory Council (PAC) established five numerical targets to delist the Degradation of Benthos BUI. The first target required the removal of areas where the sediment is toxic to aquatic organisms. Currently, the only known area that exceeds the target is the Division Street Outfall and this location is currently being evaluated for remedial dredging with a feasibility study. The remaining four targets required two years of monitoring data at 5 year intervals for delisting. Using the information from this study and the data collected in 1999, the benthic community met the delisting targets. *Hexagenia spp.* was present in the littoral zone near the mouth of the Muskegon River in 2000 and at similar locations during this study. The remaining three targets for oligochaetes, chironomids, and diversity also were met using the 1999 and 2006 data. The % oligochaeta (without zebra and quagga mussels) was below 75% as the 1999 and 2006 means were 69% and 45%, respectively. The target for chironomids required this group to have an abundance $\geq 500/m^2$ and the 1999 and 2006 means were $677/m^2$ and $1209/m^2$, respectively. With respect to diversity, the target required a value of >1.66 and the results for 1999 and 2006 were 1.88 and 2.08, respectively. When the sediments at the Division Street Outfall are successfully remediated, the AOC should be able to delist the Degradation of Benthos BUI for Muskegon Lake.

The impact of sediment remediation on the composition, relative abundance, and diversity of the macroinvertebrate community inhabiting Ruddiman Creek (Muskegon Lake AOC) was evaluated in the second part of this investigation. Macroinvertebrate samples from all available habitat types at three study sites and three reference sites were collected using a Before-After, Control-Impacted (BACI) approach. Ryerson Creek, considered less disturbed with respect to heavy metal and organic chemical contaminants, served as an urbanized reference stream within the Muskegon Lake watershed. Samples were collected three months before the dredging and removal of contaminated sediment and four times over a span of 1.5 years after restoration activities were completed in Ruddiman Creek. In addition to macroinvertebrate collections, physical measurements, chemical analyses of water samples, and hydrologic measurements in Ruddiman and Ryerson Creeks were used to assess habitat changes as a result of remediation activities. The macroinvertebrate community in Ruddiman Creek was reduced in both abundance and diversity three months following sediment removal, but over one year after remediation, the abundance and diversity of Ruddiman Creek's macroinvertebrate community had returned to levels comparable to pre-remediation conditions. The Family Biotic Index (FBI) suggested some improvement in the overall condition of the two upstream sites on Ruddiman Creek, while the

most heavily remediated downstream site remained in a degraded state. Stream quality FBI rankings in the fairly poor category throughout the project suggested that hydrologic impairments continue to negatively influence the macroinvertebrate community after remediation and additional restoration activities are needed to improve the ecological integrity of the Ruddiman Creek watershed.

1.0 Introduction

Muskegon Lake is a 16.8 km^2 drowned river mouth lake located in western Michigan. The lake was listed as an Area of Concern (AOC) by the International Joint Commission (IJC) because of severe environmental impairments related to the historic discharge of municipal and industrial wastes. The 1987 Remedial Action Plan listed five Beneficial Use Impairments (BUIs), including Degradation of Benthos (Wuycheck 1987). The inclusion of this BUI was supported by studies of benthic communities and associated sediments that indicated a severely degraded benthic fauna along with high levels of sediment contaminants (Evans 1976; Peterson 1951; Surber 1954). Data from a 1972 survey (Evans 1976) showed that pollution tolerant oligochaete worms comprised 89% of the total benthic population, chironomid numbers were low (< 200/m^2), and species diversity (Shannon Weaver) was only 0.68. In 1974, the direct discharge of municipal and industrial wastewater to Muskegon Lake was eliminated by the construction of an advanced tertiary treatment facility. Later, industrial pretreatment programs, hazardous waste site remediation projects, and numerous conservation and nonpoint source reduction efforts resulted in large improvements in water quality (Rediske *et al.* 2002; Carter *et al.* 2006; Steinman *et al.* 2008). By 1999, Shannon Weaver diversity improved to 1.66, oligochaetes were reduced to 68% of the total population, and chironomid numbers increased to over 600/m^2 (Carter *et al.* 2006).

Impacted benthic communities also are present in Ruddiman Creek, a tributary of Muskegon Lake and part of the AOC. The creek is included on the Michigan 303(d) list (MDEQ 2004) for poor benthic invertebrate populations and is currently undergoing remediation/restoration under the Great Lakes Legacy Act program because of highly contaminated sediment (EPA 2005). In consideration of the improvements to the benthic community in the lake and the remediation/restoration of Ruddiman Creek, the Muskegon Lake Public Advisory Council (PAC) voted in 2005 to establish numerical criteria to delist the Degradation of Benthos BUI. The targets are summarized below:

> This BUI will be considered restored when average benthic macroinvertebrate populations in Muskegon Lake reflect the following conditions (two sampling periods in 5 years):

Indicator	Target
Sediment Toxicity	**Amphipod Survival >60%**
Hexagenia	**Present in river mouth littoral zone**
% Oligochaeta	**< 75%**
Chironomidae (#/m^2)	**> 500**
Diversity (Shannon Weaver)	**> 1.5**

> In addition, benthic populations in Ruddiman Creek must show an improvement from pre-remediation conditions with respect to trophic status and an increasing trend in species diversity.

The last benthic survey of Muskegon Lake was conducted in 1999 (Rediske et al 2002; Carter *et al.* 2006). The results of the 1999 investigation indicated that macroinvertebrate populations were influenced to a greater extent by organic/nutrient loadings from the Muskegon River than concentrations of heavy metals and/or organic chemicals in the sediment. Most environmental study methodologies utilize reference (control) systems as a benchmark to gauge the degree of pollution impact or level of ecosystem recovery. The depth variations and widespread nature of historical pollution in Muskegon Lake make the selection of control sites within the lake very difficult. Similar drowned river mouth lakes (White Lake, Lake Macatawa, and Mona Lake) also have experienced varying degrees of anthropogenic contaminant inputs and, with the exception of White Lake, have shallower bathymetry. In consideration of the difficulties inherent in the selection of a suitable reference system, the delisting targets for Muskegon Lake were established to reflect benthic communities consistent with the current level of nutrient/organic enrichment from the Muskegon River and the desire to sustain the trend of increasing species diversity. Pre and post remediation monitoring of the benthic community in Ruddiman Creek has not been performed. We conducted a monitoring program of the benthic macroinvertebrate communities in Muskegon Lake and Ruddiman Creek that will provide sufficient information for the PAC to determine if the numerical targets have been met and pending a favorable outcome, prepare a request to the MDEQ for delisting the Degradation of Benthos BUI.

2.0 Project and Task Description

The main project goal was to conduct a benthic invertebrate monitoring program in Muskegon Lake and Ruddiman Creek that supported the PAC's efforts to delist the Degradation of Benthos BUI. The secondary goal was to communicate this information in an understandable manner to decision makers and stakeholder groups associated with the AOC. The sampling locations we evaluated are shown in Figure 1. Fifteen locations in Muskegon Lake were sampled in triplicate during the fall of 2006 for benthic macroinvertebrates using a petite Ponar. These locations corresponded to the stations used in 1999 (Rediske *et al.* 2002; Carter *et al.* 2006) and also provided overlap with the investigation conducted in 1972 (Evans 1976). Benthic macroinvertebrates were enumerated and identified to genus/species level and analyzed by species diversity and trophic status metrics used previously (Rediske *et al.* 2002; Carter *et al.* 2006). Sediment chemistry and toxicity were extensively examined at these locations in 1999. Due to the funding limitations of this grant program, we did not collect data on sediment chemistry and toxicity. Due to the absence of significant anthropogenic inputs of toxic chemicals and the stability of sediments in large lakes, we assumed that the sediments have similar quality and characteristics as in 1999. If significant changes in species composition or diversity were observed at any of the locations, an additional evaluation of sediment chemistry and toxicity was required to determine the causative factors.

The benthic survey of Ruddiman Creek was conducted using upper Ryerson Creek as a reference site. The upper part of Ryerson Creek has similar land use and flow conditions to Ruddiman Creek, and has been impacted only by urban stormwater. During the summer of 2006 and 2007, invertebrates were sampled from the locations on Ruddiman Creek and Ryerson Creek shown in Figure 1.1. Triplicate macroinvertebrate samples were collected from representative vegetation types and

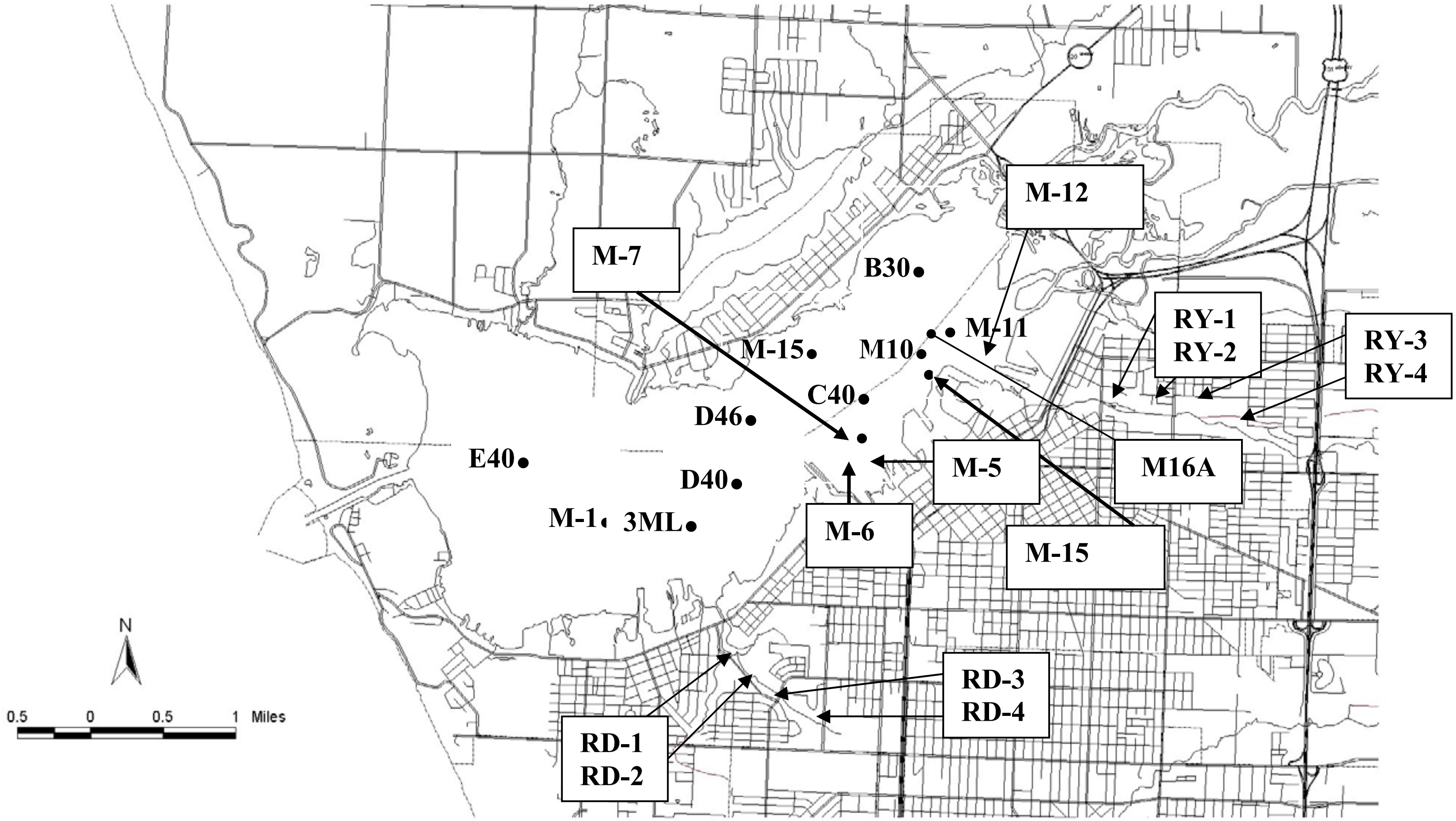


Figure 1. 1. Locations selected for Benthic Macroinvertebrate Monitoring in the Muskegon Lake AOC.

analyzed according to methods described by Uzarski *et al.* (2004). Samples were collected from the same locations in the summer of 2005, prior to remediation, and preserved in ethanol. We analyzed these samples, in addition to the 2006 and 2007 collections, to assemble a data set that includes macroinvertebrate populations indicative of pre and post remediation conditions.

The use of standardized collection protocols, historical data sites, and triplicate samples provided a high level of quality assurance and resulted in data that can be used for decision making and environmental assessment. Project tasks are described below:

Task 1: Sampling and Analysis of Benthic Macroinvertebrate Samples from Muskegon Lake

- AWRI collected triplicate samples at each site shown in Figure 1 with a petite Ponar. Samples were washed into a large tub and then into an elutriation device with a 0.5-mm, nitex-mesh sleeve to remove silt and other fine particles. Retained material was preserved in 10% buffered formaldehyde with rose bengal stain. All organisms were identified to the lowest practical taxonomic level.

- Chemical and physical parameters were monitored at each study site. A Hydrolab DataSonde 4a was used to measure pH, redox potential, chlorophyll *a*, DO, DO% saturation, temperature, total dissolved solids, turbidity and specific conductance as vertical profiles. Grab samples were analyzed for soluble reactive phosphorus, nitrate-N, and ammonia-N at the Annis Water Resources Institute laboratory. Analytical procedures and quality assurance/control followed recommended procedures outlined in Standard Methods for the Examination of Water and Wastewater (APHA, 1998).

- Macroinvertebrate populations were analyzed by standard statistical methods and ecological metrics as described by Rediske *et al.* (2002) and Carter *et al.* (2006).

Task 2: Sampling and Analysis of Benthic Macroinvertebrate Samples from Ruddiman Creek and Ryerson Creek

- During the summer of 2006 and 2007, invertebrates were sampled from Ruddiman Creek and Ryerson Creek at the locations shown on Figure 1. Aquatic invertebrates were collected from each sample site using 0.5 mm mesh D-frame dip nets. To ensure sampling of all microhabitats, dip net sampling involved a sweep of the surface, mid-depth, and just above the streambed. Rocks over 2 cm in diameter were washed in dip nets to dislodge macroinvertebrates. Three replicate samples were collected for each habitat type to obtain a measure of variance. Habitat types consisted of those associated with the following vegetation types: *Typha*, wetland grasses, and submergents. For stream macroinvertebrates, chironomids and oligochaetes were identified to family and/or tribe level. In addition to the surveys in 2006 and 2007, archived samples collected from the same locations were identified and enumerated.

- Chemical and physical parameters were monitored at each study site during sample events. A Hydrolab DataSonde 4a was used to determine pH, redox potential, chlorophyll *a*, DO, DO% saturation, temperature, total dissolved solids, turbidity and specific conductance. Grab samples were analyzed for soluble reactive phosphorus, nitrate-N, ammonia-N, total kjeldahl nitrogen (TKN-N), and total phosphorus (TP-P), at the Annis Water Resources Institute laboratory. Physical characteristics of each site were determined including substrate composition, vegetative cover, and stream discharge. Stream discharge measurements were collected using a Marsh-McBirney Flo-Mate Model 2000 Portable Flowmeter. Analytical procedures and quality assurance/control followed recommended procedures outlined in Standard Methods for the Examination of Water and Wastewater (APHA 1998).

- Macroinvertebrate populations were analyzed by standard statistical methods and ecological metrics as described by Uzarski *et al.* (2004).

This report is organized into two chapters. Chapter I contains the methods, results, and discussion of the Muskegon Lake benthos investigation. Chapter II contains the methods, results, and discussion of the post remediation investigation of Ruddiman Creek.

3.0 References

APHA (American Public Health Association). 1998. Standard Methods for the Evaluation of Water and Wastewater. 20th edition. APHA, Washington, DC.

Carter, G. S., T. F. Nalepa, and R. R. Rediske. 2006. Status and trends of benthic populations in a coastal drowned river mouth lake of Lake Michigan. Journal of Great Lakes Research 32:578-595.

EPA 2005. Ruddiman Creek/Pond Dredging Project Will Remove Contaminants. U.S. Environmental Protection Agency. Region V. Chicago. Il.

Evans E. 1976. Final Report of the Michigan Bureau of Water Management's investigation of the sediments and benthic communities of Mona, White, and Muskegon Lakes, Muskegon County, Michigan, 1972.

MDEQ (Michigan Department of Environmental Quality). 2004. Water Quality and Pollution Control in Michigan: 2004 Sections 303(d) and 305(b) Integrated Report. Michigan Department of Environmental Quality, Lansing, MI.

Peterson K. 1951. A Biological Survey of Muskegon Lake. Michigan Department of Natural Resources, Institute for Fisheries Research, 1276.

Rediske R., Thompson C., Schelske C., Gabrosek J., Nalepa T. F., and Peaslee G. 2002. Preliminary investigation of the extent of sediment contamination in Muskegon Lake, MI. U. S.

Environmental Protection Agency, Great Lakes National Program Office, Chicago IL. GL-97520701-01

Steinman, A., M. Ogdahl, R. Rediske, C. Ruetz, B. Biddanda and L. Nemeth. (2008): Current status and trends in Muskegon Lake, Michigan. J. Great Lakes Research 34: 169-188.

Surber E. 1954. Results of a Biological Survey of Muskegon Lake in the Vicinity of the Central Paper Co. Michigan Water Resources Commission.

Uzarski, D.G., Burton, T.M., and Genet, J.A. 2004. Validation and performance of an invertebrate index of biotic integrity for Lakes Huron and Michigan fringing wetlands during a period of lake level decline. *Aquat. Ecosyst. Health Manage.* 7(2): 269-288.

Wuycheck J. D. 1987. Muskegon Lake Area of Concern Remedial Action Plan. Michigan Department of Natural Resources, Lansing. MI.

CHAPTER I. ASSESSMENT OF BENTHIC MACROINVERTEBRATE POPULATIONS IN MUSKEGON LAKE

Prepared By

Richard R. Rediske, Ying Hong, and Kate Rieger

Annis Water Resources Institute
Grand Valley State University
740 West Shoreline Drive
Muskegon, Michigan 49441

Prepared For

Michigan Department of Environmental Quality
Tracking Code: 481047-06

January 2009

Executive Summary

Benthic macroinvertebrate populations in the Muskegon Lake Area of Concern (AOC) were evaluated in support of delisting the Degradation of Benthos Beneficial Use Impairment (BUI). Benthic macroinvertebrates were collected and analyzed at 15 locations in 2006 and compared to data collected in 1999. A significant increase in chironomids and decrease in oligochaetes was observed in 2006, indicating that the benthic invertebrate community in Muskegon Lake continued to improve from 1999 conditions. Shannon Weaver diversity, total benthic organisms, and the chironomid trophic index were not significantly different between years, suggesting stable benthic conditions. All of the metrics either indicated stable or improving conditions in the benthic macroinvertebrate community.

The Muskegon Lake Public Advisory Council (PAC) established five numerical targets to delist the Degradation of Benthos BUI. The first target required the removal of areas where the sediment is toxic to aquatic organisms. Currently, the only known area that exceeds the target is the Division Street Outfall and this location is currently being evaluated for remedial dredging with a feasibility study. The remaining four targets required two years of monitoring data at 5 year intervals for delisting. Using the information from this study and the data collected in 1999, the benthic community met the delisting targets. *Hexagenia spp*. was present in the littoral zone near the mouth of the Muskegon River in 2000 and at similar locations during this study. The remaining three targets for oligochaetes, chironomids, and diversity also were met using the 1999 and 2006 data. The % oligochaeta (without zebra and quagga mussels) was below 75% as the 1999 and 2006 means were 69% and 45%, respectively. The target for chironomids required this group to have an abundance $\geq 500/m^2$, and the 1999 and 2006 means were $677/m^2$ and $1209/m^2$, respectively. With respect to diversity, the target required a value of >1.66 and the results for 1999 and 2006 were 1.88 and 2.08, respectively. When the sediments at the Division Street Outfall are successfully remediated, the AOC should be able to delist the Degradation of Benthos BUI.

I.1 Introduction

The composition of the benthic macroinvertebrate community is widely considered an effective tool for evaluating environmental (trophic) conditions. Benthic macroinvertebrates are found in most habitats and relatively easy to sample quantitatively (Wiederholm 1980; Canfield *et al.* 1996). Moreover, they form stable communities that integrate conditions of both pelagic and benthic zones over relatively long periods of time (Wiederholm 1980; Nalepa 1987). Since the benthos is confined to habitat that continually receives autochthonous and allochthonous material it also serves as an integral measure of autotrophic and heterotrophic processes in lakes (Wiederholm 1980). Species assemblages and the presence or absence of key "indicator" species reflect environmental conditions and can be used to assess a lake's trophic state (Wiederholm 1980; Milbrink 1983; Nalepa 1987). For example, the tubificid worm *Limnodrilus hoffmeisteri* is often found in high densities in areas with gross organic pollution; conversely, the mayfly *Hexagenia* is generally found in relatively pristine or less productive (oligotrophic) habitats (Howmiller and Scott 1977; Milbrink 1983; Schloesser *et al.* 1995). Benthic surveys give a "snapshot" of trophic conditions in a lake at the time samples were taken and may be a reflection of recent events. However, comparisons with historical records can be used to assess trends in environmental conditions and trophic state (Nalepa *et al.* 2000). Indicator species have been particularly effective in quantifying changes based on comparisons with historical records (Carr and Hiltunen 1965; Nalepa 1991; Krieger and Ross 1993; Harman 1997; Lang 1998). Nutrient abatement efforts implemented in the mid-1970s have been credited for increases in less tolerant taxa as well as an overall decrease in abundances of benthic macroinvertebrates in Lakes Michigan, Erie, and Ontario (Nalepa 1987, 1991; Schloesser *et al.* 1995). As seen prior to the mid-1970s, increased densities of most benthic groups and reduced densities of intolerant taxa generally reflected eutrophication resulting from increased nutrient loads (especially phosphorus). Given the difficulty of lake-wide experimental manipulations, historical comparisons may be the only practical method to assess changes resulting from human activities (Barton and Anholt 1997). Historical records provide a baseline against which more recent studies can be compared, in order to determine degree, extent, and rate of improvement (or decline) of habitat conditions in a particular lake or following a particular event (Wiederholm 1980; Nalepa 1987; Schloesser *et al.* 1995).

In this study, the abundance and species composition of the benthic macroinvertebrate community in Muskegon Lake was examined. Muskegon Lake is a 16.8 km^2 drowned river mouth lake located in western Michigan. The lake was listed as an Area of Concern (AOC) by the International Joint Commission (IJC) because of severe environmental impairments related to the historic discharge of municipal and industrial wastes. The 1987 Remedial Action Plan listed five Beneficial Use Impairments (BUIs), including Degradation of Benthos (Wuycheck 1987). The inclusion of this BUI was supported by studies of benthic communities and associated sediments that indicated a severely degraded benthic fauna along with high levels of sediment contaminants (Evans 1976; Peterson 1951; Surber 1954). Data from a 1972 survey (Evans 1976) showed that pollution tolerant oligochaete worms comprised 89% of the total benthic population, chironomid numbers were low ($< 200/m^2$), and species diversity (Shannon Weaver) was only 0.68. In 1974, the direct discharge of municipal and industrial wastewater to Muskegon Lake was eliminated by the construction of an advanced tertiary treatment facility. Later, industrial pretreatment programs, hazardous waste site remediation projects, and numerous conservation and nonpoint source reduction efforts resulted in large improvements in water quality (Rediske *et al.* 2002; Carter *et al.* 2006; Steinman *et al.* 2008). By 1999, Shannon Weaver diversity improved to 1.66, oligochaetes were reduced to 68% of the total population, and chironomid numbers increased to over $600/m^2$ (Carter *et al.* 2006).

In consideration of the improvements to the benthic community in the lake and the remediation/restoration of Ruddiman Creek, the Muskegon Lake Public Advisory Council (PAC) voted in 2005 to establish numerical criteria to delist the Degradation of Benthos BUI. The targets are summarized below:

> This BUI will be considered restored when average benthic macroinvertebrate populations in Muskegon Lake reflect the following conditions (two sampling periods in 5 years):

Indicator	Target
Sediment Toxicity	**Amphipod Survival >60%**
Hexagenia	**Present in river mouth littoral zone**
% Oligochaeta	**< 75%**
Chironomidae (#/m^2)	**> 500**
Diversity (Shannon Weaver)	**> 1.5**

The 1999 data met the above delisting criteria. The purpose of this project was to conduct the second sampling event to determine the current status of the benthic macroinvertebrate community and if the delisting criteria were met.

I.2 Methods

I.2.1 Study Area

Muskegon Lake is a large drowned river mouth lake (1,680 ha) with a well-developed and industrialized shoreline. Mean depth is 7.1 m (maximum is 21 m), water volume is about 119 million m^3, and mean hydraulic retention time is about 23 days. The lake receives 95% of its tributary inputs from the Muskegon River, which enters on the lake's east side (Figure I-1). This river is the second longest in the state (352 km) and has a watershed of 6,819 km^2. Mean annual flow into Muskegon Lake is 55.5 m^3/s^1. Lake outflow is through a navigation channel on the west side of the lake that is connected to Lake Michigan (Figure I-1).

Anthropogenic activity has affected Muskegon Lake since the early 1800s when lumber barons harvested the region's timber resources and left behind a legacy of barren riparian zones and severe erosion. Saw mills were then constructed on the shoreline, and much of the littoral zone was filled with sawdust, wood chips, timber wastes, and bark. This was followed in the 1900s by an era of industrial expansion related to heavy industry and shipping. In the 1960s and early 1970s, the lake received over

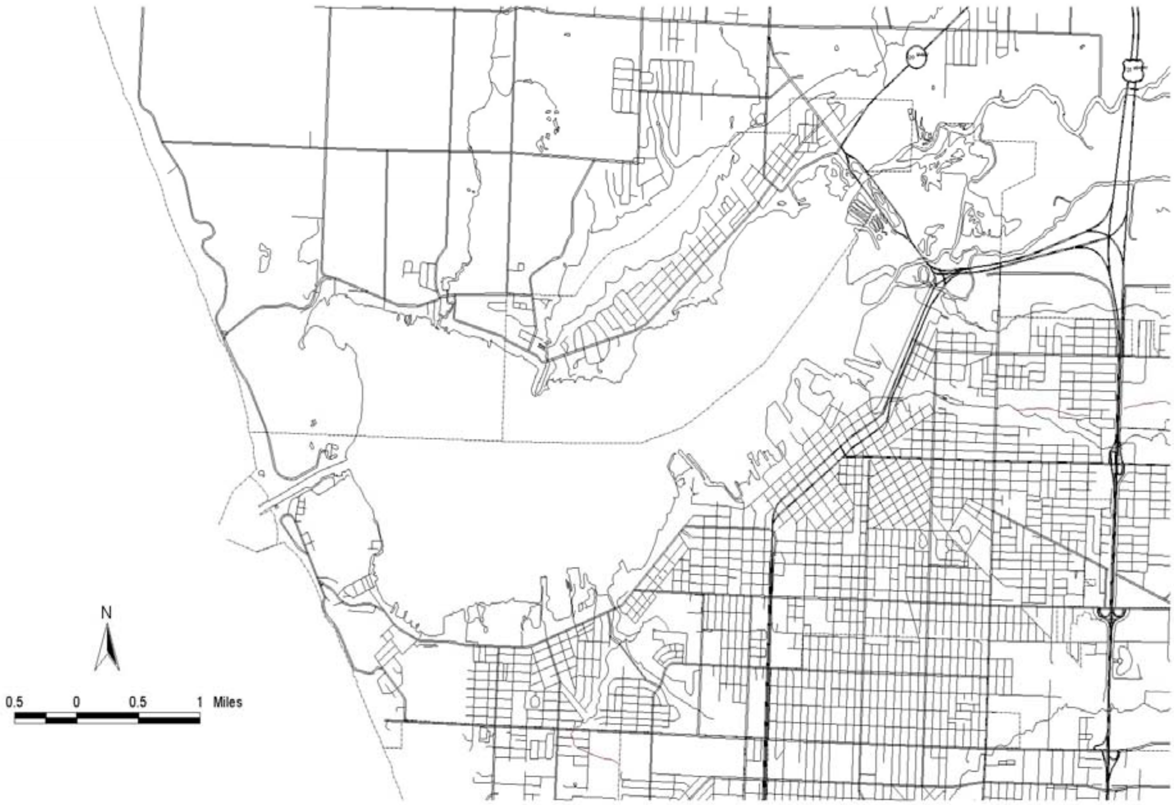


Figure I- 1. Muskegon Lake, Michigan.

100,000 m^3 of wastewater from direct discharge from industrial and municipal sources (Great Lakes Commission 2000; Evans 1992). These discharges included effluents from pulp and paper, petrochemical, organic chemical, metal finishing, and manufactured gas facilities. Wuycheck (1987) and Evans (1992) provided detailed reviews of studies that described extensive water quality problems related to nutrient enrichment, nuisance algal blooms, fish tainting, excessive macrophyte growth, contaminated sediments from the discharge of heavy metals and organic chemicals, winter fish kills, thermal pollution, oil slicks, and anoxia. A tertiary wastewater treatment facility was constructed in 1973, and the discharge was diverted to a location 25 km upstream on the Muskegon River. Persistent contaminants, however, remain in sediments from some lake areas (Evans 1992; Rediske *et al.* 2002).

I.2.2 Invertebrate Sampling and Analysis

Samples were collected on November 1, 2006 at each of the 15 stations (Figure I-2, Table I-1). These stations were located at the same coordinates as previously described (Carter *et al.* 2006) Triplicate samples were taken at each station with a petite Ponar grab (15.24cm x

15.24cm). Each sample was washed into a large tub and then washed into an elutriation device with a 0.5mm nitex mesh sleeve, to wash out silt and other fine particles. Material and organisms retained were then preserved in 10% buffered formalin, containing rose bengal stain. In addition to these 15 stations, two littoral locations near the mouth of the Muskegon River were sampled (Hex 1 and Hex 2) for the mayfly *Hexagenia*. Hex 1 was located in the macrophyte

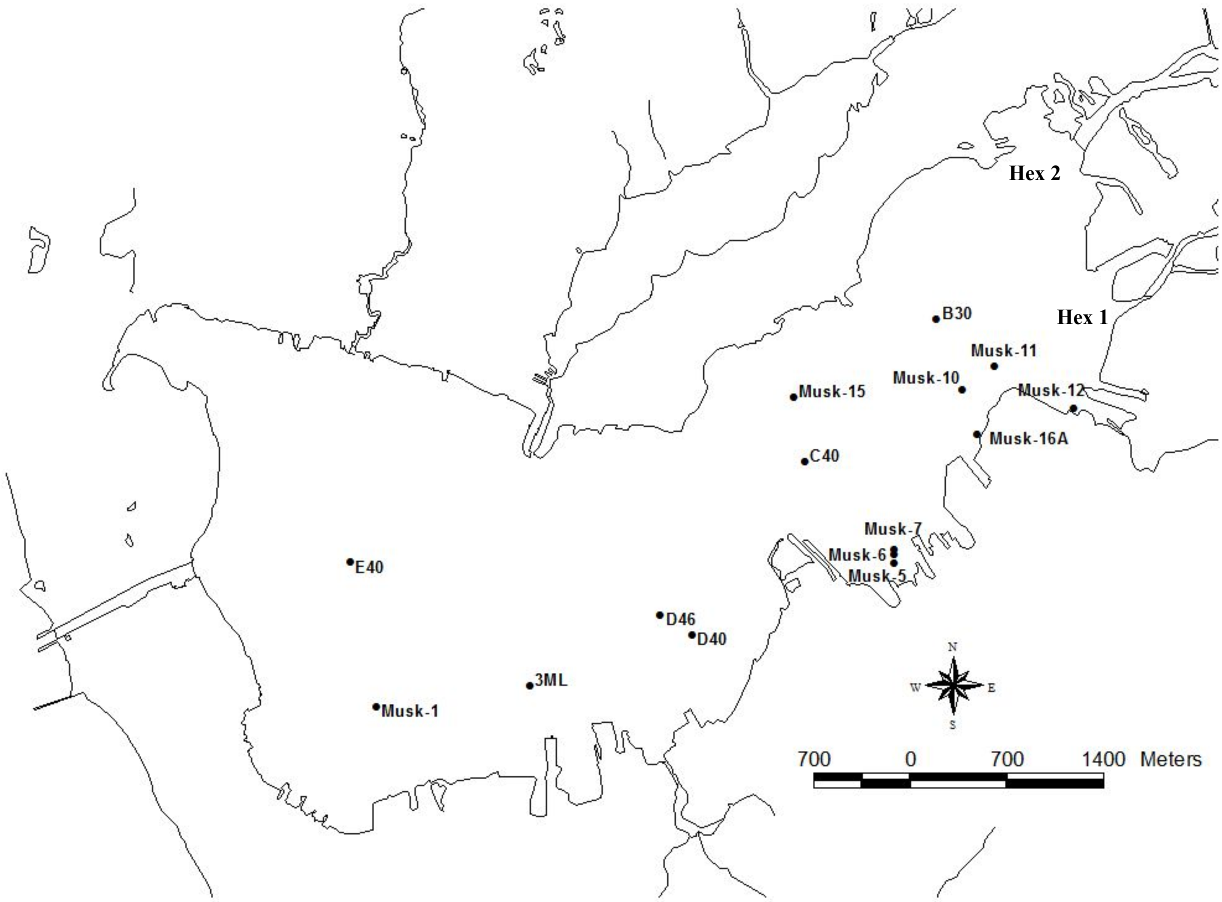


Figure I- 2. Muskegon Lake Sampling Locations (November 2006).

beds at Fisherman's Landing and Hex 2 was located in a similar environment at the Muskegon Outdoor Environmental Education Center. Dip nets were used to collect surficial sediment and the material was processed in the field through a 1 mm sieve. Hexagenia were removed from the sieve and transferred to vials containing 90% ethanol. These collections were designed to be qualitative in nature to determine the presence/absence of the mayfly.

In the laboratory, the retained residue was transferred to a white enamel pan and all organisms were removed and sorted into groups (amphipods, oligochaetes, sphaeriids, chironomids, *Dreissena*, gastropods, and other) using a 1.75X magnifier lamp. Samples with large numbers of *Dreissena* were split (one quarter to one half) and a randomly selected portion was picked and applied to the rest of the sample. All organisms were identified to the lowest practical taxonomic level. Oligochaetes were reduced proportionately with a Folsom plankton splitter whenever counts were ≥ 200 in a sample so at least 100 were identified. Chironomids and oligochaetes were placed in lacto-no-phenol and warmed for 20 min at 60°C for clearing. Specimens were then mounted on slides in 100% glycerol for identification. The taxonomic group and the keys used for species identifications are as follows: Oligochaetes: Kathman and Brinkhurst (1998); Chironomidae: Epler (1995); Hirudinea: Klemm (1972); Tricoptera: Wiggins (1977); and Ephemeroptera: Burks (1953).

Table I- 1. Coordinates and depth of sites sampled by petite PONAR for benthic macroinvertebrates in Muskegon Lake, November 2006.

Station	Lat	Long	Depth (m)
Musk-1	43 13.387	86 18.690	12.0
Musk-5	43 13.953	86 15.922	6.4
Musk-6	43 13.986	86 15.922	4.1
Musk-7	43 14.007	86 15.925	7.9
Musk-10	43 14.631	86 15.561	7.4
Musk-11	43 14.723	86 15.391	8.8
Musk-12	43 14.562	86 14.961	4.2
Musk-15	43 14.604	86 16.461	9.6
Musk-16A	43 14.457	86 15.479	5.4
3ML	43 13.47	86 17.87	20.4
B30	43 14.91	86 15.70	8.7
C40	43 14.35	86 16.40	11.4
D40	43 13.67	86 17.00	12.5
D46	43 13.75	86 17.17	14.1
E40	43 13.95	86 18.83	12.2

I.2.3 Statistical Analyses

Data collected in the present study were compared to data collected in 1999 (Carter et al. 2006). The 1972 study sampled the same sites with the same collection and enumeration methods. The following metrics were calculated for each year: Shannon-Weaver diversity (with $\log_2$), oligochaete-chironomid ratio (O/C, calculated as ol/ ol + chir and reported as a percentage), the mean number of taxa at each site, relative oligochaete density, and the chironomid trophic status index. The O/C generally reflects the tendency for tolerant oligochaete species to increase their abundance relative to sedentary chironomids in conditions of nutrient enrichment (Wiederholm 1980). The chironomid trophic index used tolerance values from Hilsenhoff (1987) and Barbour *et. al* (1999) and calculated the index on a ten-point scale. In addition, differences between major taxonomic groups were evaluated. The differences in paired sites between the two years were evaluated using the Mann-Whitney U-test.

I.2.4 Chemical Analyses

Chemical and physical parameters were monitored at each location using a Hydrolab DataSonde 4a. We measured pH, redox potential, chlorophyll *a*, DO, DO% saturation, temperature, total dissolved solids, turbidity and specific conductance as vertical profiles. Grab samples were analyzed for soluble reactive phosphorus, nitrate-N, and ammonia-N at the Annis Water Resources Institute laboratory. A summary of analytical methods is given in Table I-2.

Table I- 2. Laboratory Analytical Methods

Parameter	Preparation	Preservation	Holding Time	Methods Reference
pH*	*	*	*	Hydrolab 1998
Redox potential*	*	*	*	Hydrolab 1998
Chlorophyll *a**	*	*	*	Hydrolab 1998
Dissolved Oxygen*	*	*	*	Hydrolab 1998
Temperature*	*	*	*	Hydrolab 1998
Specific conductance*	*	*	*	Hydrolab 1998
Turbidity	*	*	*	Hydrolab 1998
Total Dissolved Solids	*	*	*	Hydrolab 1998
SRP-P	0.45 *u*m filter in field	Freeze -10°C	28 days	4500-P F.
NH_3-N	filter to remove turbidity	H_2SO_4 Cool to 4°C	28 days	4500-NH_3 H.
NO_3-N	0.45 *u*m filter	Freeze -10°C	28 days	4100 C.
SO_4	0.45 *u*m filter	Freeze -10°C	28 days	4100 C.
Cl	0.45 *u*m filter	Freeze -10°C	28 days	4100 C.

* Measured directly in the field

I.3 Results

The results of the field and laboratory analyses are given in Table I-3. Limited variation in chemical parameter concentrations was observed between locations in Muskegon Lake at the time of sampling.

The results of the enumeration and identification of the benthos sampled at 15 sites in 2006 are given in Table I-4. A total of 55 taxa were identified, with an average of 10 ± 2.48 taxa per station. Total density was generally high and ranged between 3,555 and 67,473 organisms/m^2 with 10 of 15 sites having >5000 organisms/m^2. Oligochaeta were the most

Table I- 3. Physical/Chemical Parameters Measured in Muskegon Lake (November 2006).

Location	Date Sampled	Temperature (ºC)	Dissolved Oxygen (%D.O.)	Dissolved Oxygen (mg/L)	Spec. Conductance (µS/cm)	TDS (g/L)	Turbidity (NTU)	pH	ORP (mV)	NH_3-N (mg/L; D.L.=0.01)	SRP (mg/L; D.L.=0.01)	Cl^- (mg/L; D.L.=0.1)	SO_4-S (mg/L; D.L.=0.1)	NO_3-N (mg/L; D.L.=0.01)
Musk-1	11/1/2006	20.24	95.5	9.24	380.8	0.2445	16	8.18	329	0.02	0.01	18	27	0.01
Musk-5	11/1/2006	20.10	76.8	7.30	396.7	0.2543	23	7.99	329	0.05	<0.01	21	25	<0.01
Musk-6	11/1/2006	22.13	114.6	10.16	430.2	0.2751	15	8.45	355	0.08	<0.01	37	21	<0.01
Musk-7	11/1/2006	20.88	89.4	8.00	349.3	0.2236	18	7.86	373	0.01	<0.01	21	25	<0.01
Musk-10	11/1/2006	23.05	105.9	9.11	383.2	0.2451	14	8.17	368	0.03	<0.01	23	23	<0.01
Musk-11	11/1/2006	22.90	112.7	9.90	364.2	0.2360	16	8.21	377	0.03	<0.01	25	25	<0.01
Musk-12	11/1/2006	21.16	86.9	7.70	374.5	0.2393	21	7.90	381	0.02	<0.01	17	22	<0.01
Musk-15	11/1/2006	22.15	86.8	7.64	382.3	0.2447	14	8.04	270	0.03	<0.01	26	27	<0.01
Musk-16A	11/1/2006	21.69	86.7	7.70	370.6	0.2378	18	7.83	376	0.04	0.02	22	28	0.02
3ML	11/1/2006	23.09	102.2	8.88	371.6	0.2379	17	8.51	363	0.04	<0.01	23	28	<0.01
B30	11/1/2006	21.29	63.5	5.62	395.9	0.2534	19	7.85	223	0.04	<0.01	21	25	<0.01
C40	11/1/2006	22.53	149.7	13.01	367.6	0.2352	13	8.98	328	0.02	<0.01	26	29	<0.01
D40	11/1/2006	20.06	85.0	7.74	391.9	0.2509	11	8.22	370	0.04	<0.01	28	28	<0.01
D46	11/1/2006	20.92	96.8	8.69	353.2	0.2262	14	8.49	366	0.02	<0.01	21	24	<0.01
E40	11/1/2006	21.15	72.0	6.51	403.7	0.2588	17	7.70	431	0.08	<0.01	25	20	<0.01

Table I- 4. Mean Abundance of Benthic Macroinvertebrates in Muskegon Lake (November 2006).

Station	**Musk-1**		**Musk-5**		**Musk-7**		**Musk-10**		**Musk-11**		**Musk-12**		**Musk-15**	
Taxa	Mean #/m^2	SE	Mean #/m^2	SE	Mean #/m^2	SE	Mean #/m^2	SE	Mean #/m^2	SE	Mean #/m^2	SE	Mean #/m^2	SE
Turbellaria	26	5	320	105	90	34	286	112	62	20	769	234	33	9
Oligochaeta	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lumbriculidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Stylodrilus heringianus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Naididae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Arcteonais lomondi	17	5	0	0	0	0	0	0	0	0	0	0	0	0
Dero digitata	606	193	43	20	27	12	49	20	94	7	253	55	271	90
Dero flabelliger	0	0	43	21	0	0	0	0	0	0	0	0	0	0
Piguetiella michiganensis	0	0	0	0	0	0	0	0	0	0	228	58	0	0
Haemonais waldvogeli	0	0	0	0	35	5	23	9	0	0	0	0	0	0
Tubificidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aulodrilus americanus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aulodrilus limnobius	82	12	8	2	75	25	43	11	43	4	13	7	66	26
Aulodrilus pigueti	291	78	706	133	56	20	66	15	119	56	638	228	700	162
Aulodrilus pluriseta	541	268	0	0	0	0	0	0	39	7	43	12	133	7
Ilyodrilus templetoni	43	9	0	0	0	0	0	0	0	0	43	16	0	0
Isocheatides freyi	43	8	0	0	43	13	0	0	0	0	0	0	39	15
Limnodrilus cervix variant	0	0	0	0	0	0	0	0	43	20	0	0	0	0
Limnodrilus hoffmeisteri	43	6	43	15	0	0	43	9	66	29	43	7	0	0
Limnodrilus maumeensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Limnodrilus udekemianus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Potamothrix moldaviensis	0	0	0	0	0	0	0	0	0	0	43	20	0	0
Quistadrilus multisetosus	43	13	43	13	46	21	43	15	277	92	43	10	450	129
Immatures w/o hair chaetae	631	241	844	201	1344	189	1067	409	2620	1174	4396	858	466	89
Immatures w/hair chaetae	43	12	137	58	166	72	148	45	1264	397	168	74	729	125
Total Oligochaetes	2382	847	1867	464	1792	356	1482	533	4566	1787	5911	1344	2853	643
Polychaeta	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Manayunkia speciosa	0	0	0	0	0	0	33	10	0	0	0	0	0	0
Hirudinea	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Glossiphoniidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Alboglossiphonia heteroclita	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Helobdella stagnalis	41	1	0	0	0	0	0	0	0	0	0	0	0	0
Helobdella elongata	0	0	0	0	0	0	0	0	57	16	0	0	0	0
Mollusca	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gastropoda	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Amnicola sp.	0	0	78	22	248	94	386	167	0	0	277	89	0	0
Bithynia sp.	0	0	18	9	0	0	0	0	0	0	0	0	0	0
Valvata tricarinata	0	0	290	72	321	82	44	12	0	0	30	13	56	26
Valvata sincera	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Bivalvia	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pisidium sp.	0	0	2291	768	903	436	2321	700	444	152	976	168	806	210
Sphaerium sp.	0	0	94	37	346	146	837	411	241	68	470	117	107	3
Dreissena rostriformis	0	0	32	15	29	1	0	0	0	0	0	0	0	0
Dreissena polymorpha	0	0	895	365	363	114	198	71	27	11	57656	7746	24	7
Isopoda	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Caecidotea	0	0	0	0	47	14	0	0	0	0	0	0	0	0
Amphipoda	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gammarus sp.	22	4	263	75	30	9	59	28	107	23	781	327	222	78
Hyalella sp.	61	29	63	8	0	0	0	0	0	0	14	3	0	0
Echinogammarus sp.	0	0	0	0	0	0	0	0	0	0	223	73	0	0
Total Amphipods	83	33	327	83	30	9	59	28	107	23	1018	403	222	78
Diptera	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ceratopogonidae*	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Probezzia sp.	0	0	0	0	28	12	27	10	0	0	0	0	0	0
Chaoboridae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chaoborus sp.	192	18	41	19	141	50	534	96	139	42	0	0	119	38
Chironomidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chironominae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chironomus sp.	615	247	25	8	65	11	21	2	340	64	47	16	502	63
Cladopelma sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cryptochironomus sp.	0	0	274	12	302	73	28	12	342	165	42	21	159	6
Cryptochironomus digitatus	0	0	0	0	0	0	0	0	0	0	0	0	19	6
Dicrotendipes sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Paratanytarsus sp.	0	0	0	0	0	0	0	0	0	0	0	0	27	6
Polypedilum spp.	0	0	0	0	0	0	0	0	0	0	31	11	0	0
Tanytarsus sp.	21	5	0	0	0	0	0	0	0	0	0	0	0	0
Tribelos jucundum	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Orthocladiinae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Heterotrissocladius oliveri	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tanypodinae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ablabesmyia annulata	0	0	0	0	0	0	32	12	0	0	0	0	0	0
Coelotanypus concinnus	0	0	2126	499	496	246	513	211	409	143	0	0	62	10
Paraphaenocladius	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Procladius sp.	303	102	270	82	217	50	96	19	561	283	174	54	164	58
Total Chironomids	939	353	2695	601	1081	380	691	256	1653	656	294	103	933	149
Ephemeroptera	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Caenis sp.	0	0	0	0	0	0	0	0	0	0	46	16	0	0
Tricoptera	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ocetis sp.	0	0	26	9	66	18	0	0	0	0	26	10	0	0
Neureclipsis sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Total organisms	3662	1256	8974	2569	5483	###	6898	2406	7295	2775	67473	10243	5153	1164

Table I-4 Cont'd) Mean Abundance of Benthic Macroinvertebrates in Muskegon Lake (November 2006).

Station	Musk-16A		3ML		B-30		C-40		D-40		D-46		E-40	
Taxa	Mean #/m^2	SE	Mean #/m^2	SE	Mean #/m^2	SE	Mean #/m^2	SE	Mean #/m^2	SE	Mean #/m^2	SE	Mean #/m^2	SE
Turbellaria	241	80	43	19	44	16	63	28	43	15	110	45	1638	690
Oligochaeta	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Lumbriculidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Stylodrilus heringianus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Naididae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Arcteonais lomondi	43	11	43	17	0	0	0	0	0	0	0	0	0	0
Dero digitata	0	0	0	0	17	5	80	32	43	13	29	9	101	26
Dero flabelliger	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Piguetiella michiganensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Haemonais waldvogeli	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tubificidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Aulodrilus americanus	336	89	0	0	0	0	0	0	0	0	0	0	0	0
Aulodrilus limnobius	76	16	43	14	0	0	72	30	14	4	0	0	0	0
Aulodrilus pigueti	184	66	427	104	137	50	397	35	139	44	25	8	0	0
Aulodrilus pluriseta	0	0	43	15	0	0	95	44	43	14	43	16	0	0
Ilyodrilus templetoni	0	0	43	14	0	0	14	5	43	14	43	24	0	0
Isocheatides freyi	0	0	1	1	0	0	14	3	43	10	14	4	0	0
Limnodrilus cervix variant	0	0	0	0	0	0	43	8	114	49	0	0	0	0
Limnodrilus hoffmeisteri	0	0	43	4	43	14	76	11	43	20	43	4	2	1
Limnodrilus maumeensis	0	0	43	14	0	0	43	4	43	7	22	8	0	0
Limnodrilus udekemianus	14	5	0	0	43	6	43	11	43	4	14	4	0	0
Potamothrix moldaviensis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Quistadrilus multisetosus	159	54	143	44	43	16	163	48	43	11	134	50	0	0
Immatures w/o hair chaetae	3955	1078	454	154	334	159	778	330	2331	90	2830	474	1239	104
Immatures w/hair chaetae	132	29	222	84	120	13	391	103	358	45	110	36	0	0
Total Oligochaetes	4899	1348	1506	462	737	264	2209	663	3300	326	3307	637	1342	131
Polychaeta	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Manayunkia speciosa	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Hirudinea	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Glossiphoniidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Alboglossiphonia heteroclita	0	0	0	0	0	0	0	0	0	0	0	0	102	27
Helobdella stagnalis	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Helobdella elongata	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Mollusca	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gastropoda	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Amnicola sp.	675	255	0	0	27	6	104	43	41	6	71	26	0	0
Bithynia sp.	31	3	0	0	0	0	0	0	0	0	0	0	63	23
Valvata tricarinata	151	55	0	0	42	17	0	0	0	0	31	13	0	0
Valvata sincera	54	20	26	5	32	10	0	0	0	0	0	0	0	0
Bivalvia	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Pisidium sp.	486	161	398	136	1627	266	142	12	126	60	513	137	161	40
Sphaerium sp.	76	20	0	0	150	76	53	24	0	0	31	1	0	0
Dreissena rostriformis	0	0	0	0	0	0	0	0	0	0	0	0	9678	3283
Dreissena polymorpha	8595	4220	207	98	59	22	71	9	128	42	0	0	0	0
Isopoda	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Caecidotea	0	0	0	0	29	6	0	0	0	0	17	7	457	213
Amphipoda	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Gammarus sp.		0	371	95		0	32	13	61	16	98	31	1377	263
Hyalella sp.	26	10	31	8	0	0	21	8	0	0	0	0	97	20
Echinogammarus sp.	0	0	0	0	0	0	0	0	0	0	0	0	525	118
Total Amphipods	26	10	402	103	0	0	53	21	61	16	98	31	1999	400
Diptera	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ceratopogonidae*	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Probezzia sp.	0	0	0	0	0	0	0	0	30	16	0	0	0	0
Chaoboridae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chaoborus sp.	75	13	199	56	183	91	797	148	106	36	434	56	47	2
Chironomidae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chironominae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Chironomus sp.	697	241	608	145	35	8	661	142	1661	728	657	205	1302	642
Cladopelma sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Cryptochironomus sp.	90	34	65	18	189	51	0	0	68	24	0	0	56	27
Cryptochironomus digitatus	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Dicrotendipes sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Paratanytarsus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Polypedilum spp.	0	0	19	6	0	0	0	0	0	0	0	0	0	0
Tanytarsus sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Tribelos jucundum	0	0	0	0	0	0	0	0	0	0	0	0	49	20
Orthocladiinae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Heterotrissocladius oliveri	32	15	0	0	0	0	0	0	0	0	0	0	0	0
Tanypodinae	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ablabesmyia annulata	0	0	0	0	0	0	33	16	0	0	0	0	0	0
Coelotanypus concinnus	0	0	15	2	1708	479	0	0	15	2	0	0	0	0
Paraphaenocladius	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Procladius sp.	106	14	95	28	41	16	30	12	197	23	0	0	213	67
Total Chironomids	925	304	802	199	1973	554	724	169	1941	777	657	205	1620	756
Ephemeroptera	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Caenis sp.	0	0	0	0	21	10	0	0	0	0	0	0	0	0
Tricoptera	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Ocetis sp.	70	9	0	0	0	0	0	0	0	0	0	0	0	0
Neureclipsis sp.	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Total organisms	16303	6496	3583	1079	4922	1339	4215	1117	5776	1293	5269	1157	17107	5566

Table I- 5. Mean Abundance (#/m^2) of Major Taxonomic Groups in Muskegon Lake (November 2006).

	Oligochaetes		Chironomids		Sphaeriids		Amphipods		*Dreissena*		Other		Total
	Mean #/m^2	%	Mean #/m^2	%	Mean #/m^2	%	Mean #/m^2	%	Mean #/m^2	%	Mean #/m^2	%	Mean #/m^2
Musk-1	2382	65%	939	26%	0	0%	83	2%	0	0%	259	7%	3662
Musk-5	1867	21%	2695	30%	94	1%	327	4%	927	10%	3064	34%	8974
Musk-7	1792	32%	1081	19%	346	6%	30	1%	393	7%	1935	35%	5577
Musk-10	1416	21%	691	10%	837	12%	59	1%	198	3%	3696	54%	6897
Musk-11	4566	63%	1653	23%	241	3%	107	1%	27	0%	702	10%	7295
Musk-12	5911	9%	294	0%	470	1%	1018	2%	57656	85%	2124	3%	67473
Musk-15	2853	55%	933	18%	107	2%	222	4%	24	0%	1013	20%	5153
Musk-16A	4899	30%	925	6%	76	0%	26	0%	8595	53%	1782	11%	16303
3ML	1506	42%	802	23%	0	0%	402	11%	207	6%	638	18%	3555
B-30	737	15%	1973	40%	150	3%	0	0%	59	1%	2004	41%	4922
C-40	2209	52%	724	17%	53	1%	53	1%	71	2%	1106	26%	4215
D-40	3300	57%	1941	34%	0	0%	61	1%	128	2%	336	6%	5767
D-46	3307	63%	657	12%	31	1%	98	2%	0	0%	1176	22%	5269
E-40	1342	8%	1620	10%	0	0%	1512	9%	9678	58%	2469	15%	16620

abundant group at all but three of the sites sampled, comprising between 737/m^2 and 5,911/m^2 (Table I-5). Immature tubificids were 70% of the total abundance of Tubificidae from all stations. Densities of Chironomidae ranged between 657/m^2 and 2,695/m^2 at all sampling sites (Table I-5). Relative oligochaete density was variable (range 8% to 65%) but exceeded 50% at 6 sites sampled (Table I-5). The proportion of oligochaetes was lowest at sites Musk-12 (9%) and E-40 (8%) whereas this taxa group was the second most abundant group at 12 of the stations sampled (Table I-4). A total of 14 taxa were identified (Table I-4). *Chironomus* spp. and *Procladius* spp. were found at all sites except D-46 (Table I-4). With the exception of *Coelotanypus concinnus* and *Cryptochironomus* spp., the remaining species were found infrequently and were generally low in abundance. The Sphaeriidae as a group were variable, ranging from being absent at four sites to relatively high abundance (837/m^2) at Musk-10 (Table I-5). The amphipods, which included three taxa, were found at all but one site and had generally low abundances (≤ 400/m^2) with the exception of three sites (Table I-5). Two sites, Musk-12, and E-40, had relatively high abundances of amphipods with 1018/m^2 and 1512/m^2, respectively. *Gammarus* spp. tended to be the dominant taxon of the three identified, and was found at all but two sites (Table I-4). Abundances of the remaining groups and constituent taxa can be found in Table I-4. *Hexagenia spp.* was present at locations Hex 1 and Hex 2. A total of 30 individuals were collected from a 10 m^2 area at Fisherman's Landing (Hex 1) and 21 individuals were taken from a similar sized area at Hex 2.

Comparisons of 1999 and 2006 data for total organisms, total oligochaetes, and total chironomids are shown in Figures I-3, and I-4, respectively and summarized in Table I-6. Total benthic organisms showed little variability between 1999 and 2006 levels at all stations except Musk-12, Musk-15, and E-40. Changes at these locations were related to increases in the 2006 densities of zebra mussels at Musk-12 and quagga mussels at E-40. Zebra mussel densities decreased at Musk 15 in 2006 compared to 1999. The difference in total organisms between years was not significant (9,652±2918) vs. 11,579±2872; Mann-Whitney U; ρ=0.96). Mean oligochaete densities were lower at all stations in 2006 with the exception of Musk-12. This station had the highest abundance of zebra mussels. Mean oligochaete densities for 2006 were significantly lower than 1999 (4,562±664 vs. 2,725±700; Mann-Whitney U; ρ=0.024). In contrast, chironomid densities were significantly greater in 2006 (Table I-5; 677± 75 vs. 1,209±390; Mann-Whitney U; ρ=0.009).

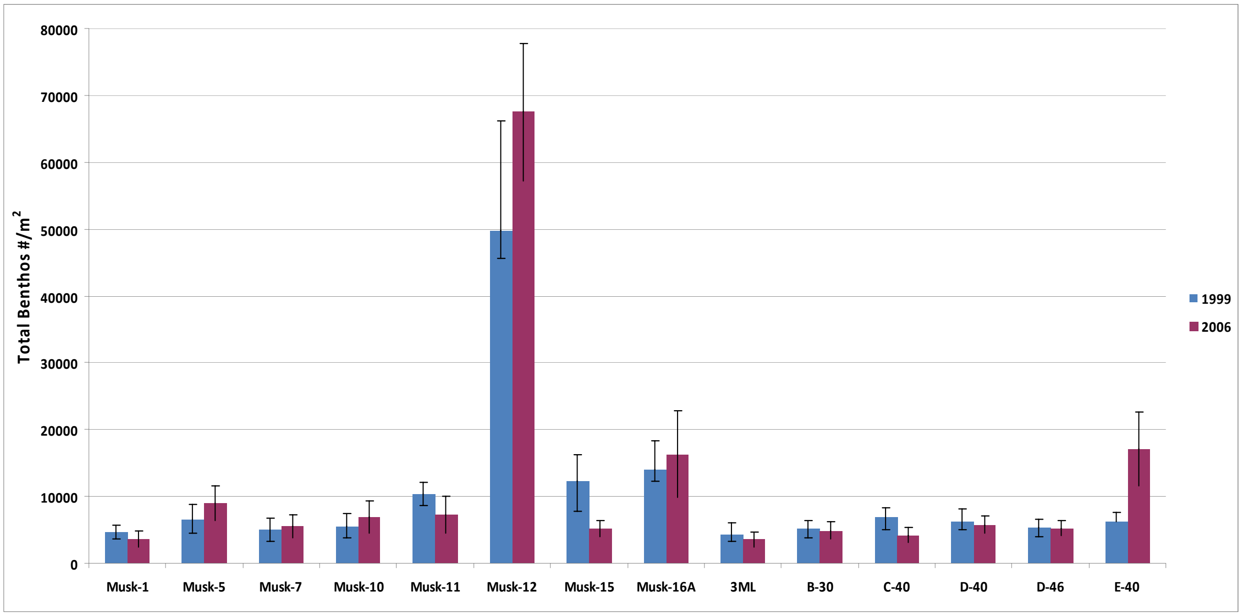


Figure I- 3. Total benthos abundance from Muskegon Lake in 1999 and 2006. Error bars represent standard error. Mann-Whitney U (1999-2006); ρ=0.96.

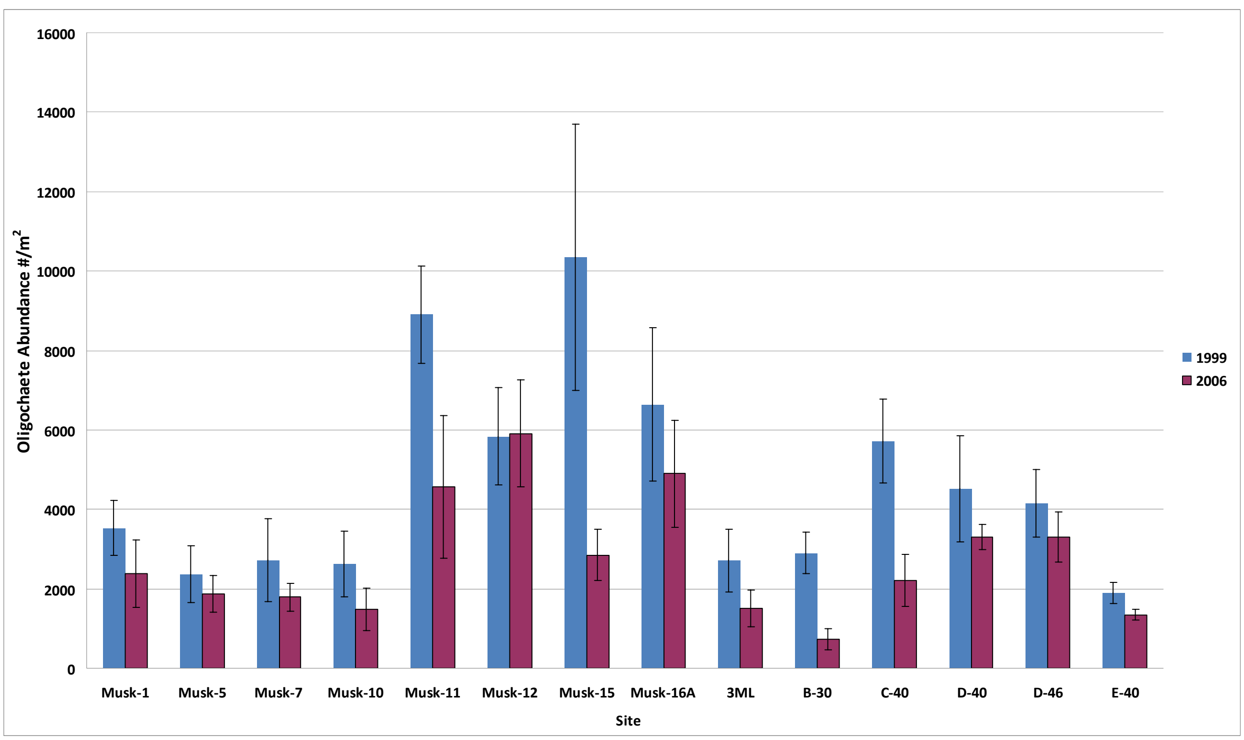


Figure I- 4. Oligochaete abundance from Muskegon Lake in 1999 and 2006. Error bars represent standard error. Mann-Whitney U (1999-2006); ρ=0.024.

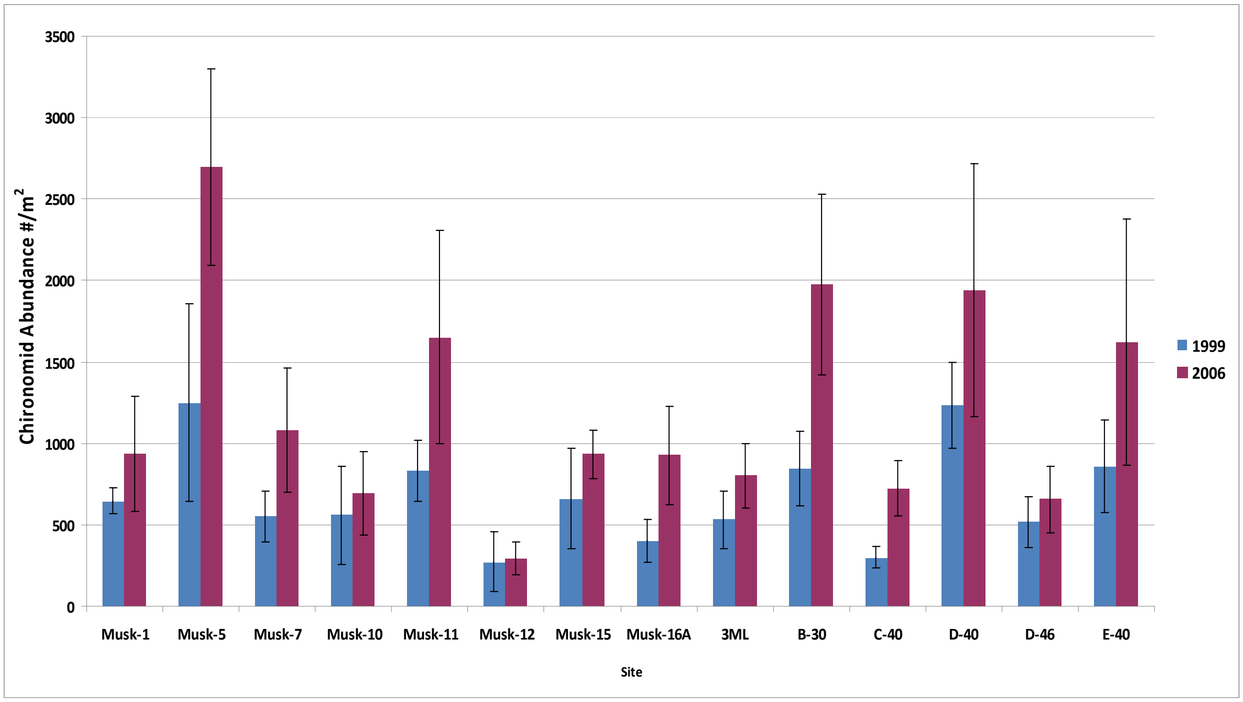


Figure I- 5. Chironomid abundance from Muskegon Lake in 1999 and 2006. Error bars represent standard error. Mann-Whitney U (1999-2006); ρ=0.009.

Table I- 6. Mean (S.E.) and Mann-Whitney U-test p-values for selected benthic taxa and parameters of 15 matched pairs of sites. Chironomid Trophic Index is based on Hilsenhoff (1987) and Barbour *et. al* (1999). Benthic groups are #/m^2. The oligochaete/chironomid ratio is in %. Diversity is Shannon-Weaver ($\log_2$).

	1999	2006	p-value
Total Benthos	9,652(2918)	11,579(2872)	0.96
Total Oligochaeta	4,562(664)	2,725(700)	**0.024**
Total Chironomidae	677(75)	1,209(390)	**0.009**
Oligochaete/Chironomid	84(2)	66(5)	**0.005**
Chironomid Trophic Index	8.71(0.38)	8.80(0.45)	0.86
Shannon Weaver Diversity	1.88(0.17)	2.08(0.22)	0.137

Comparisons of 1999 and 2006 data for Shannon Weaver Diversity, Oligochaete/Chironomid ratio, and Chironomid Trophic Index are shown in Figures I-6, I-7, and I-8, respectively. Shannon Weaver diversity values were higher in 2006 however the difference between years was not significant (1.66±0.17 vs. 2.08±0.22; Mann-Whitney U; ρ=0.137). Similarly, chironomid trophic index values were not significantly different between years (8.71± 0.38 vs. 8.80±0.45; Mann-Whitney U; ρ=0.86). The oligochaete/chironomid ratio in 2006 was significantly lower than in 1996 (84%± 2 vs. 66%± 5; Mann-Whitney U; ρ=0.005).

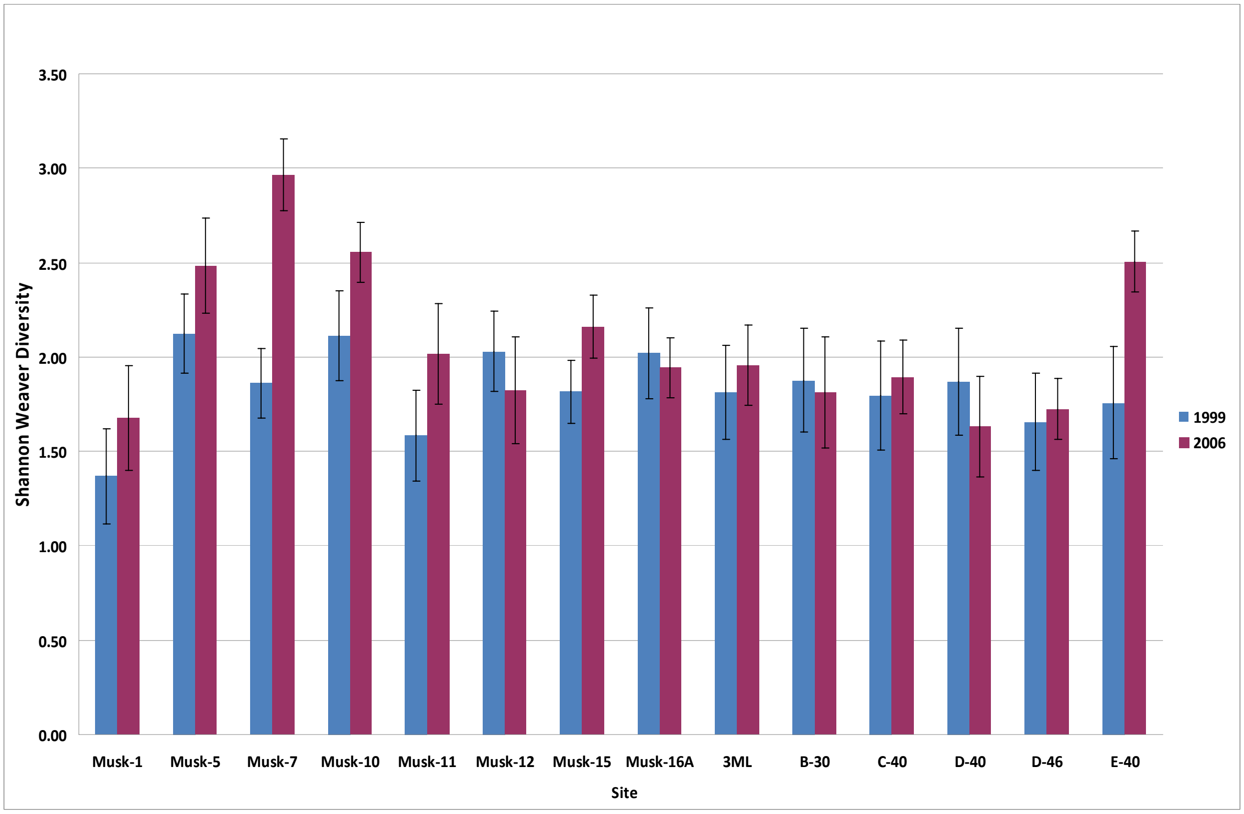


Figure I- 6. Shannon-Weaver diversity ($\log_2$) for Muskegon Lake benthos in 1999 and 2006 (error bars represent standard error). Oligochaetes were lumped into one group due to the high number of immatures. Mann-Whitney U (1999-2006); ρ=0.137.

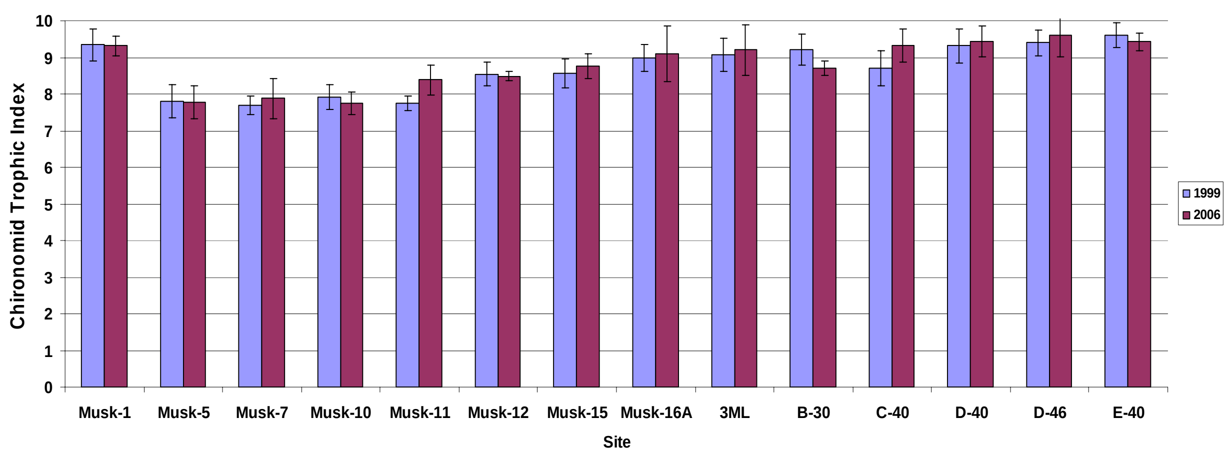


Figure I- 7. Chironomid-based Trophic Status Index in Muskegon Lake for 1999 and 2006. Mann-Whitney U (1999-2006); ρ=0.86.

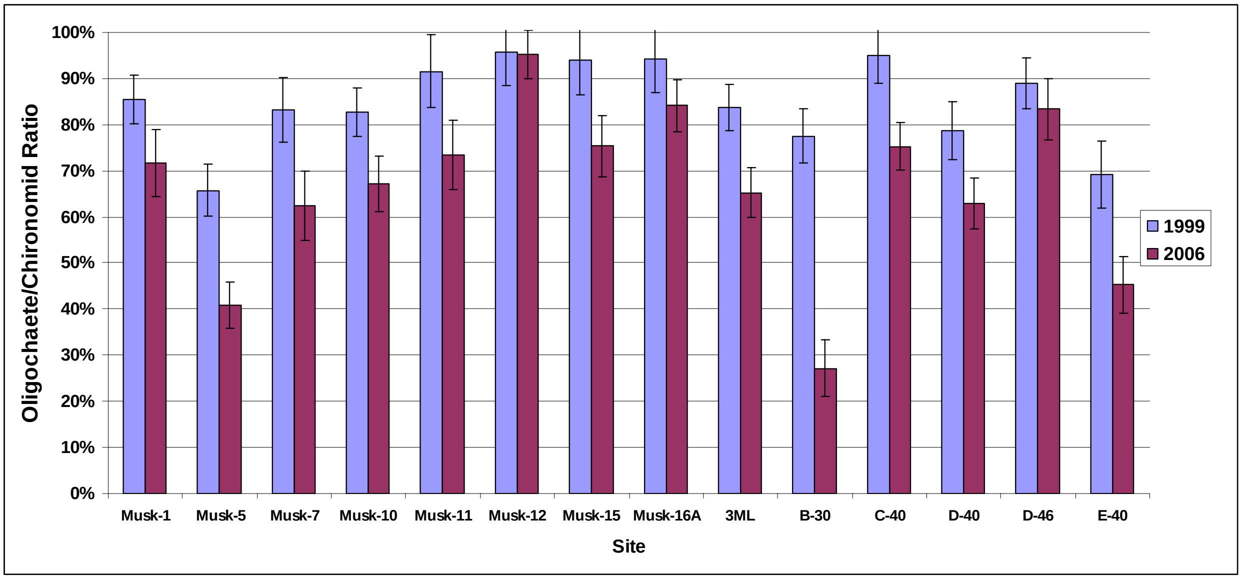


Figure I- 8. Oligochaete-chironomid ratio (%, error bars represent standard error) for 1999 and 1972 in Muskegon Lake. Mann-Whitney U (1999-2006); ρ=0.005.

I.4 Discussion

The significant increase in chironomids and decrease in oligochaetes shows that the benthic invertebrate community in Muskegon Lake continues to improve (Table I-6). Shannon Weaver diversity and total benthic organisms were not significantly different between years, indicating stable benthic conditions. Krieger and Ross (1993) found increases in Chironomidae and reduced relative density of oligochaetes were particularly indicative of improved conditions in the Cleveland Harbor area of Lake Erie. The chironomid trophic index was not significantly different between the two years (Table I-6). The numerically dominant species in each year were *Chironomus* spp., *Cryptochironomus* spp, *Coelotanypus concinnus*, and *Procladius* spp. (all indicative of eutrophic conditions). Affects of sewage spills into eastern Muskegon Lake during 1999 and 2006 may have masked the chironomid trophic index by eliminating or reducing less tolerant species from areas sampled. Another possibility is that the chironomid populations have reached a point that reflects the current trophic status of the Muskegon Lake and will require ecosystem wide changes to organic deposition rates and/or hypolimnetic oxygen depletion to alter chironomid trophic index. Finally, hypolimnetic conditions (including sediments) may lag improvement in epilimnetic conditions due to different flushing rates. Further sampling is required to determine if the chironomid community found in 1999 is the norm or if there was indeed an impact from the sewage spill.

The Muskegon Lake PAC established numerical targets to delist the Degradation of Benthos BUI. The targets and their current status are presented in Table I-7. The 1999 investigation of Muskegon Lake (Rediske *et al.* 2002) identified sediment toxicity in the vicinity of the Division Street Outfall (amphipod toxicity at Musk-5: 60% survival). Subsequent investigations in 2005 (Rediske 2005) found five locations with 37%-60% amphipod survival closer to the stormwater outlet. A feasibility study is currently under review by EPA/MDEQ to

Table I- 7. Status of Muskegon Lake Degradation of Benthos Delisting Targets.

Indicator	Target	1999	2006
1. Sediment Toxicity	Amphipod Survival >60%	DSO	DSO
2. *Hexagenia*	Present in river mouth littoral zone	Yes	Yes
3. % Oligochaeta (w/o ZM)	< 75%	69	45
4. Chironomidae (#/m^2)	> 500	677	1209
5. Diversity (Shannon Weaver)	> 1.5	1.88	2.08

evaluate remedial alternatives and sediment removal guidelines. The completion date for remedial design is projected to be late 2009. The remedial dredging proposed will remove areas of significant sediment contamination and toxicity and address this target. The remaining targets require two years of monitoring data at 5 year intervals to consider delisting. *Hexagenia spp*. were present in the littoral zone near the mouth of the Muskegon River in 2000 (Rediske unpublished data) and at similar locations during this study. In addition, annual mayfly hatches are reported annually in the cities of Muskegon, and North Muskegon, confirming the presence of this organism in the AOC. Based on these data, the presence of *Hexagenia* in the littoral zone of Muskegon Lake near the mouth of the Muskegon River has been documented in 2000 and 2006, demonstrating that the requirements of the second target have been met.

Targets for oligochaetes, chironomids, and diversity also were met using the 1999 and 2006 data (Table I-7). The % oligochaeta (without zebra and quagga mussels) needed to be below 75% and the 1999 and 2006 means were 69% and 45%, respectively. Chironomids were required to have an abundance ≥500/m^2 and the 1999 and 2006 means were 677/m^2 and 1209/m^2, respectively. With respect to diversity, the target required a value of >1.66 and the results for 1999 and 2006 were 1.88 and 2.08, respectively.

I.5 Conclusions

Benthic macroinvertebrate populations in the Muskegon Lake Area of Concern (AOC) were evaluated in support of delisting the Degradation of Benthos Beneficial Use Impairment (BUI). Benthic macroinvertebrates were collected and analyzed at 15 locations in 2006 and compared to previous data from 1999. A significant increase in chironomids and decrease in oligochaetes was observed in 2006, indicating that that the benthic invertebrate community in Muskegon Lake continued to improved from 1999 conditions. Shannon Weaver diversity, total benthic organisms, and the chironomid trophic index were not significantly different between years, indicating stable benthic conditions. All of the metrics either indicated stable or improving conditions in the benthic macroinvertebrate community.

The Muskegon Lake PAC established five numerical targets to delist the Degradation of Benthos BUI. The first target required the removal of areas where the sediment is toxic to aquatic organisms. Currently, the only known area that exceeds the target is the Division Street Outfall and this location is currently being evaluated for remedial dredging with a feasibility study. The remaining four targets required two years of monitoring data at 5 year intervals for delisting. According to the results of this study and the data collected in 1999, the benthic community is meeting the delisting targets. *Hexagenia spp*. was present in the littoral zone near the mouth of the Muskegon River in 2000 and at similar locations during this study. The remaining three targets for oligochaetes, chironomids, and diversity also were met using the 1999 and 2006 data. The % oligochaeta (without zebra and quagga mussels) was below 75% as the 1999 and 2006 means were 69% and 45%, respectively. The target for chironomids required this group to have an abundance $\geq 500/m^2$ and the 1999 and 2006 means were $677/m^2$ and $1209/m^2$, respectively. With respect to diversity, the target required a value of >1.66 and the results for 1999 and 2006 were 1.88 and 2.08, respectively. When the sediments at the Division Street Outfall are successfully remediated, the Muskegon Lake PAC should be able to delist the Degradation of Benthos BUI in the Muskegon Lake AOC.

I.5 References

APHA (American Public Health Association). 1998. Standard Methods for the Evaluation of Water and Wastewater. 20th edition. APHA, Washington, DC.

Barbour, M. T., J. Gerritsen, B. D. Snyder and J. B. Stribling. 1999. Rabid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates, and fish, second edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water, Washington, D. C.

Barton, D. R. and B. Anholt. 1997. The macrobenthos of Lake Ontario during 1964 to 1966, and subsequent changes in the deepwater community. Aquat. Sci. 59: 158-175.

Burks, B. D. 1953. The mayflies, ephemeroptera of Illinois. Bull. Ill. Nat. Hist. Surv. 26: 1-216.

Carter, G. S., T. F. Nalepa, and R. R. Rediske. 2006. Status and trends of benthic populations in a coastal drowned river mouth lake of Lake Michigan. Journal of Great Lakes Research 32:578-595.

Canfield, T. J., F. J. Dwyer, J. F. Fairchild, P. S. Haverland, C. G. Ingersoll, N. E. Kemble, D. R. Mount, T. W. Lapoint, G. A. Burton and M. C. Swift. 1996. Assessing contamination in Great Lakes sediments using benthic invertebrate communities and the sediment quality triad approach. J. Great Lakes Res. 22: 565-583.Carr, J. F. and J. K. Hiltunen. 1965. Changes in the bottom fauna of western Lake Erie from 1930 to 1961. Limnol. Oceanog. 10: 551-569.

Epler, J. H. 1995. Identification manual for the larval chironomidae (Diptera) of Florida. Florida Department of Environmental Quality, Tallahassee, FL.

Evans, E. 1976. Final report of the Michigan Bureau of Water Management's investigation of the sediments and benthic communities of Mona, White, and Muskegon Lakes, Muskegon County, Michigan, 1972.

---. 1992. Mona, White, and Muskegon Lakes in Muskegon County, Michigan: the 1950's to the 1980's. MI/DNR/SWQ-92/261. MI. Department of Natural Resources, Lansing, MI.

Great Lakes Commission. 2000. *Assessment of the Lake Michigan monitoring inventory: A report on the Lake Michigan tributary monitoring project*. Great Lakes Commission, Ann Arbor, MI.

Harman, W. N. 1997. Otsego Lake macrobenthos communities between 1968 and 1993: indicators of decreasing water quality. Journal of Freshwater Ecology 12: 465-476.

Hilsenhoff, W. L. 1987. An improved biotic index of organic stream pollution. Great Lakes Entomol. 20: 31-39.

Howmiller, R. P. and M. A. Scott. 1977. An environmental index based on relative abundance of oligochaete species. J. Wat. Poll. Con. Fed. 49: 809-815.

Kathman, R. D. and R. O. Brinkhurst. 1998. Guide to the freshwater oligochaetes of North America. Aquatic Resources Center, College Grove, TN.

Klemm, D. J. 1972. Freshwater leeches (Annelida: Hirudinea) of North America. 18050 ELDO5/72. U. S. Environmental Protection Agency, Washington D. C.

Krieger, K. A. and L. S. Ross. 1993. Changes in the benthic community of Cleveland Harbor area of Lake Erie from 1978 to 1989. J. Great Lakes Res. 19: 250-257.

---. 1998. Contrasting response of oligochaetes (Annelida) and chironomids (Diptera) to the abatement of eutrophication in Lake Neuchatel. Aquat. Sci. 61: 206-214.

Milbrink, G. 1983. An improved environmental index based on the relative abundance of oligochaete species. Hydrobiologia 102: 89-97.

Nalepa, T. F. 1987. Long-term changes in the macrobenthos of southern Lake Michigan. Can. J. Fish. Aquat. Sci. 44: 515-524.

---. 1991. Status and trends of Lake Ontario macrobenthos. Can. J. Fish. Aquat. Sci. 48: 1558-1567.

Nalepa, T. F., G. A. Lang and D. L. Fanslow. 2000. Trends in benthic macroinvertebrate populations in southern Lake Michigan. Verh. Internat. Verein Limnol. 27: 2540-2545.

Peterson K. 1951. A Biological Survey of Muskegon Lake. Michigan Department of Natural Resources, Institute for Fisheries Research, 1276.

Rediske R., Thompson C., Schelske C., Gabrosek J., Nalepa T. F., and Peaslee G. 2002. Preliminary investigation of the extent of sediment contamination in Muskegon Lake, MI. U. S. Environmental Protection Agency, Great Lakes National Program Office, Chicago IL. GL-97520701-01.

Rediske, R. Solid phase toxicity assessment of the Division Street Outfall and Ryerson Creek Muskegon, Michigan. Prepared for Weston Solutions, Chicago, Il. 59 pp.

Schloesser, D. W., T. B. Reynoldson and B. A. Manny. 1995. Oligochaete fauna of western Lake Erie1961 and 1982: signs of sediment recovery. J. Great Lakes Res. 21: 294-306.

Steinman, A., M. Ogdahl, R. Rediske, C. Ruetz, B. Biddanda and L. Nemeth. (2008): Current status and trends in Muskegon Lake, Michigan. J. Great Lakes Research 34: 169-188.

Surber E. 1954. Results of a Biological Survey of Muskegon Lake in the Vivinity of the Central Paper Co. Michigan Water Resources Commission.

Wiederholm, T. 1980. Use of benthos in lake monitoring. J. Wat. Poll. Con. Fed. 52: 537-547.

Wiggins, G. B. 1977. Larve of the North American caddisfly genera (Tricoptera). University of Toronto Press, Toronto.

Wuycheck J. D. 1987. Muskegon Lake Area of Concern Remedial Action Plan. Michigan Department of Natural Resources, Lansing. MI.

Chapter II. IMPACTS TO THE MACROINVERTEBRATE COMMUNITY STRUCTURE OF RUDDIMAN CREEK FOLLOWING SEDIMENT REMEDIATION

Prepared By

Laurie Beth Nederveld and Richard R. Rediske

Annis Water Resources Institute
Grand Valley State University
740 West Shoreline Drive
Muskegon, Michigan 49441

Neil W. MacDonald and James P. Dunn

Grand Valley State University
Department of Biology
1 Campus Drive
Allendale, Michigan 49401

Donald G. Uzarski

Central Michigan University
Department of Biology
Mount Pleasant, Michigan 48859

January 2009

Executive Summary

The impact of sediment remediation on the composition, relative abundance, and diversity of the macroinvertebrate community inhabiting Ruddiman Creek (Muskegon Lake AOC) was evaluated in the second part of this investigation. Macroinvertebrate samples from all available habitat types at three study sites and three reference sites were collected using a Before-After, Control-Impacted (BACI) approach. Ryerson Creek, considered less disturbed with respect to heavy metal and organic chemical contaminants, served as an urbanized reference stream within the Muskegon Lake watershed. Samples were collected three months before the dredging and removal of contaminated sediment and four times over a span of 1.5 years after restoration activities were completed in Ruddiman Creek. In addition to macroinvertebrate collections, physical measurements, chemical analyses of water samples, and hydrologic measurements in Ruddiman and Ryerson Creeks were used to assess habitat changes as a result of remediation activities. The macroinvertebrate community in Ruddiman Creek was reduced in both abundance and diversity three months following sediment removal, but over one year after remediation, the abundance and diversity of Ruddiman Creek's macroinvertebrate community had returned to levels comparable to pre-remediation conditions. The Family Biotic Index (FBI) suggested some improvement in the overall condition of the two upstream sites on Ruddiman Creek, while the most heavily remediated downstream site remained in a degraded state. Stream quality FBI rankings in the fairly poor category throughout the project suggested that hydrologic impairments continue to negatively influence the macroinvertebrate community after remediation and additional restoration activities are needed to improve the ecological integrity of the Ruddiman Creek watershed.

II.1 Introduction

Streams flowing over landscapes modified by anthropogenic activity are often subject to varying forms and degrees of impairment. Habitat degradation and nonpoint source pollution inputs resulting from poor land use practices have been identified as causes of macroinvertebrate community impairment (Plafkin *et al.* 1989; Barbour *et al.* 1995; Stewart *et al.* 2000). Industrial effluents, municipal stormwater runoff, and poor agricultural practices can impair the ecological integrity of a stream by causing unstable flow, excessive input of fine sediment and nutrients (i.e. phosphorus and nitrogen), thermal regime modifications, heavy metals, and toxic organic compounds. Macroinvertebrate populations exposed to these degraded environmental conditions can exhibit decreases in abundance and diversity (Fisher *et al.* 1982; McElravy *et al.* 1989; Scrimgeour and Winterbourn 1989). Impacts to the organism can include growth reductions (Mattingly et al. 1981), declines in feeding efficiency (Broekhizen *et al*. 2001; Waters 1995), deformity (Camargo 1991), diminished reproduction (Mulvey and Diamond 1991), and altered competition behavior (Vuori 1994). Because macroinvertebrates are sensitive to environmental degradation, they are used extensively as indicators of general stream health and water quality over time (Bennett *et al*. 2004; Resh *et al*. 1996).

Our study stream, Ruddiman Creek, drains an urbanized watershed located in Muskegon County, Michigan, USA. Past wastewater and stormwater discharges, improper hazardous waste disposal, and contaminated groundwater inputs contributed to the contamination and degradation of this resource (Rediske 2002). As a result of past land use practices, numerous pollutants were introduced into the stream, including heavy metals (e.g. lead, cadmium, and chromium), polychlorinated biphenyls (PCBs), and Benzo(a)pyrene (Snell Environmental Group 2000; Earth Tech 2002). Snell Environmental Group (2000) and Earth Tech (2002) each identified contaminated sediments as the primary source of chemical exposure since groundwater and stormwater samples from the site were found to contain only low levels of anthropogenic chemicals. Heavy metal and PCB concentrations exceeded the Michigan Department of Environmental Quality's (MDEQ) site-specific sediment quality criteria for human contact and aquatic life (Rediske 2004). In addition, stream sediments contained visible oils (Nederveld 2005, personal observation). Consequently, the warmwater fishery and macroinvertebrate community have been rated as poor by the MDEQ, and the stream has been included on the Michigan 303(d) List of Impaired Waters (Wuycheck and Creal 2002, LeSage and Smith 2008). Sources of impairment include contaminated sediments and hydrologic instability (Wuycheck 1990). Because of degraded stream conditions and recent public health concerns expressed by residents, the Great Lakes National Program Office and the MDEQ conducted a 10.6 million dollar project to dredge and remove contaminated sediments in Ruddiman Creek and Ruddiman Pond. In addition to sediment remediation, limited hydrologic improvements were installed including the construction of a retention basin and channel braiding at two of three dredge areas. Between August 2005 and April 2006, Ruddiman Pond and seven sections of the main branch of Ruddiman Creek were dredged removing 68,710 m^3 of contaminated sediments (Janesak 2006). The primary objective of this remediation project was to "reduce the relative risks to humans, wildlife, and aquatic life" (Hilgeman 2005), yet an assessment of the implications for the biological community was not included in the project plan.

While sediment removal may result in an immediate decline in macroinvertebrate abundance and diversity directly following remediation activities (Quigley and Hall 1999; Gilkinson *et al.* 2005) these same metrics have been shown to increase over time following stream remediation (Adams *et al.* 2005). However, Kelaher *et al.* (2003) demonstrated the potential for sediment remediation activities to ultimately degrade biotic communities of stream systems rather than improve them.

The objective of our study was to evaluate the impact of sediment remediation on the biotic community of Ruddiman Creek, using macroinvertebrates as the primary indicator. Ryerson Creek, a system also impacted by urbanization but considered less disturbed with respect to heavy metal and organic chemical contaminants served as a reference stream for this evaluation. Remediation of Ruddiman Creek was intended to improve existing stream conditions through the dredging and removal of contaminated sediments. Our investigation used macroinvertebrate collections, chemical analyses of water samples, hydrologic measurements, and habitat evaluations in Ruddiman and Ryerson Creeks to evaluate impacts of stream remediation on the macroinvertebrate community. If anthropogenic chemicals were the main factor contributing to the degraded macroinvertebrate community, remediation of contaminated sediments in Ruddiman Creek would result in an increase in relative abundance and diversity compared to pre-remediation conditions. Since only a limited amount of hydrologic restoration was performed, flashy stream condition may still be impacting the macroinvertebrate community after sediment remediation. This investigation was conducted to measure the effects of sediment remediation and provide the basis for the development of a habitat restoration strategy.

II.2 Materials and Methods

II.2.1 Study Area

Study sites were located in the Ruddiman Creek watershed (13.0 km^2), an urbanized area that included the city of Muskegon (Figure II-1). A mix of residential (54%), commercial (20%), and industrial (11%) development cover the landscape. Natural features of the watershed included three stream reaches (north branch, .55 km; west branch, 2.14 km; and main branch, 3.09 km), a pond (0.04 km^2), and several forested, emergent, and scrub-shrub wetland areas (0.12 km^2). Approximately 44% of the main branch is enclosed in an underground storm sewer before it emerges from a 2.54 m storm sewer outfall. The main branch flows through residential and wetland areas and discharges into Ruddiman Pond, and ultimately Muskegon Lake. Three study sites were located on the main branch, upstream of its confluence with Ruddiman Pond. Sites 2 and 3 underwent sediment remediation, while Site 1 was influenced by upstream structures intended to moderate hydrologic extremes (Table II-1).

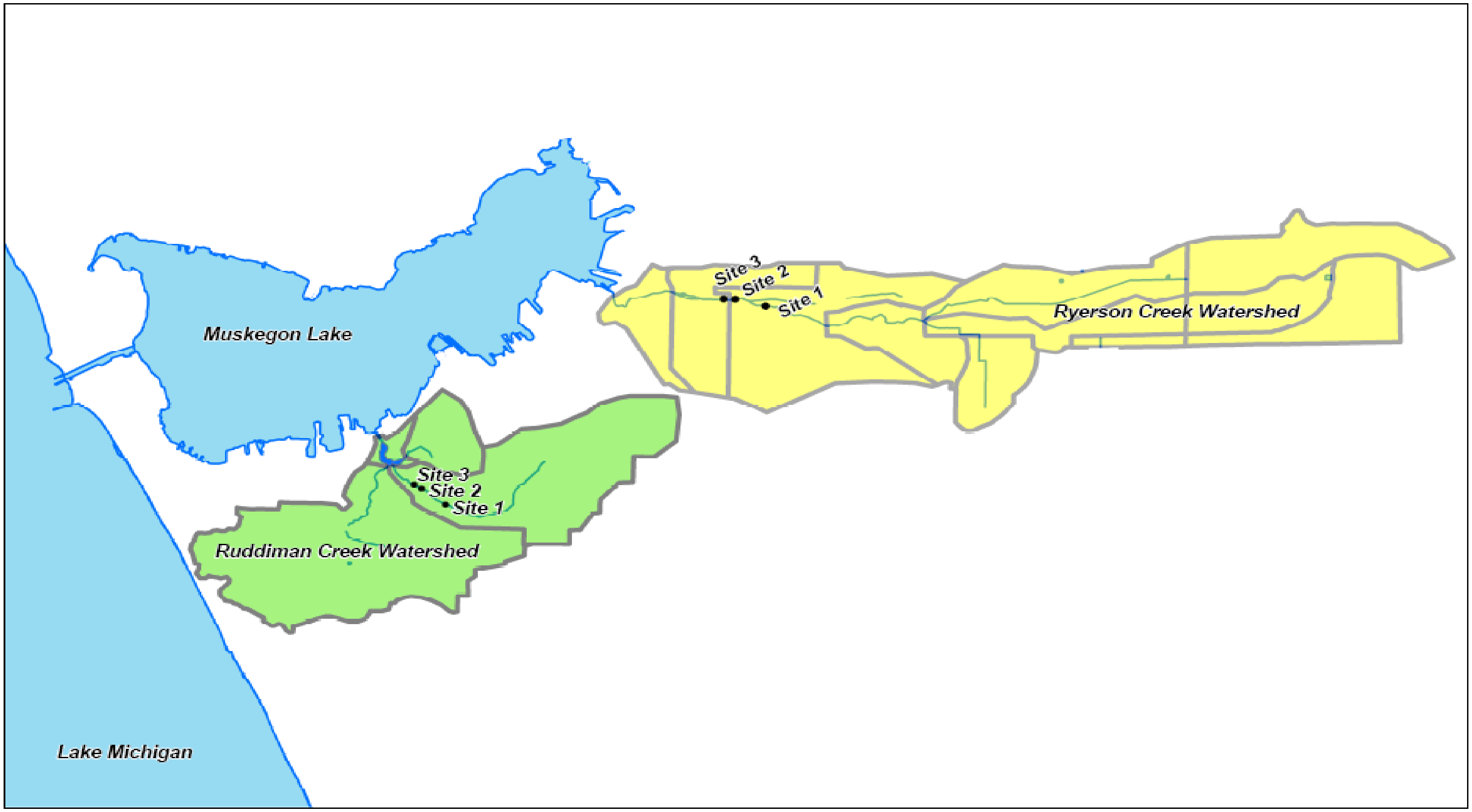


Figure II- 1. Study Area - Map of study and reference sites and major subwatersheds located within the Ruddiman Creek and Ryerson Creek watersheds, Muskegon County, MI, USA.

Table II- 1 Remediation Practices Implemented - Remediation practices implemented between August 2005 and May 2006 at sites 1-3 located on the main branch of Ruddiman Creek. Sites 2 and 3 underwent sediment remediation in December and February, 2006, respectively. Site 1, located upstream of sites 2 and 3, did not undergo sediment remediation, but was affected by installation of upstream structures.

Site	Dredge Depth	Dredge Volume	Replacement Sediment/Materials	Upstream Energy Dissipation Structures	Other Upstream Remediation Practices	Riparian Area Restoration
1	0 m	None	None	Channel armoring; Riprap wing dams (2)	Diversion channel; Retention basin; Sump pit	None
2	0.31 m	176 m^3	Sand (0.15 m); Non-woven geotextile fabric 7.62- centimeter gravel (0.15 m)	Riffle structure	Diversion channel	Seeding
3	0.61 – 1.83 m	931 m^3	Sand (0.15 m); Non-woven geotextile fabric	Riffle structure	None	Seeding

Ryerson Creek served as an urbanized reference (control) stream during this investigation. The watershed (21.0 km^2), located partially within the city of Muskegon, includes similar land use types as Ruddiman Creek. Although Ryerson Creek also has been affected by nonpoint source pollution, it was considered less degraded than Ruddiman Creek in terms of heavy metal and organic chemical contaminants. Three reference sites, with characteristics comparable to those on Ruddiman Creek (i.e. substrate, habitat, and location relative to stream mouth), were located on Ryerson Creek upstream of its confluence with Muskegon Lake.

II.2.2 Macroinvertebrate Sampling

Our study was designed using a Before-After, Control-Impacted (BACI) approach similar to that of Stewart-Oaten *et al.* (1986). Three paired sites were sampled in Ruddiman Creek (treatment) and Ryerson Creek (control) three months prior to remediation (August 2005) and during May and August of both 2006 and 2007. The BACI design helps determine whether changes observed at the treatment site were from the remediation and were not due to temporal variability or regional trends. Replicated samples were collected from available habitat types at each site. Habitats were of three potential types: 1) *Typha*, 2) overhanging riparian vegetation (e.g. *Phalaris arundinacea*, *Impatiens capensis*), and 3) floating/submergent vegetation (e.g. *Elodea canadensis, Nymphaea odorata, Potamogeton foliosus*). The distribution of samples collected at each site was proportional to the amount of available habitat. At Sites 2 and 3 on each stream, three replicate samples were collected for each habitat type since habitats were approximately equally distributed. Because riparian habitat represented 2/3 of available in-stream habitat at Site 1 on each stream, 6 replicates from this habitat, while 3 replicates were collected from the reaming 1/3 floating/submergent habitat (Table 2). Thus, a total of 9 sample replicates were collected from each site on five dates: August 2005 (one month prior to dredging activities), May 2006 (one month after dredging activities), August 2006, May 2007, and August 2007. No more than eight days elapsed between sample collections from Ruddiman Creek and Ryerson Creek during any collection period.

Macroinvertebrates were collected from Ruddiman and Ryerson Creeks following a modified sampling procedure originally developed for wetlands by Burton *et al.* (1999) and Uzarski *et al.* (2004). Replicate samples were collected from all available habitat types at each site. Habitats were of three potential types: 1) *Typha*, 2) overhanging riparian vegetation (e.g. *Phalaris arundinacea*, *Impatiens capensis*), and in-stream floating/submergent plants (e.g. *Elodea canadensis, Nymphaea odorata, Potamogeton foliosus*). The distribution of samples collected at each site was proportional to the amount of available habitat. Three replicate samples were collected for each habitat type at sites 2 and 3 on both streams since habitats were approximately equally distributed. Overhanging riparian vegetation represented 2/3 of available habitat and in-stream floating/submergent vegetation represented 1/3 of available habitat at site 1 on each stream, therefore 6 replicates from the latter habitat and 3 replicates from the former habitat were collected from these sites (Table II-2). Thus, a total of 9 sample replicates were collected from each site on five dates: August 2005 (i.e. three months prior to dredging), May 2006 (i.e. three months after dredging), August 2006, May 2007, and August 2007. No more than 8 days elapsed

Table II- 2. Habitat Types Present at Sampling Sites - Habitat types present at sampling sites located in Ruddiman and Ryerson Creeks.

Stream	Site 1	Site 2	Site 3
Ruddiman Creek	Overhanging riparian vegetation, floating/submergent plants	*Typha*, overhanging riparian vegetation, floating/submergent plants	*Typha*, overhanging riparian vegetation, floating/submergent plants
Ryerson Creek	Overhanging riparian vegetation, floating/submergent plants	*Typha*, overhanging riparian vegetation, floating/submergent plants	*Typha*, overhanging riparian vegetation, floating/submergent plants

between sample collections from Ruddiman Creek and Ryerson Creek during any collection period.

We used D-frame dip nets with a 0.5-mm mesh to collect macroinvertebrate samples. To ensure sampling of microhabitats, sampling involved sweeps from the streambed through the entire water column, while keeping in contact with the vegetation. When present in the immediate vicinity of a sample area, cobbles and stones over 5 cm in diameter were hand washed in dip nets to dislodge invertebrates. The resulting composite samples were emptied into white pans and organisms were picked from each sample replicate for one-half-person-hour. After picking, organisms were tallied and picking continued to the next multiple of 50 unless the nominal maximum of 150 organisms had been reached (Burton *et al.* 1999, Uzarski *et al.* 2004).

If counts were well below 50, 100, or 150 organisms after one-half-person hour, then picking continued until the next multiple of 25 was reached. This ensured that enough organisms were picked to provide a representative sample, but meant that our sampling effort tended to be greater at low densities. Specimens, including semi-aquatic adult insects, were preserved in 70% ethanol and later sorted and identified to family (Hilsenhoff 1988). Exceptions included Oligochaeta and Hydrachnida, which were more difficult to identify, and were identified to order. Taxonomic keys developed by Thorp and Covich (2001), Merritt and Cummins (1996), and other mainstream literature were used for identification.

Since habitat type was not found to be a significant variable in initial Non-Metric Multi-Dimensional Scaling (NMDS) analyses, macroinvertebrate metrics were calculated using cumulative replicate samples pooled across habitat types within each of the three sampling sites per stream, producing three composite sample replicates for each sampling site on each date. Relative abundance percentages were determined for the sampled population and common taxa; abundance values were relative since composite sample replicates contained roughly 75 to 450 total organisms. The percentage of taxa in Ephemeroptera and Trichoptera (ET) was determined and used as an indicator of water quality. Simpson's Diversity Index (1/D) (Magurran 1988), Shannon's Diversity Index (H') (Magurran 1988), and Pielou's Evenness Index (J) (Pielou 1975), were used to assess dominance, richness, and evenness of the macroinvertebrate community. A Family Biotic Index (FBI), as described by Hilsenhoff (1988), was utilized to assess stream conditions based on macroinvertebrate community composition. The FBI is intended to be a rapid, field-based assessment, weighting the relative abundance of each family by its relative tolerance value to determine a total community score. Tolerance values for families of stream arthropods in the western Great Lakes region were used, and supplemented with values from Lenat (1993), Bode (1996), and Barbour (1999). Although finer taxonomic resolution can be more useful, because tolerance can change within a genus (Resh and Unzicker 1975, Hilsenhoff 1987), family level identifications have been used in previous investigations of stream health (Mattsson and Cooper 2006; Linke *et al.* 1999).

II.2.5 Chemical and Physical Measurements

Chemical parameters were recorded at four locations on each stream during macroinvertebrate collection dates and three storm events: September 7, 2007 (2-year storm), June 5, 2008 (2-year storm), and September 4, 2008 (10-year storm). We used a Hydrolab DataSonde 4a to determine temperature, specific conductance, total dissolved solids, pH, oxidation-reduction (redox) potential, dissolved oxygen (DO), and DO saturation (%). Water samples were also collected in 1-liter acid-washed polyethylene bottles and analyzed for chloride, sulfate, nitrate-N, ammonium-N, soluble reactive phosphorus (SRP), total phosphorus (TP), and alkalinity. One duplicate water sample was collected from one randomly chosen location on each stream. Laboratory analytical procedures and quality assurance/control followed recommended procedures outlined in Standard Methods for the Examination of Water and Wastewater (APHA 1998). In addition to duplicate samples, matrix spikes and matrix spike duplicates for all analytes were analyzed at a frequency of 10% with precision limits of ±15% relative standard deviation and accuracy control limits of 90-110% recovery.

Physical habitat measurements included visual estimates of substrate composition, large woody debris cover, and in-stream vegetation cover. Substrate assessments were based on the visible substrate layer, and included estimates of sand, fine and coarse particulate organic matter, and coarse fragments. Substrate parameters were recorded at a randomly selected 0.1-m^2 area within sample replicate locations for macroinvertebrate collections on six dates: November 2005 (i.e. one month prior to remediation), May 2006 (i.e. three months after remediation), August 2006, November 2006, May 2007, and August 2007.

II.2.5 Hydrology

We measured stream discharge on macroinvertebrate sampling dates and storm sampling dates with a Marsh-McBirney Flo-Mate Model 2000 Flow Meter at site 3 in Ruddiman and Ryerson Creeks. In addition to the routine stream discharge measurements, we recorded stage height at the Glenside Boulevard and Clay Avenue road/stream crossings, located on Ruddiman and Ryerson Creeks. These sites were located near the mouth of each stream, had the potential to demonstrate the flashiness of the system, and were not influenced by Muskegon Lake levels. At these sites, we recorded culvert size, slope, and length and surveyed elevations along a downstream cross section. Using this information, Bentley Culvert Master V3.1 program calculated stage elevation over flow rate, based on a sigmoidal relationship. To determine stage height over time, In-Situ Level TROLL 300 data-loggers were placed upstream of each culvert to record water level. A 10-year storm event (i.e. 8.9 centimeters within a 22-hour duration) was recorded on September 4, 2008. Rainfall data were obtained from the Muskegon County Airport weather station (43°10’12”N, 86°14’9”W), located approximately 4 km south of Ruddiman Creek’s study sites. Collected field data allowed us to construct a hydrograph at each site for the 10-year storm event.

II.2.5 Statistical Analysis

Substrate composition data were not statistically analyzed since these data were qualitative and meant to be descriptive only. We tested inter-rater reliability among the three field crews using relative macroinvertebrate abundance data (SPSS version 14.0, Chicago, Illinois). Two-way nested ANOVA with repeated measures were used to analyze macroinvertebrate metrics and morphological data. For these analyses, sampling sites were nested within streams, while collection dates were treated as a repeated measure. One-way ANOVA with repeated measures was used to assess water quality data (SPSS version 14.0, Chicago, Illinois), with stream treated as a fixed effect, collection date treated as a repeated measure, and the four samples taken from different sites on each stream treated as replicates. When sphericity could not be assumed (Mauchly’s test statistic was significant, $p<0.05$) we used the Greenhouse-Geisser corrected F-statistics. If the Greenhouse-Geisser and Huynh-Feldt corrections were not in agreement, the average of their significance values was used to indicate the appropriate correction. Means were compared using Sidak post hoc comparisons (SPSS version 14.0, Chicago, Illinois). Differences were considered significant when $p<0.05$.

Non-metric multidimensional scaling (NMDS) (Clarke 1993) was used to interpret patterns in macroinvertebrate community structure from August 2005 to May 2006. We analyzed means calculated by habitat type for each site. NMDS was completed with the Sorenson (Bray–Curtis) distance measure, 400 maximum iterations, 40 real runs, and 50 randomized runs for the Monte Carlo permutation procedure. The dimensionality of the best solution as determined by NMDS was completed in PC-ORD (version 5.0, Gleneden Beach, Oregon).

II.3 Results

II.3.1 Macroinvertebrate Abundance

The inter-rater reliability test indicated that field crews were consistent in their collection methods; sample replicates did not significantly differ by field crew ($p=0.38$, ANOVA). Relative macroinvertebrate abundance values were comparable between streams prior to stream remediation activities (Figure II-2). While abundance values increased slightly in Ryerson Creek in May 2006 compared to August 2005, abundance significantly decreased in Ruddiman Creek at the first sampling date directly after remediation ($p=0.00$; Sidak post-hoc). This difference can be attributed primarily to site 3, which experienced a greater than 80% decline in abundance (Figure II-4). Abundance values at site 3 remained significantly depressed in comparison to sites 1 and 2 through August 2006, one year into the project investigation ($p<0.05$; Sidak post-hoc). At Ruddiman Creek, abundance at all three sites again declined noticeably in May 2007 following a 2-year storm event, in comparison to the previous collection. At the conclusion of the project investigation, abundance levels in Ruddiman Creek had increased and did not significantly differ from levels at pre-remediation or to those observed in Ryerson Creek ($p<0.05$; Sidak post-hoc).

After sediment remediation of Ruddiman Creek, the mean relative abundance of the Gammaridae population declined, but increased by August 2007 (Figure II-3). Relative abundance of Gammaridae also varied through time in Ryerson Creek, but the magnitude of variation was much less pronounced. Oligochaetes and chironomids, organisms more tolerant of environmental degradation, represented a larger percentage of the sample collected in Ruddiman Creek directly following sediment remediation (Figure II-4). Although this increase occurred at all Ruddiman Creek sites for the Oligochaeta population, this increase was significant only for the chironomid population at site 1 ($p=0.00$; Sidak post-hoc).

Odonates were significantly more dominant in Ruddiman Creek (Coenagrionids followed by Aesnhnids and Libellulids) than in Ryerson Creek (Aeshnids) prior to remediation. While % odonates declined in Ruddiman Creek following remediation, this group significantly increased, in comparison to the reference reach ($p<0.05$; Sidak post hoc), throughout the project investigation especially at site 3 (Figure II-5). Similar to the odonates, the Physidae population inhabiting Ruddiman Creek was significantly more abundant in comparison to Ryerson Creek at pre-remediation ($p=0.00$; Sidak post-hoc). However, the Physidae population, which primarily inhabited Ruddiman Creek sites 1 and 3, declined following remediation and throughout the project investigation (Figure II-6).

The ET percentage of the macroinvertebrate collection, representing the sensitive taxa, included Baetids collected in Ruddiman and Ryerson Creeks, and a nominal number of Trichopterans collected in Ryerson Creek. Prior to remediation, the ET group was comparable ($p = 0.051$; Sidak post hoc) between streams (Figure II-7). After remediation, the ET percentage

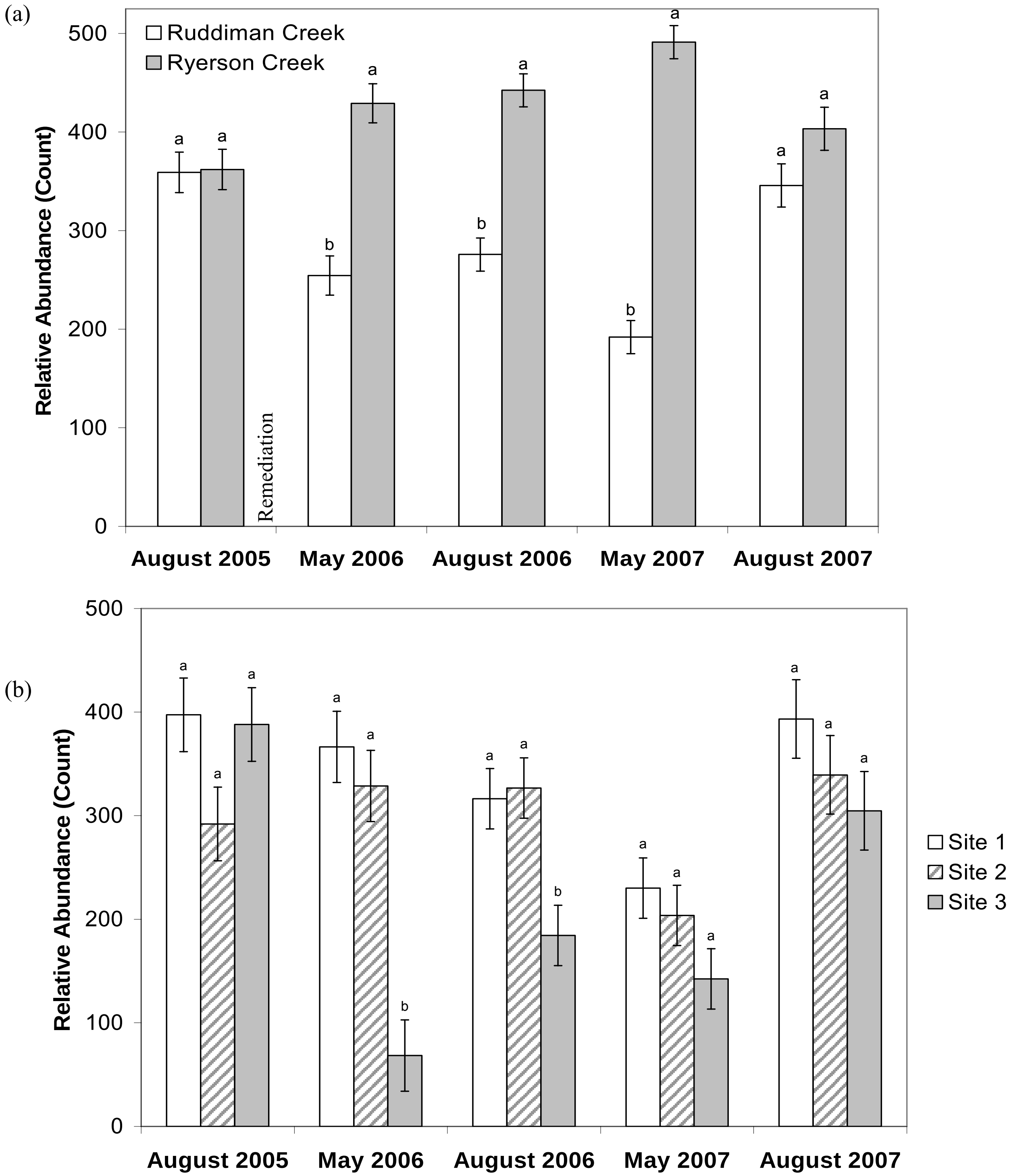
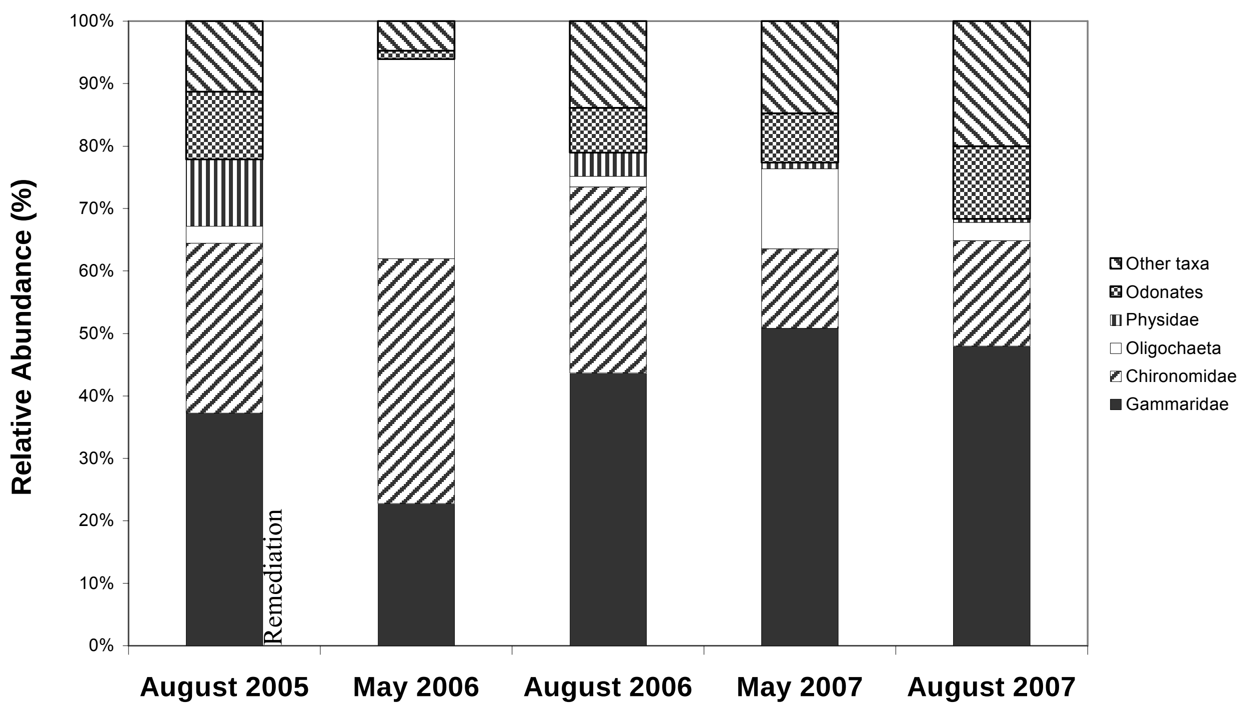


Figure II- 2. **Relative Macroinvertebrate Abundance - Mean (± pooled SE by date) relative abundance (mean counts per composite sample) of macroinvertebrates collected in Ruddiman and Ryerson Creeks (a) and at sites on Ruddiman Creek (b). Bars with different lettering differ significantly by date.**

(a)



(b)

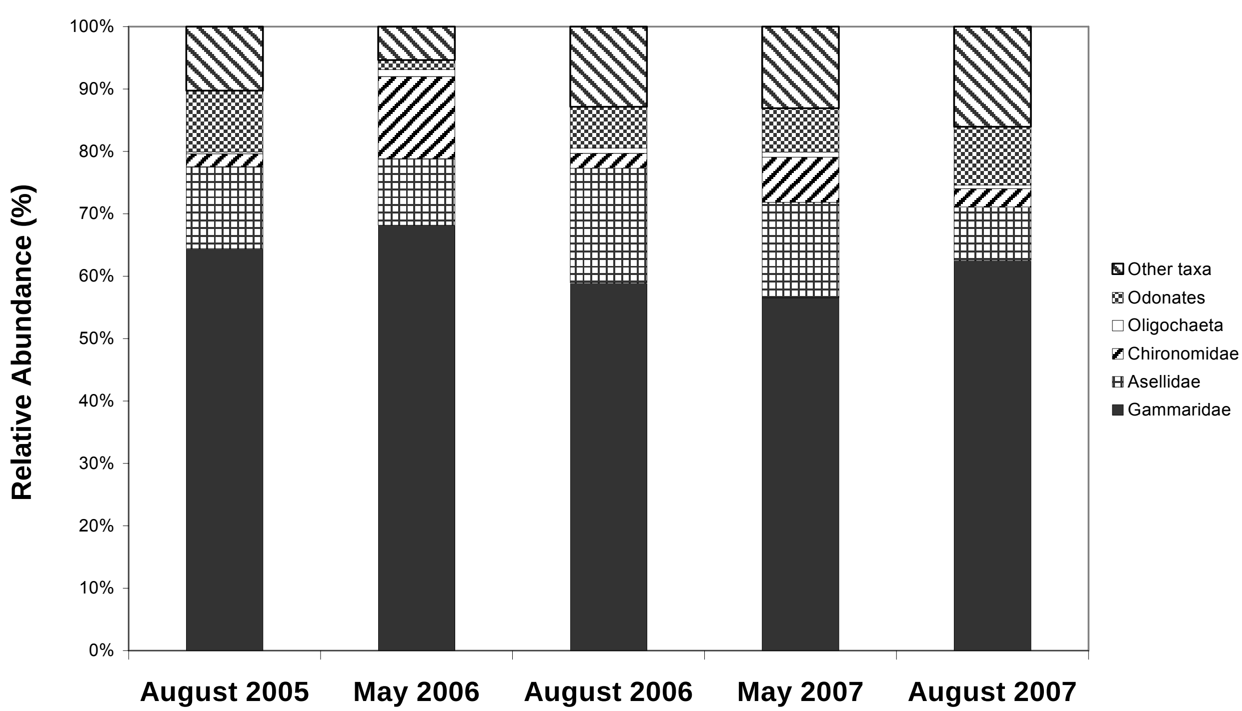
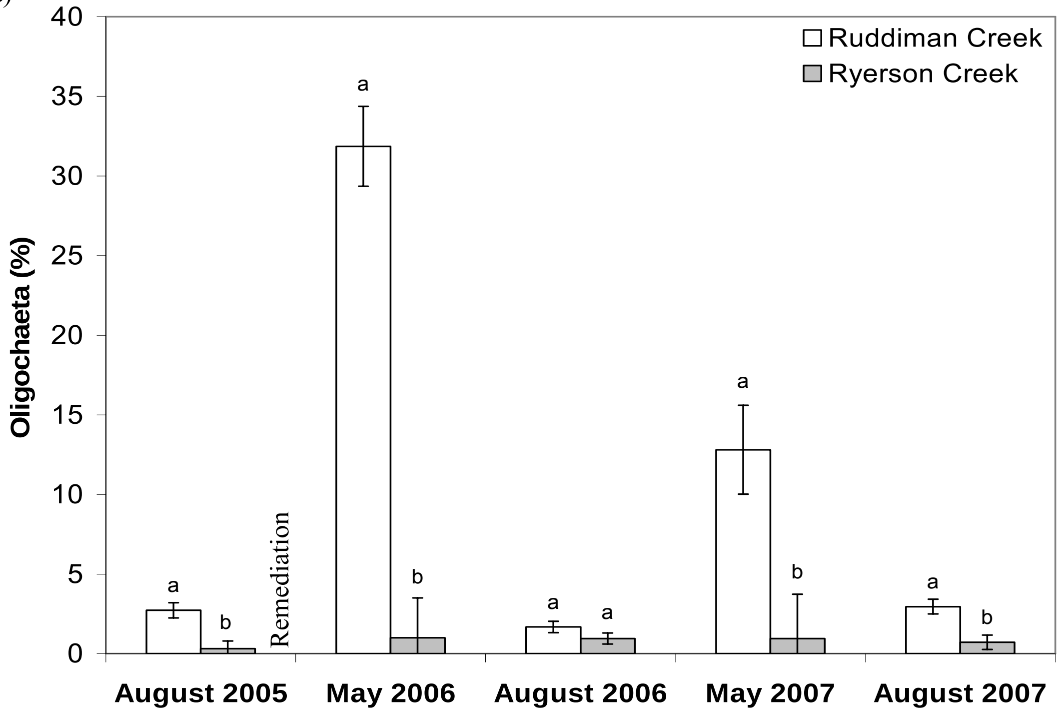


Figure II- 3. **Macroinvertebrate Composition - Mean relative abundance (%) of macroinvertebrate taxa of importance within Ruddiman (a) and Ryerson (b) Creeks.**

(a)



(b)

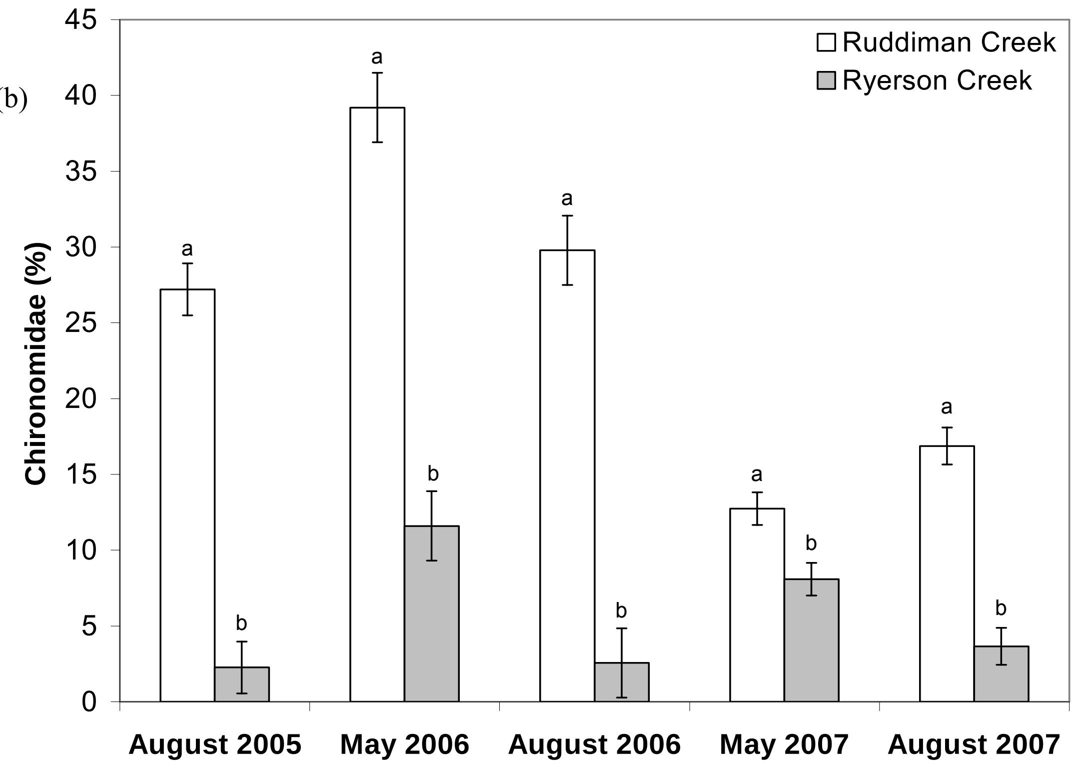


Figure II- 4. **Oligochaeta and Chironomidae Relative Abundance - Mean (± pooled SE by date) relative abundance of Oligochaeta (a) and Chironomidae (b) populations collected from Ruddiman and Ryerson Creeks. Bars with different lettering differ significantly by date.**

(c)

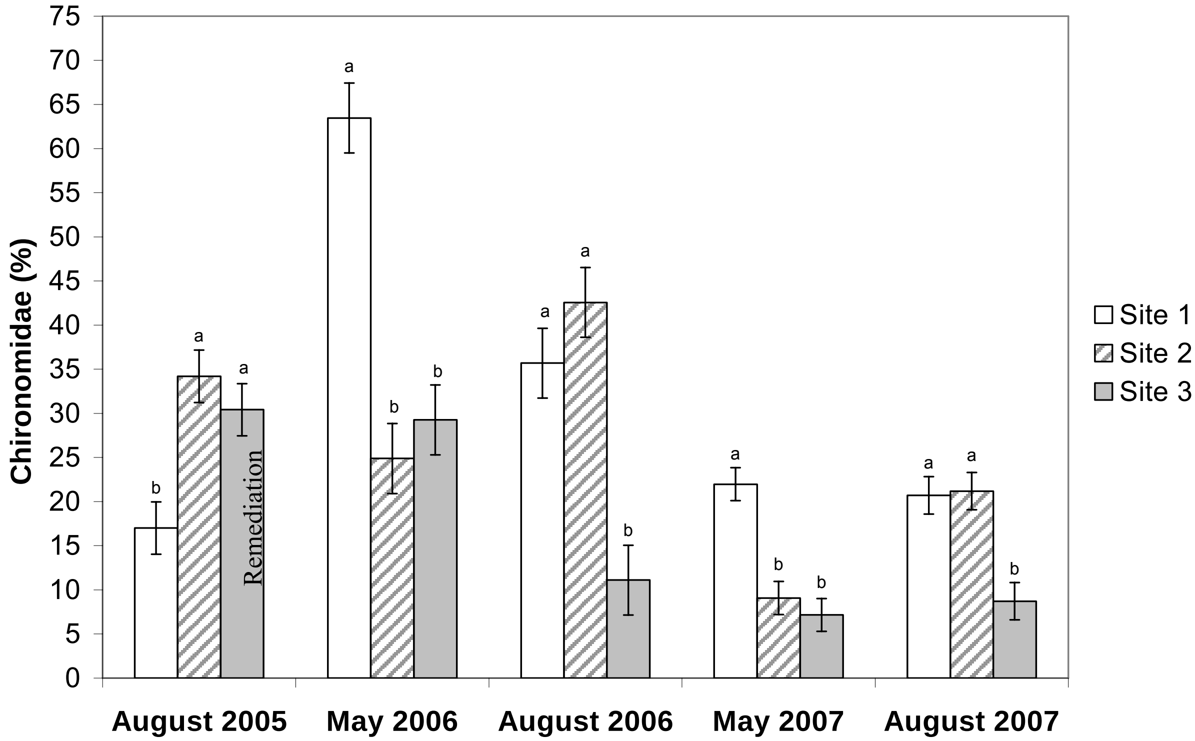


Figure II- 4. Oligochaeta and Chironomidae Relative Abundance Cont'd - Mean (± pooled SE by date) relative abundance of Chironomidae (c) populations collected at sites on Ruddiman Creek. Bars with different lettering differ significantly by date.

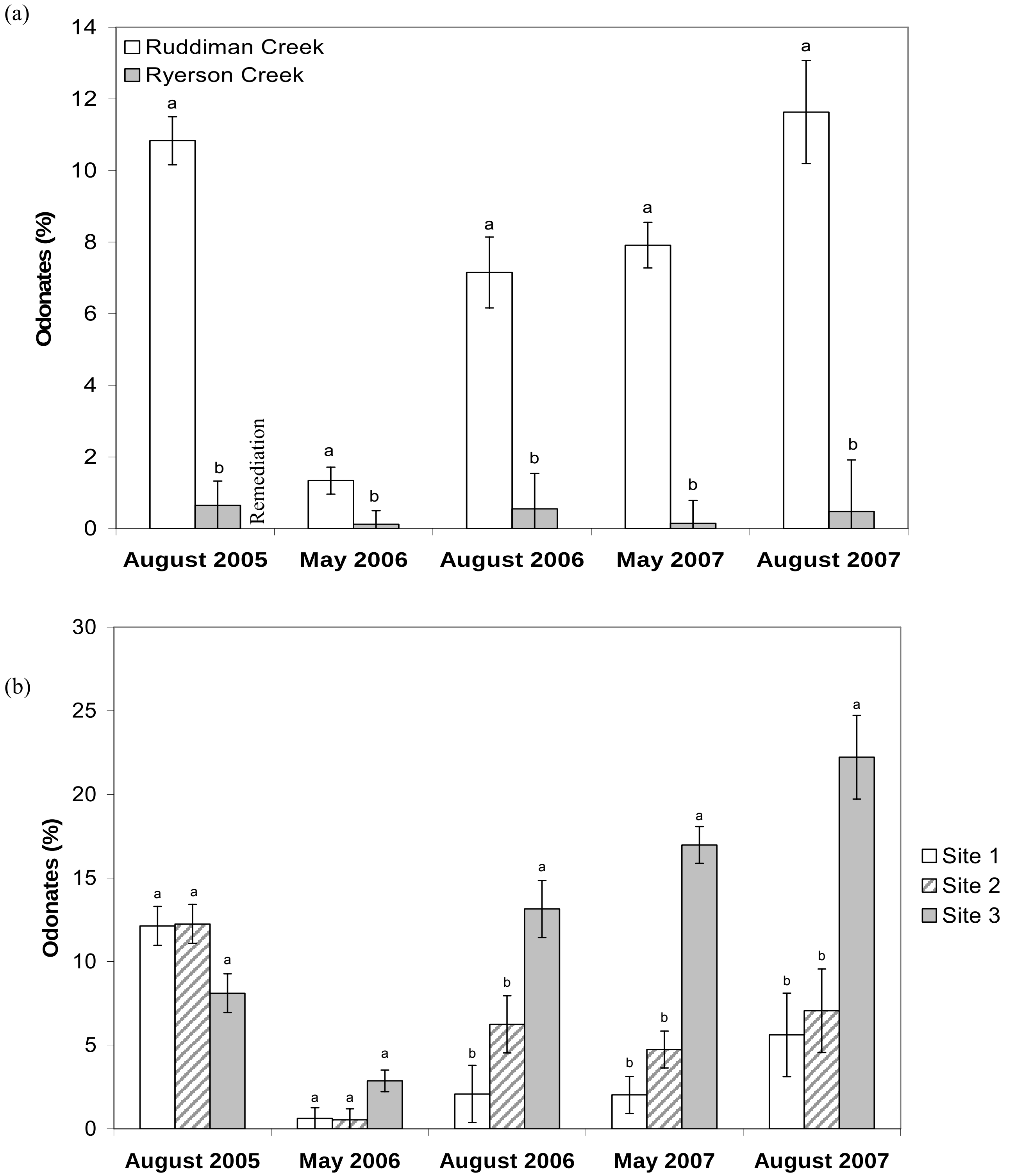


Figure II- 5. Odonata Relative Abundance - Mean (± pooled SE by date) relative abundance of Odonata collected at Ruddiman and Ryerson Creeks (a) and sites on Ruddiman Creek (b). Bars with different lettering differ significantly by date.

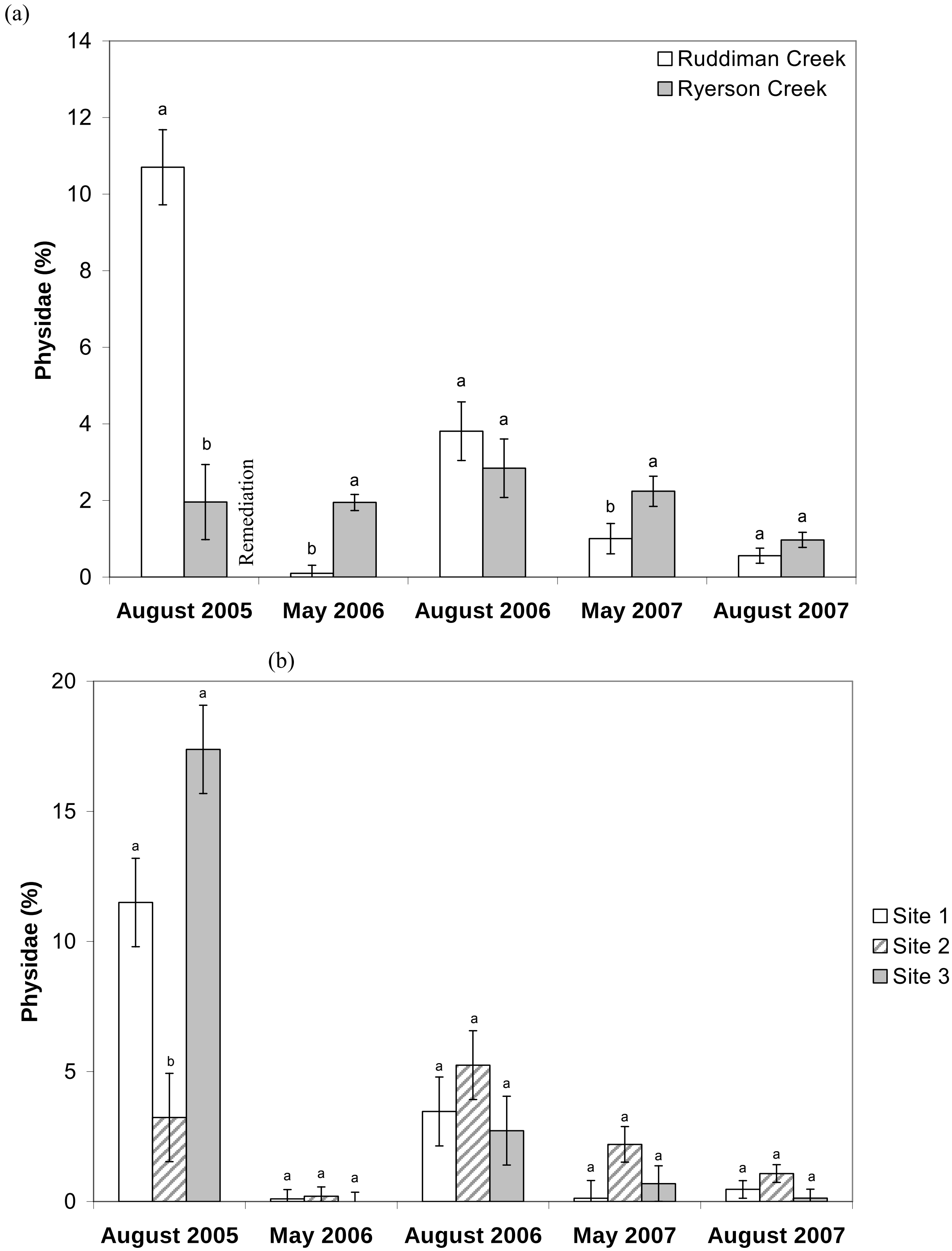


Figure II- 6. Physidae Relative Abundance - Mean (± pooled SE by date) relative abundance of Physidae populations collected at Ruddiman and Ryerson Creeks (a) and at sites on Ruddiman Creek (b). Bars with different lettering differ significantly by date.

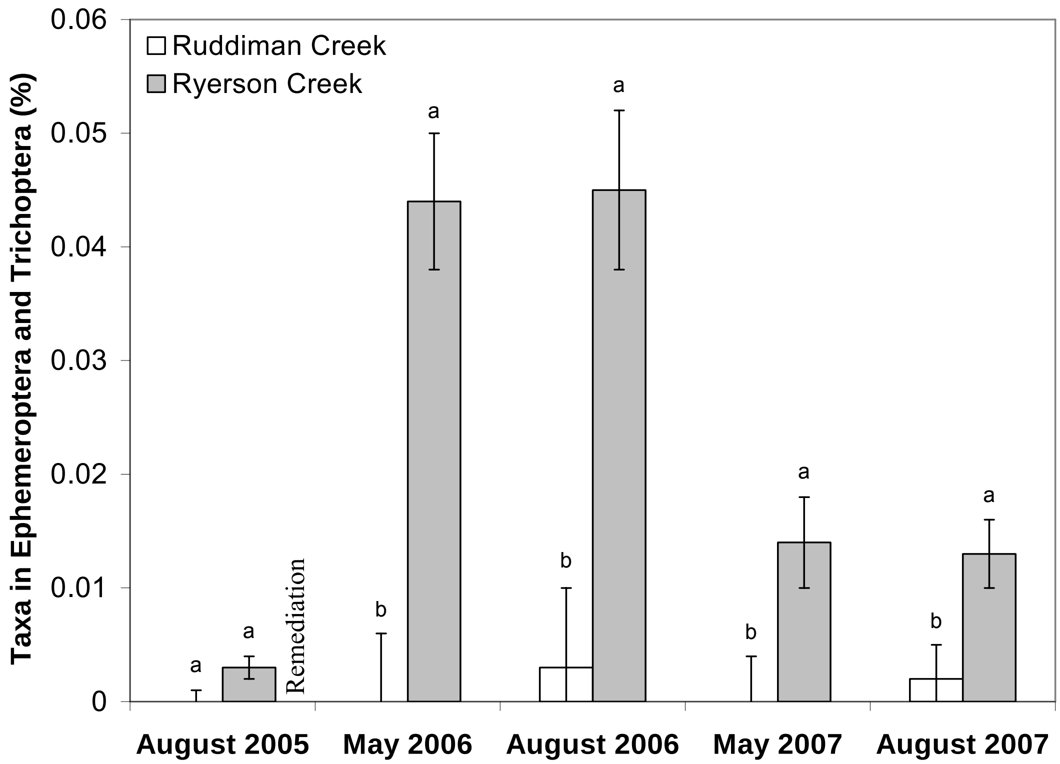


Figure II- 7. Ephemeroptera and Trichoptera Relative Abundance - Mean (± pooled SE by date) relative abundance of Ephemeroptera and Trichoptera populations collected at Ruddiman and Ryerson Creeks. Bars with different lettering differ significantly by date.

in Ryerson Creek increased and remained greater than the ET percentage of Ruddiman Creek throughout the project investigation ($p<0.05$; Sidak post hoc).

II.3.2 Macroinvertebrate Composition

Fifty-two macroinvertebrate taxa representing 4 phyla (i.e. Annelida, Anthropoda, Mollusca, Platyhelminthes) and 8 classes were collected in Ruddiman and Ryerson Creeks during the project investigation. Forty-five taxa were collected in Ruddiman Creek; 17 of which had 5 or fewer organisms and were considered rare. Thirty-nine taxa were collected in Ryerson and 19 were considered rare. In both streams, gammarids dominated sample collections and the three most abundant taxa on any given date accounted for at least 2/3 of the total collection (Figure II-3).

The community composition of Ruddiman Creek, dominated by Gammaridae and Chironomidae at pre-remediation, became dominated by Chironomidae, Oligochaeta, and Gammaridae following remediation (Figure II-3). A 2-dimensional solution was obtained for an NMDS ordination that explained 93% of the variance in the macroinvertebrate distance matrix (Figure II-8). Stream (i.e. Ruddiman Creek vs. Ryerson Creek) explained 70% of the variation in community composition along the x-axis, while site location and sampling date explained 23%

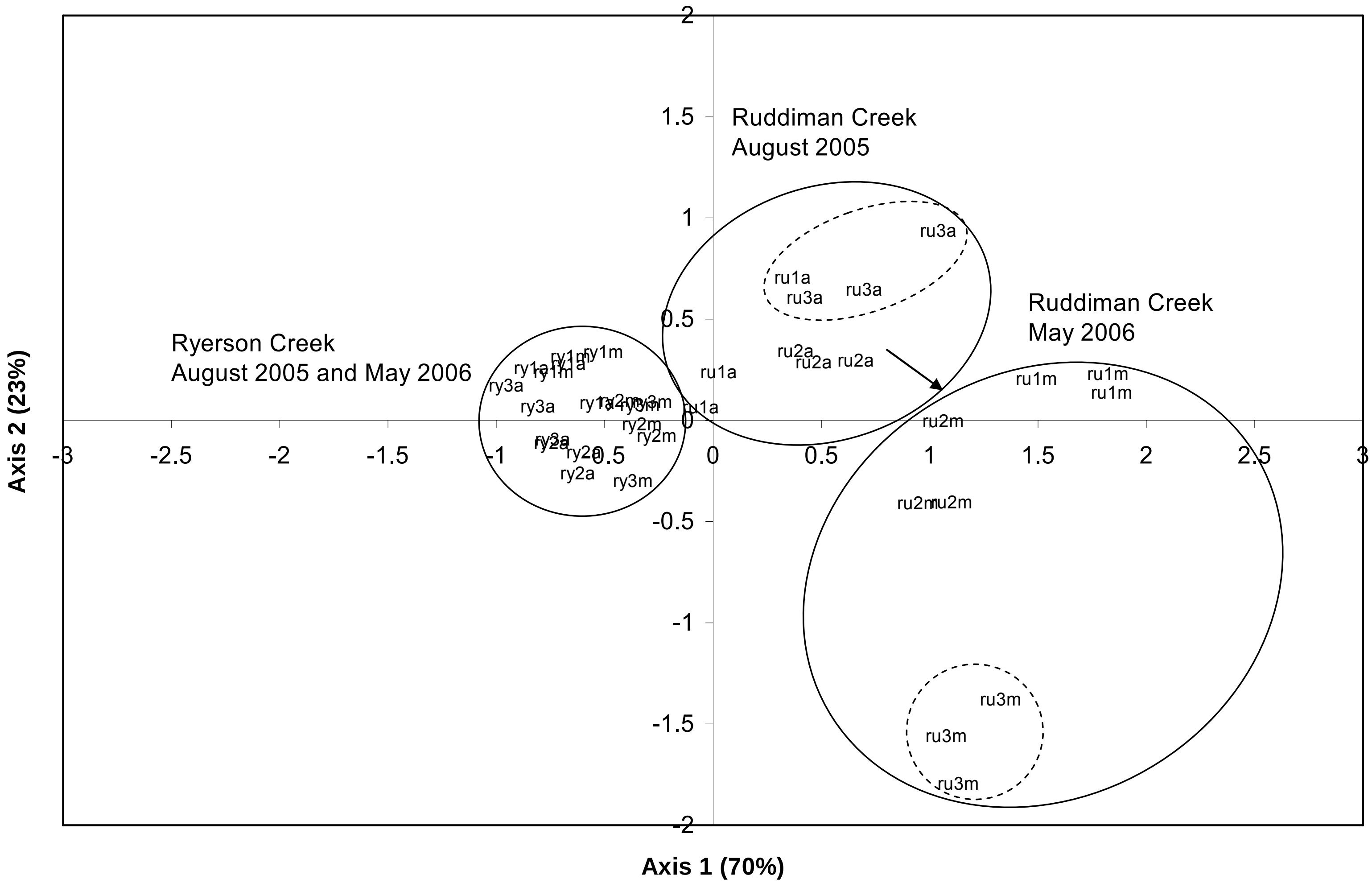


Figure II- 8. **NMDS Plot of Macroinvertebrate Composition - Non-metric multidimensional scaling plot of macroinvertebrate community composition data collected in 2005 August (a) and 2006 May (m) from study and reference sites 1-3 located on Ruddiman (ru) and Ryerson (ry) Creeks. Macroinvertebrate compositions are distinct between streams. Composition of Ruddiman Creek shifts from pre to post-remediation, and macroinvertebrates collected from site 3 of Ruddiman Creek at post-remediation are distinct from the remaining collection.**

of the variation along the y-axis. Habitat type did not explain any of the observed variation. Both axes explained significantly more variance than would be expected by chance based on Monte Carlo permutation tests ($p=0.02$, 50 permutations). NMDS revealed distinct macroinvertebrate community compositions for each stream. In Ruddiman Creek, the composition of all sites shifted over time indicating a change in community structure from pre to post-remediation. Samples collected from site 3 of Ruddiman Creek at post-remediation were distinctly different from the remaining collection. By August 2007 Gammaridae and Chironomidae were again the most abundant taxa, while in Ryerson Creek sites Gammaridae and Asellidae typically dominated samples (Figure II-3).

II.3.3 Macroinvertebrate Diversity

Prior to remediation, the dominance, richness, and evenness of the macroinvertebrate community inhabiting Ruddiman Creek was significantly greater ($p=0.00$; Sidak post hoc) than that of Ryerson Creek (Figure II-9). Simpson's Diversity Index revealed that Ruddiman Creek site 3 was significantly more diverse in taxa than sites 1 and 2 initially ($p<0.05$; Sidak post hoc). However, all three site values declined directly after remediation and were no longer distinguishable from Ryerson Creek sites. When sampled over one year later, values for Ruddiman Creek resembled pre-remediation conditions and were again significantly greater than Ryerson Creek values ($p=0.00$; Sidak post hoc), which fluctuated according to season throughout the project.

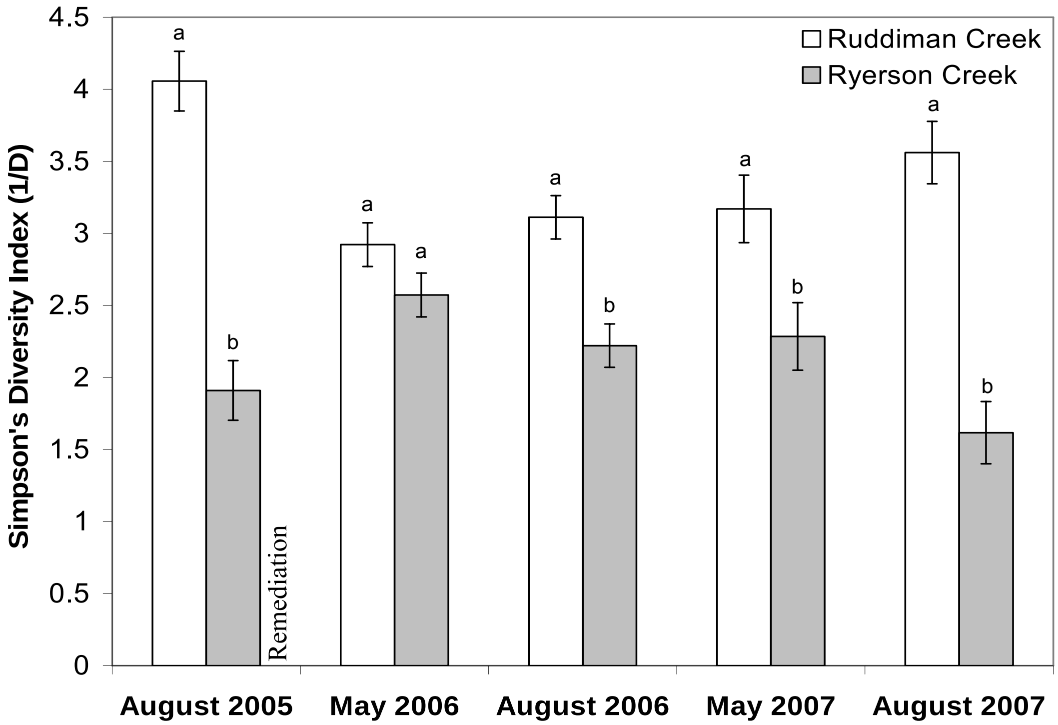
II.3.4 Macroinvertebrate Tolerance

Prior to remediation, FBI values for Ruddiman Creek were significantly higher than FBI values for Ryerson Creek ($p=0.01$; Sidak post hoc), indicating greater degradation, although values for both streams fell within the range designated for fairly poor water quality (Figure II-10). Directly following remediation activities, FBI values for Ruddiman Creek indicated greater degradation, especially at site 2, in comparison to the reference system ($p=0.00$, Sidak post hoc), which remained relatively unchanged. Throughout the remaining project investigation, Ruddiman Creek values were comparable to those of Ryerson Creek overall and no longer differed significantly ($p<0.05$; Sidak post hoc); values for both streams remained in the fairly poor water quality category. At Ruddiman Creek site 3, FBI values fell within the poor water quality range for four of the five sample collections. By the end of the project investigation, however, values for sites 1 and 2 on Ruddiman Creek were significantly improved ($p = 0.00$; Sidak post hoc), while those at Site 3 remained about the same as before remediation.

II.3.5 Chemical and Physical Characteristics

Specific conductance, total dissolved solids, chloride, sulfate, nitrate-N values declined during storm flow conditions as compared to base flow conditions (Table II-3) within the study and reference systems. SRP-P and TP-P concentrations were elevated in correspondence to the increase in discharge between base flow and storm flow. For all collections, TP was below 1 mg/l and DO was above 4 mg/l, the State of Michigan guideline for warmwater fisheries (MDEQ 1986). This also was the case for chloride, sulfate, and nitrate-N concentrations, which met water quality standards: 230 mg/l for ambient chloride, 250 mg/l for sulfate in drinking water, and 10 mg/l for nitrate-N in drinking water (USEPA 1988, 2003).

(a)



(b)

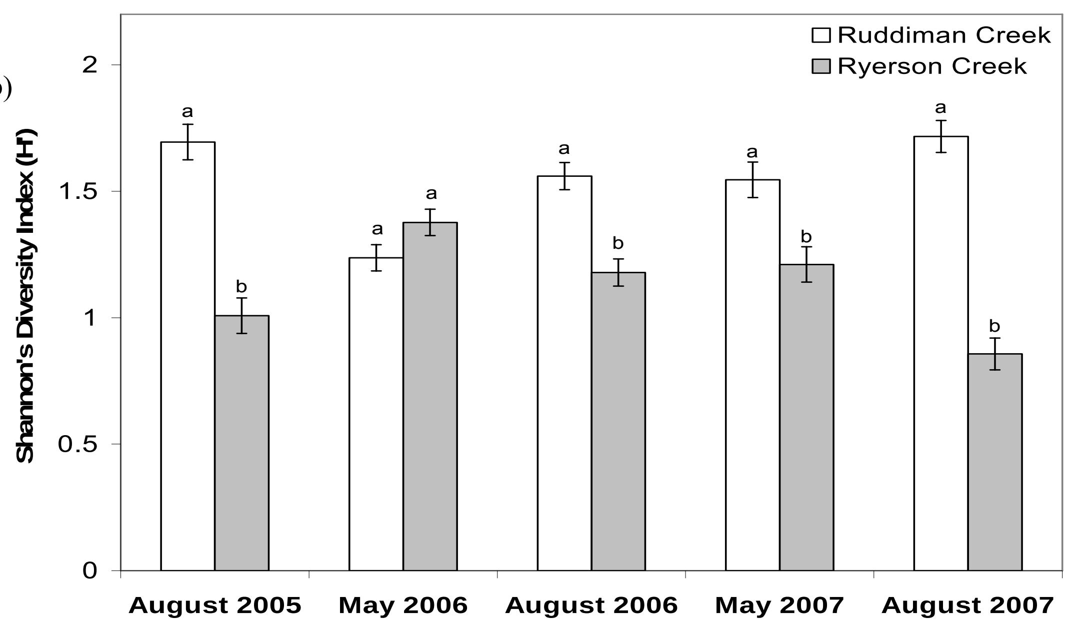
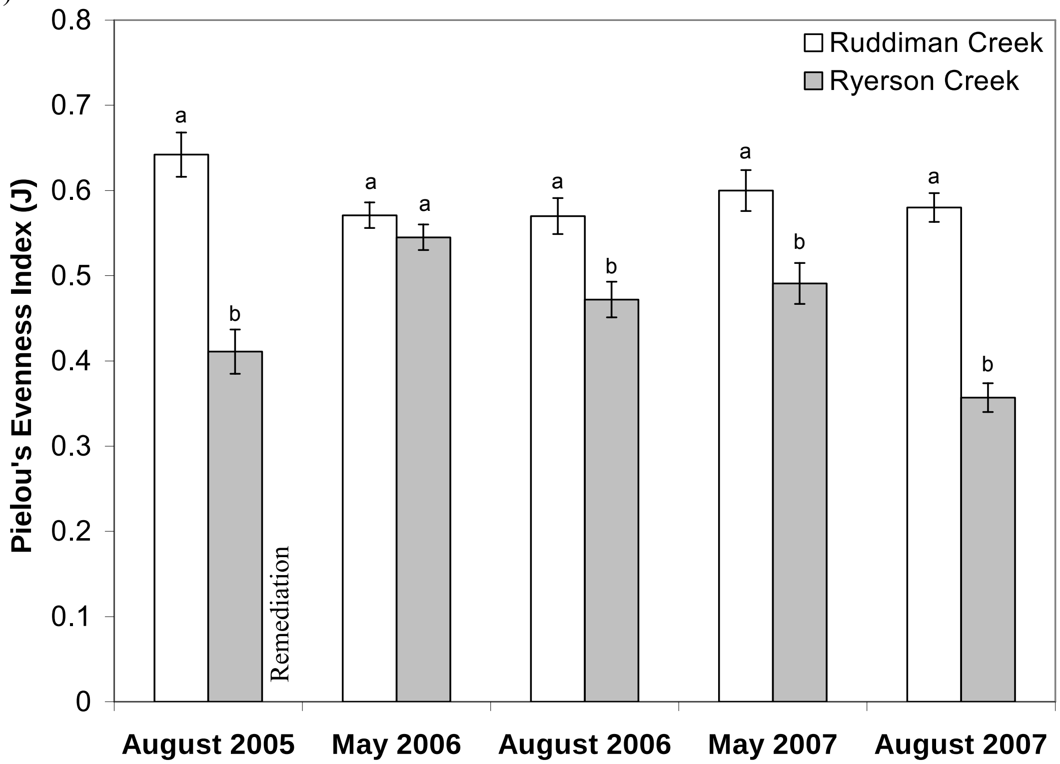


Figure II- 9. **Macroinvertebrate Diversity Indices - Mean (± pooled SE by date) Simpson's Diversity Index (a), Shannon's Diversity Index (b), and Pielou's Evenness Index values (c) for Ruddiman and Ryerson Creeks. A significant site by date interaction was determined**

only for Simpson's Diversity Index values (d). Bars with different lettering differ significantly by date.

(c)



(d)

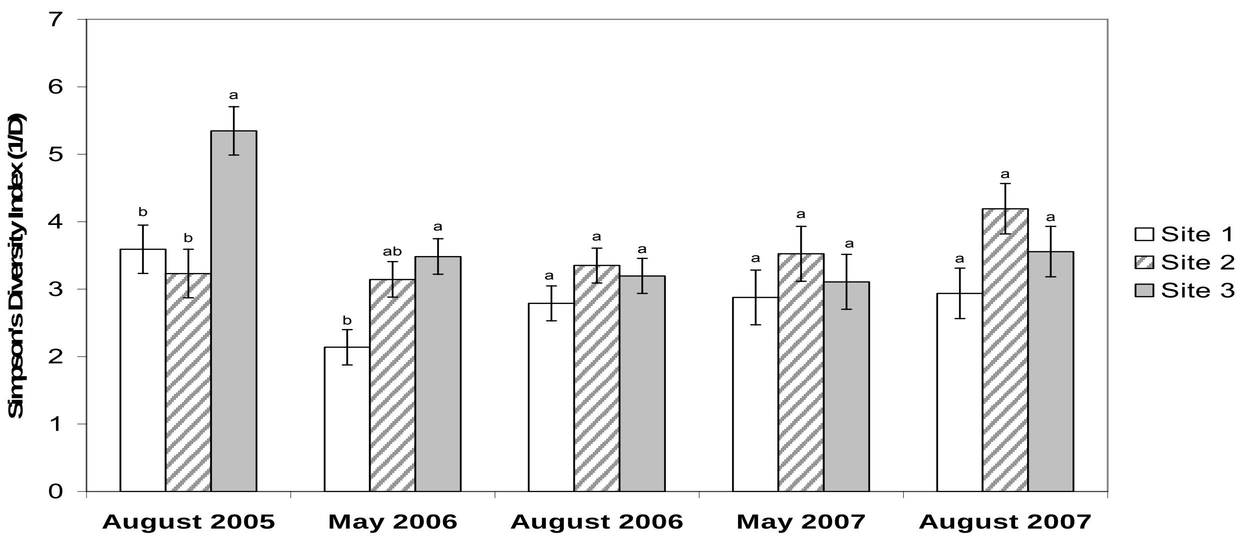


Figure II- 9. Macroinvertebrate Diversity Indices Cont'd. - Mean (± pooled SE by date) Simpson's Diversity Index (a), Shannon's Diversity Index (b), and Pielou's Evenness Index values (c) for Ruddiman and Ryerson Creeks. A significant site by date interaction was determined only for Simpson's Diversity Index values (d). Bars with different lettering differ significantly by date.

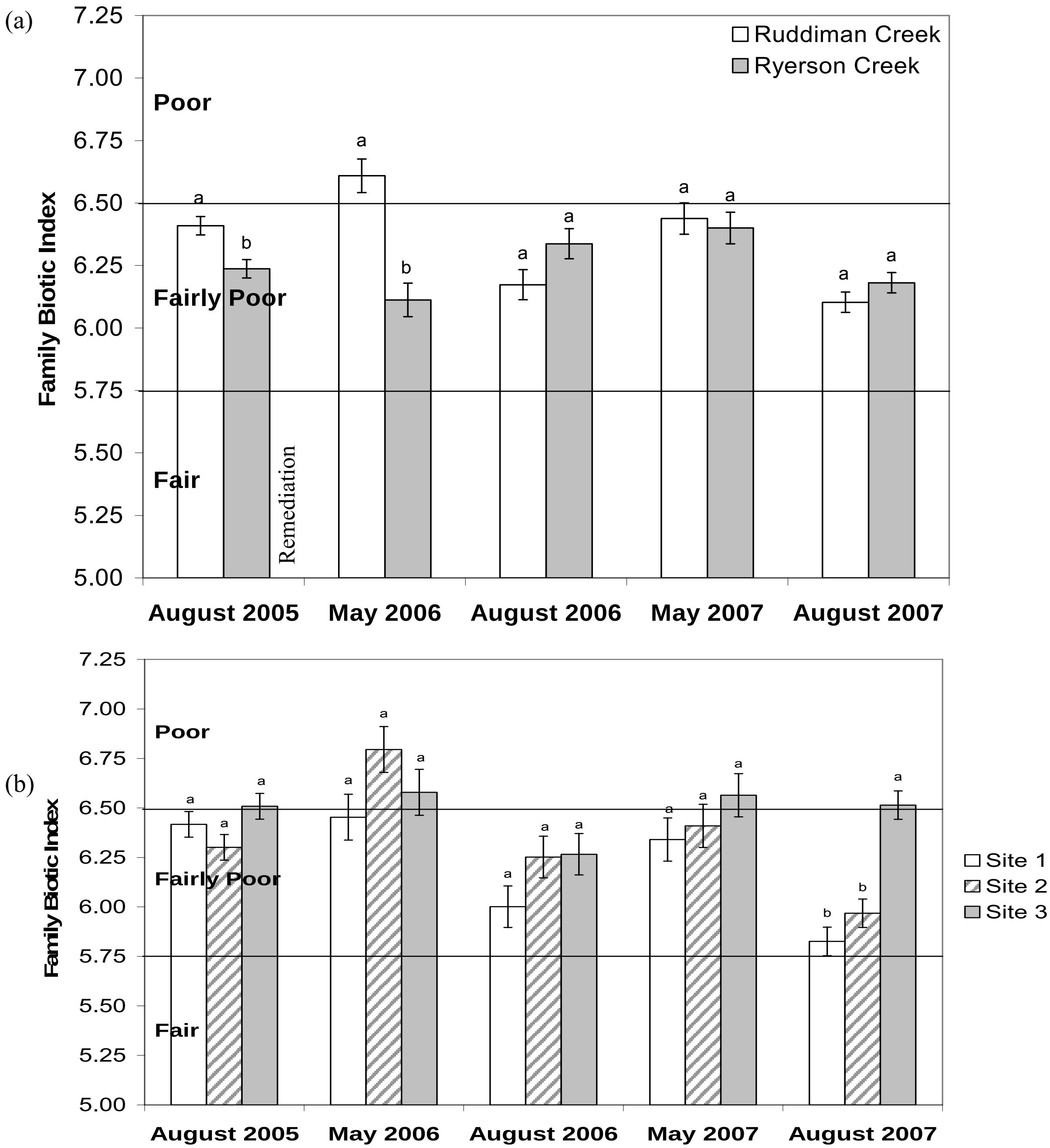


Figure II- 10. Family Biotic Index - Mean (± pooled SE by date) Family Biotic Index (Hilsenhoff 1988) values of Ruddiman and Ryerson Creeks (a) and sites 1-3 on Ruddiman Creek (b) indicating fair, fairly poor, and poor water quality conditions. Bars with different lettering differ significantly by date.

Table II- 3. Water Chemistry Parameters - Mean values of temperature (°C), specific conductance (µS/cm), total dissolved solids (g/L), redox potential (mV), dissolved oxygen percent saturation (DO%), dissolved oxygen (mg/L), chloride (mg/L), sulfate (mg/L), nitrate-N (mg/L), ammonia-N (mg/L), soluble reactive phosphorus (mg/L), total phosphorus (mg/L), alkalinity (mg/L), and pH measured for Ruddiman and Ryerson Creeks. Mean values represent the average four measurements collected from each stream by date.

Stream	Date	Temp	Specific Conductance	TDS	ORP	DO %	DO	Cl	SO4	NO3-N	NH3-N	SRP-P	TP-P	Alkalinity	pH
Ruddiman	8/12/2005	22.00	557.18	0.36	339.00	74.80	6.41	77.56	36.88	0.95	0.10	0.01	0.06	46.71	7.57
Ruddiman	5/22/2006	15.10	990.60	0.63	355.50	109.15	10.64	152.93	58.60	1.56	0.12	0.01	0.04	39.60	8.00
Ruddiman	8/14/2006	20.15	1010.53	0.65	346.50	102.03	9.42	150.75	46.88	1.22	0.07	0.00	0.02	40.91	8.15
Ruddiman	5/21/2007	15.16	1048.25	0.67	331.00	139.70	13.70	194.25	50.75	1.28	0.09	0.01	0.02	39.03	8.10
Ruddiman	8/13/2007	20.20	870.05	0.56	337.50	85.45	7.50	140.75	45.25	1.20	0.16	0.01	0.03	33.20	7.73
Ruddiman	9/7/2007	21.49	135.05	0.09	333.50	79.50	6.91	16.25	8.00	0.43	0.21	0.06	0.12	8.65	6.81
Ruddiman	6/5/2008	17.60	106.18	0.07	313.25	75.85	7.10	12.00	4.50	0.19	0.10	0.03	0.06	34.50	7.03
Ruddiman	9/4/2008	16.28	100.25	0.07	315.83	90.10	8.83	18.25	6.50	0.41	0.17	0.07	0.24	42.00	6.97
Ryerson	8/19/2005	17.24	840.00	0.51	568.25	74.40	6.99	148.36	25.59	1.07	0.03	0.01	0.08	61.60	7.77
Ryerson	5/19/2006	12.61	587.55	0.38	360.75	102.30	10.56	88.74	24.54	0.98	0.10	0.01	0.03	26.00	7.86
Ryerson	8/21/2006	16.65	887.20	0.57	369.00	87.08	8.25	158.75	32.13	1.23	0.06	0.01	0.04	35.40	7.94
Ryerson	5/22/2007	16.25	826.58	0.53	345.00	104.13	9.98	162.25	23.00	0.97	0.10	0.01	0.04	35.10	7.85
Ryerson	8/14/2007	16.24	908.53	0.58	359.75	83.38	7.96	179.50	31.75	1.20	0.06	0.01	0.04	36.60	7.83
Ryerson	9/7/2007	20.47	254.70	0.16	336.00	61.58	5.46	39.25	8.50	0.36	0.07	0.05	0.16	11.60	6.84
Ryerson	6/5/2008	17.33	171.28	0.11	317.00	58.33	5.50	28.50	6.50	0.29	0.06	0.03	0.06	43.00	6.99
Ryerson	9/4/2008	15.51	232.75	0.15	314.75	74.05	7.37	35.75	8.00	0.30	0.06	0.09	0.14	47.25	7.23

The substrate composition at sample sites in Ryerson Creek remained fairly constant throughout the sampling investigation (Figure II-11). The majority of the sand base was overlain with fine particulate organic matter, and to a lesser extent, with coarse particulate organic matter. Coarse fragments (i.e. gravel and cobble) were observed in minimal amounts, typically at sites 2 and 3. Substrate composition was similar in Ruddiman Creek, although values were more variable over time. Coarse fragments (i.e. gravel, cobble and stone), placed in the streambed at dredged locations during remediation, were observed in the greatest amounts at site 2.

Prior to sediment remediation, in-stream vegetation cover at Ruddiman Creek sites was more extensive than at Ryerson Creek sites (p = 0.00, Sidak post hoc) (Figure II-12). In-stream vegetation cover decreased at all Ruddiman Creek sites as a result of remediation activities, After remediation, Ruddiman Creek values fluctuated by season, but never reached pre-remediation values. In comparison, Ryerson Creek values also fluctuated according to season, but remained relatively constant for all 3 sites throughout the project investigation. Large woody debris coverage between streams was comparable over the study period (p = 0.76; ANOVA); however, values in Ruddiman Creek at sites 2 and 3 declined slightly following remediation (Figure II-13).

II.3.6 Hydrology

Hydrographs, developed for road/stream crossings located near the mouths of Ruddiman and Ryerson Creeks, revealed that discharge was greater in Ruddiman Creek than in Ryerson Creek for major storms of similar intensity and duration. On September 4, 2008, discharge began to respond one hour after the accumulation of 0.6 centimeters of rainfall in each stream (Figure II-14). Ruddiman Creek's discharge peaked at 1.26 cm, roughly 1.22 cm above baseflow, while Ryerson Creek's discharge peaked at 0.81 cm, 0.72 cm above baseflow. As rainfall continued throughout the day, Ruddiman Creek's discharge fell quickly following rainfall, while Ryerson Creek drained more slowly.

II.4 Discussion

Habitat modifications were substantial during the sediment remediation of Ruddiman Creek. While substrate composition and large woody debris coverage were comparable between streams three months after remediation (Figures II-11 and 13), vegetation coverage had changed dramatically (Figure II-12). Removal of in-stream and riparian vegetation resulted in a reduction in cover at all sample sites in comparison to Ryerson Creek, which experienced only slight fluctuations at each site. The most dramatic changes in Ruddiman Creek included the removal of a large floating cattail mat at site 3 and several in-stream cattail stands at site 2. As of August 2008, these cattail stands had just begun to reestablish at site 2. Vegetative habitat at site 3 on Ruddiman Creek may never reestablish to pre-remediation conditions due to increases in streambed depth, ultimately decreasing sunlight penetration to the streambed. A reduction in vegetative coverage also occurred at site 1, which did not undergo sediment remediation. The dewatering of the upstream remediation area and installation of the retention basin may have modified the hydrologic regime during remediation and influenced vegetative growth. Sediment

(a)

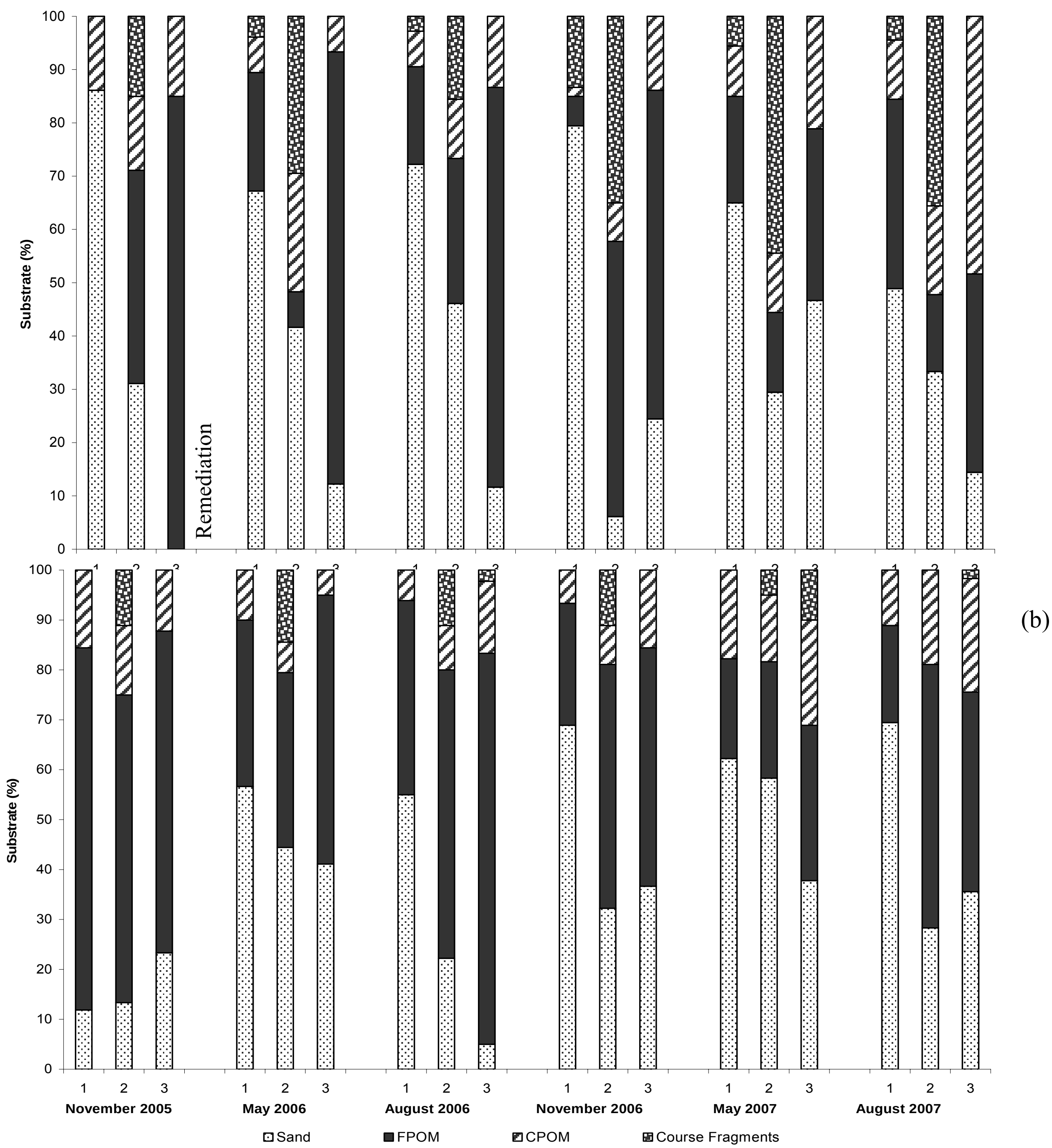
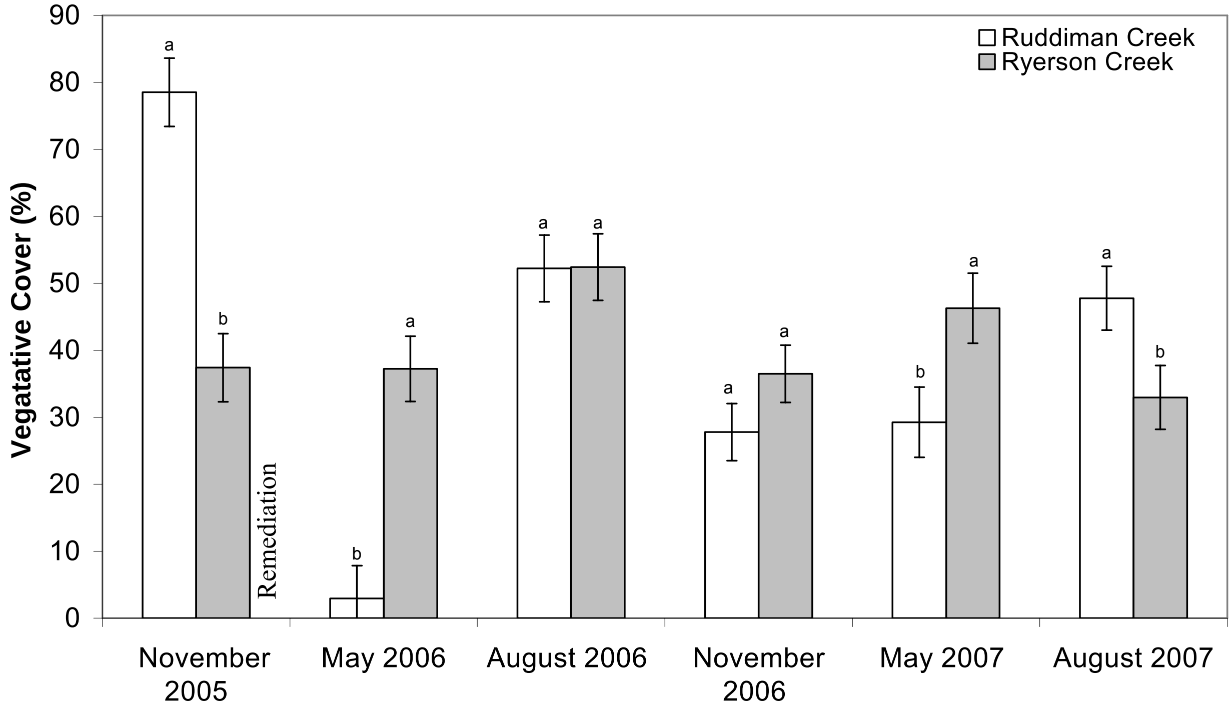


Figure II- 11. **Substrate Composition - Mean substrate composition for Ruddiman (a) and Ryerson (b) Creeks, including sand, fine particulate organic matter (FPOM), coarse particulate organic matter (CPOM), and coarse fragments**.

(a)



(b)

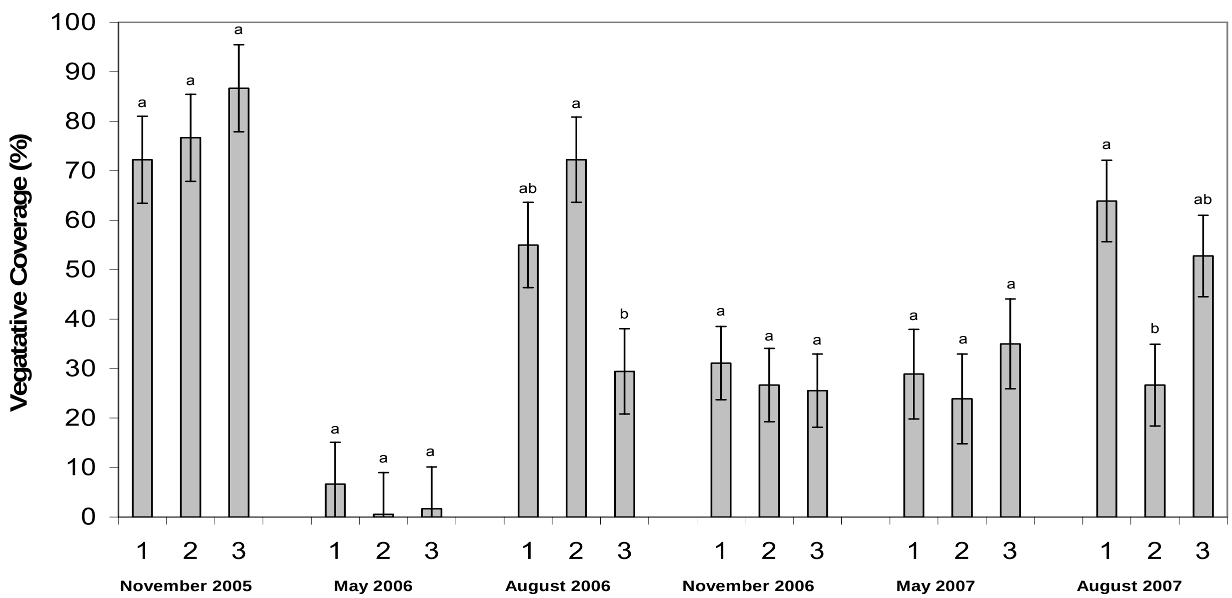
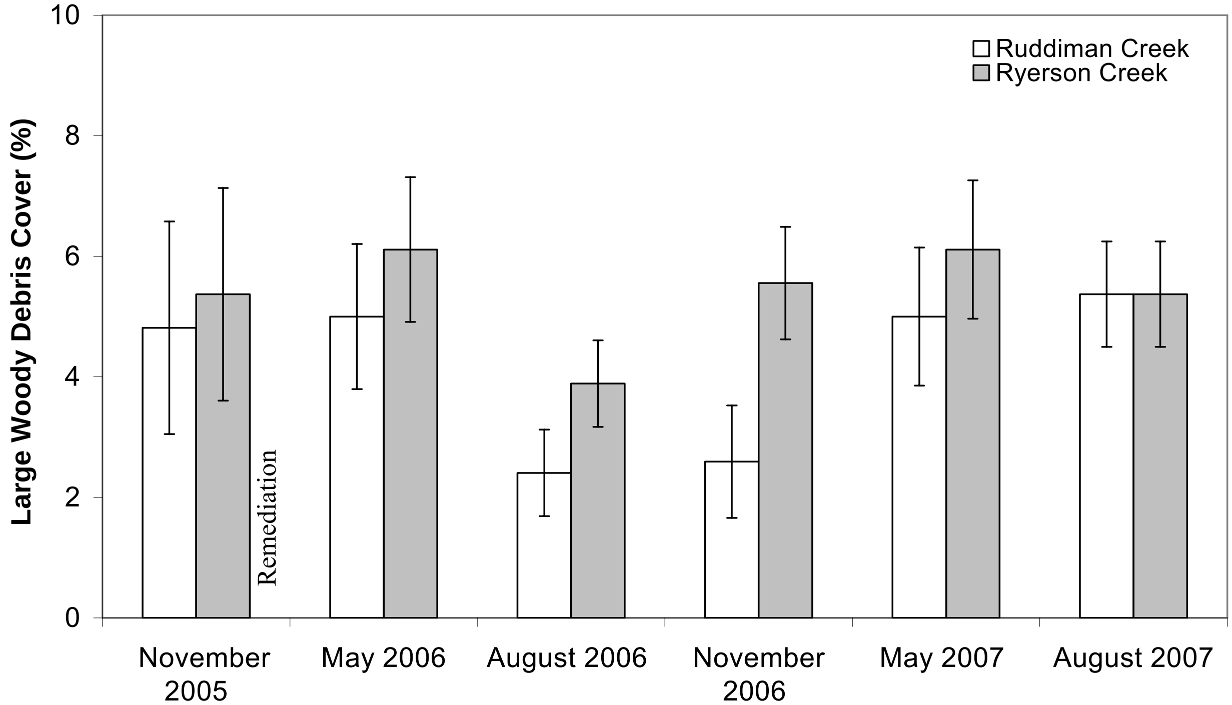


Figure II- 12. **Vegetative Coverage - Mean (± pooled SE by date) vegetative cover within Ruddiman and Ryerson Creeks (a) and sites on Ruddiman Creek (b). Bars with different lettering differ significantly by date.**

(a)



(b)

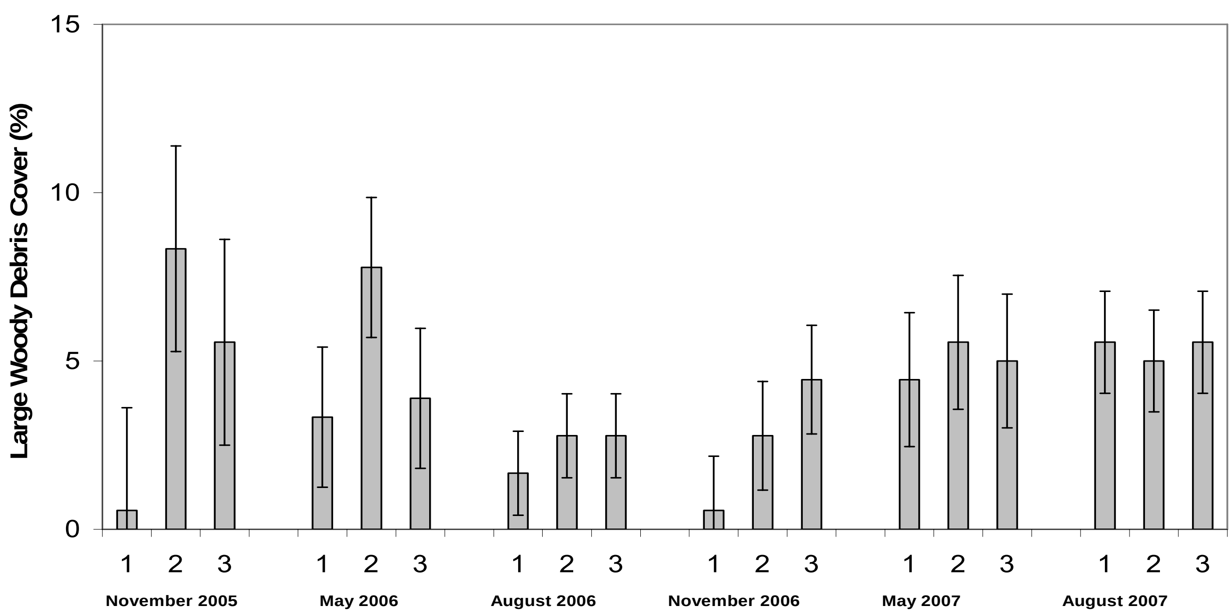
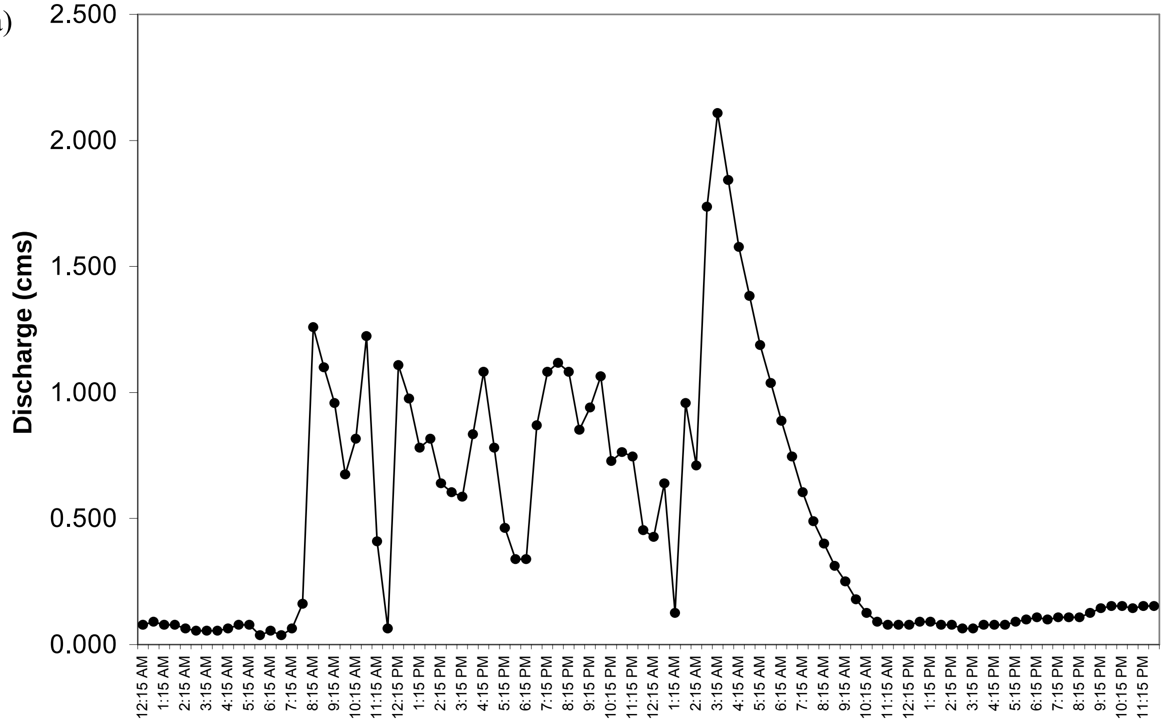


Figure II- 13. Large Woody Debris Coverage - Mean (± pooled SE by date) large woody debris cover within Ruddiman and Ryerson Creeks (a) and sites on Ruddiman Creek (b). There are no significant stream by date or site by date interactions.

(a)



(b)

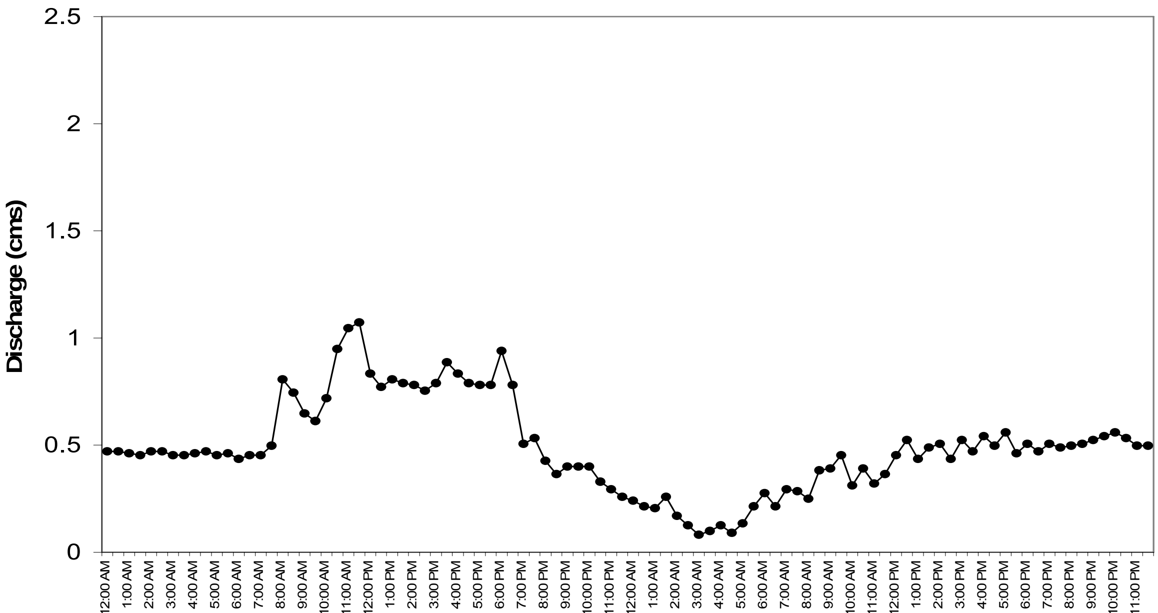


Figure II- 14. **Measured Hydrographs - Hydrographs for downstream locations on Ruddiman (a) and Ryerson (b) Creeks showing a response to a 10-year storm event (8.9 centimeters).**

(c)

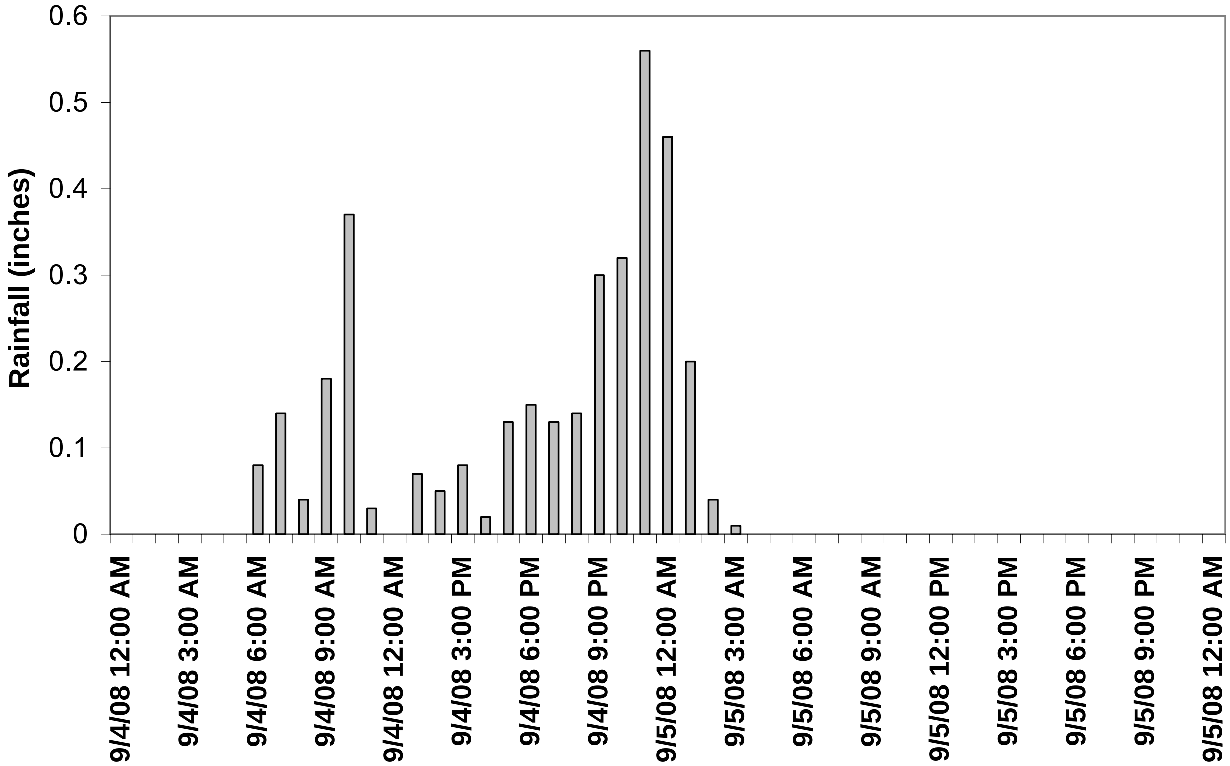


Figure II- 14. Measured Hydrographs Cont'd - Rain event occurring over a 22-hour duration on September 4 and 5, 2008 (c).

disturbed during upstream construction may have also adversely affected vegetation growth at site 1.

Macroinvertebrate community structure changed considerably at Ruddiman Creek as a result of sediment remediation activities (Figure II-3). Ruddiman Creek experienced a significant decrease in relative macroinvertebrate abundance and sensitive macroinvertebrate taxa in comparison to Ryerson Creek. The Gammaridae, Physidae, and Odonata populations experienced the greatest reductions, and subsequently, Chironomidae and Oligochaeta represented a larger portion of sample collections. This was especially the case at site 3, where macroinvertebrate abundance was significantly reduced compared to sites 1 and 2 during the May and August, 2006 collections (Figure II-2). Following remediation, abundance values steadily improved throughout the project investigation, but the May 2007 sample collection was an exception to this trend. Prior to this sample collection, Ruddiman Creek had experienced high stream flows, as indicated by freshly undercut banks and flattened riparian vegetation. It appeared that hydrological fluctuations had the potential to influence macroinvertebrate community structure through habitat modifications. Over one year after sediment remediation, macroinvertebrate community composition was comparable to pre-remediation conditions (Figure II-3), with the exception of the Physidae population (Figure II-6), which had not recovered. The ET population, representing the sensitive taxa of the system, remained minimal throughout the two-year project period (Figure II-7), suggesting the overall system remained impaired.

Diversity indices indicated that the macroinvertebrate community of Ruddiman Creek was significantly more diverse than Ryerson Creek prior to remediation, especially at site 3 (Figure II-9). However, based on the FBI score and minimal ET percentage at pre-remediation, it is clear that Ruddiman Creek was diverse in mainly tolerant taxa. Following remediation activities, Ruddiman Creek's diversity declined, and was comparable to Ryerson Creek. An immediate decline in macroinvertebrate diversity after sediment remediation has been previously documented (Quigley and Hall 1999; Gilkinson *et al.* 2005). During the period following remediation, Ruddiman Creek sites again became significantly more diverse than Ryerson Creek sites, but did not exceed initial diversity values. Macroinvertebrate diversity of Ryerson Creek fluctuated according to season throughout the project investigation.

As expected, FBI values suggested that initial water quality conditions of Ruddiman Creek were significantly more degraded than Ryerson Creek (Figure II-10). Macroinvertebrate community composition indicated that both streams experienced fairly poor water quality and hydrologic conditions during the project investigation. The one exception occurred directly after the dredging of Ruddiman Creek when an increase in tolerant taxa indicated poor water quality conditions in Ruddiman Creek. Following remediation, conditions within Ruddiman Creek quickly approached those of Ryerson Creek, where water quality remained fairly poor. Notably, site 3 on Ruddiman Creek experienced poor water quality conditions for the majority of the sample investigation, indicating that this site was more degraded than sites 1 and 2, despite the initial high level of diversity (Figure II-9). Ruddiman Creek sites 1 and 2 generally showed improvement in FBI scores following remediation, with the exception of the May 2007 sample date when the observed 2-year storm event occurred.

The hydrograph developed for Ruddiman Creek's main branch illustrated the flashiness of the system (Figure II-14). A flashy watershed is considered to be one with stream hydrographs that peak early and have relatively large peak flows. Ruddiman Creek exhibited this pattern, while the hydrograph for Ryerson Creek did not. Storm sewer inputs from the city of Muskegon and surrounding municipalities most likely account for the flashiness observed in Ruddiman Creek's main branch. The Ryerson Creek watershed, although urbanized on its western portion, had a significantly greater percentage of forests, open land, and farm land in the headwater region and very limited storm sewer inputs, helping to explain the hydrologic differences between these two systems. While intermediate disturbance may increase diversity, intense disturbance can be detrimental depending on its degree of severity, frequency, intensity, duration, and area of impact. The Intermediate Disturbance Hypothesis (Connell 1978) proposes that biodiversity is highest when disturbance is neither too rare nor too frequent. As evident from the relative macroinvertebrate abundance and FBI data, Ruddiman Creek's frequent hydrologic fluctuations have the potential to significantly degrade the aquatic ecosystem, which has also been shown by Marsalek *et al.* (2001) and Pitt (2003). During these elevated flows, water velocity typically increases and the stream bed is scoured (Leopold *et al.* 1964). These conditions can damage aquatic habitat (Scullion and Stinton 1983; Gurtz *et al.* 1988), and damage or dislodge invertebrates (Sagar 1986). During the May 2007 collection, which followed a 2-year storm event, the number of organisms collected for many of the major taxa groups (e.g. Gammaridae, Chironomidae, Physidae, and Odonata) declined as compared to the previous collection. Macroinvertebrate data collected for Ryerson Creek one day after the Ruddiman Creek collection in May, 2007 did not follow this trend. In fact, macroinvertebrate abundance in May collections was consistently greater than August collections in Ryerson Creek throughout the project.

Increases in stream discharge, as a result of storm events, led to a dilution of specific conductance, total dissolved solids, chloride, sulfate, and nitrate-N, in comparison to base flow conditions in both streams (Table II-3). SRP and TP concentrations were greater during these events indicating that these nutrients were introduced into the system eroded sediment or surface runoff. Despite these elevated concentrations during storm flows, chemical parameters did not exceed State or Federal water quality standards. In fact, DO, often indicative of water quality degradation, did not significantly differ between streams directly following remediation activities (Table II-3). Other chemical parameters that did differ in May 2006 illustrated less dramatic differences as a result of remediation disturbance than from storm flow disturbances. It seems that water chemistry parameters were not indicative of the disturbances to the ecosystem and that changes to the habitat structure, rather than changes in water chemistry parameters, were responsible for the apparent stream degradation we observed immediately after remediation in Ruddiman Creek. Clearly, water quality data alone could not lead to the conclusions we made using measurements of macroinvertebrate community structure. The results of our study point to the necessity of bioassessments in augmenting our understanding of the aquatic environment and stream health.

Ruddiman Creek was remediated due to elevated levels of heavy metals and organic chemicals. Our investigation sought to assess the impacts on the macroinvertebrate community from dredging and removal of contaminated sediment. However, when physical and chemical disturbances are present in streams, physical factors have been shown to have a more dominant role in structuring the macroinvertebrate community. Peeters *et al.* (2001) found that physical/environmental variables had a stronger influence than metals and Polycyclic Aromatic Hydrocarbons in the Rhine-Meuse delta region of the Netherlands. The strong influence of environmental factors also was noted by Carew *et al.* (2007) during an evaluation of the effects of metal pollution on Chironomid communities. While anthropogenic pollutants can impact the macroinvertebrate community (Pollard and Yuan 2006; Doi 2007), our data show that it is difficult to attribute contaminant impacts to an impaired location when physical disturbances, such as extreme hydrologic fluctuations, are present.

II.5 Conclusions

Our investigation evaluated the success of the Ruddiman Creek remediation project in terms of its impact on the composition, relative abundance, and diversity of the macroinvertebrate community. This investigation concluded that although remediation activities resulted in a significant initial decline in macroinvertebrate abundance and diversity, the community recovered to pre-remediation conditions quickly. Despite this rapid recovery, the macroinvertebrate community at the most heavily remediated site on Ruddiman Creek remained in a degraded state comparable to pre-remediation conditions. In contrast, the community-level

FBI suggested some improvement in the overall condition of the two upstream sites on Ruddiman Creek. Results of our study suggest that hydrologic impairments continue to impact the macroinvertebrate community after remediation and that additional restoration actions are necessary to improve the ecological integrity of Ruddiman Creek. Future implementation activities will need to consider and address the complex factors associated with degraded water quality, altered hydrology, and sediment contamination to effectively achieve overall improvement within this urban system.

II.6 References

Adams, S. M., Ryon, M. G., and Smith, J. G. 2005. Recovery in diversity of fish and invertebrate communities following remediation of a polluted stream: investigating causal relationships. In: Developments in Hydrobiology, Aquatic Biodiversity II (Segers, H., and Martens, K., Editors). Springer Netherlands. 180: 77-93.

American Public Health Association (APHA). 1998. Standard Methods for the Examination of Water and Wastewater, 20th edition. APHA, Washington, DC.

Barbour, M.T., Stribling, J.B., and Karr, J.R. 1995. Multimetric approach for establishing biocriteria and measuring biological condition. In: Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making (Davis, W.S., and Simon, T.P., Editors). Lewis Publishers, Boca Raton, Florida, USA. pp. 63-77.

Barbour, M.T., Gerritsen, B.D., Snyder, B.D., and Stribling, J.B. 1999. Rapid bioassessment protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency, Office of Water, Washington, D.C.

Bennett, H.H., Mullen, M.W., Stewart, P.M., Sawyer, J.A., and Webber, E.C. 2004. Development of an invertebrate community index for an Alabama coastal plain watershed. Journal of the American Water Resources Association. 40(1): 43-55.

Bode, R. W., Novak, M.A., and Lawrence, E.A. 1996. Quality assurance work plan for biological stream monitoring in New York State. Stream Biomonitoring Unit, Bureau of Monitoring and Assessment, Division of Water, New York State Department of Environmental Conservation, Albany, New York. 89 pp.

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

Broekhuizen, N., Parkyn, S., and Miller, D. 2001. Fine sediment effects on feeding and growth in the invertebrate grazers *Potamopyrgus antipodarum* (Gastropoda, Hydrobiidae) and *Deleatidium* sp. (Ephemeroptera, Leptophlebiidae). Hydrobiologia. 457: 125-132.

Camargo, J.A. 1991. Toxic effects of residual chlorine on larvae of *Hydropsyche pellucidula* (Trichoptera, Hydropsychidae): a proposal of biological indicator. Bulletin of Environmental Contamination and Toxicology. 47: 261-265.

Carew, M.E., Pettigrove, V., Cox, R.L., and Hoffmann, A.A. 2007. The response of Chironomidae to sediment pollution and other environmental characteristics in urban wetlands. Freshwater Biology. 52: 2444-2462.

Clarke, K. R. 1993. Non-parametric multivariate analysis of changes in community structure. Australian Journal of Ecology. 18:117-143.

Connell, J. H. 1978. Diversity in tropical rain forests and coral reefs. Science. 199:1302–1310.

Doi, H. 2007. Stream macroinvertebrate community affected by point-source metal pollution. International Review of Hydrobiology. 92: 258-266.

Earth Tech. 2002. Ruddiman Creek sediment remedial investigation. Grand Rapids, Michigan.

Fisher, S.G., Gray, L.J., Grimm, N.B., and Busch, D.E. 1982. Temporal succession in a desert stream ecosystem following flash flooding. Ecological Monographs. 52: 93-110.

Gjerløv, C., Hildrew, A.G., and Jones, J.I. 2003. Mobility of stream invertebrates in relation to disturbance and refugia: A test of habitat templet theory. Journal of the North American Benthological Society. 22(2): 207-223.

Gilkinson, K.D., Gordon Jr., D.C., MacIsaac, K.G., McKeown, D.L., Kenchington, E.L.R., Bourbonnais, C., and Vass, W.P. 2005. Immediate impacts and recovery trajectories of macrofaunal communities following hydraulic clam dredging on Banquereau, eastern Canada. ICES Journal of Marine Science. 62(5): 925-947.

Gurtz, M.E., Marzolf, G.R., Killingbeck, K.T., Smit, D.L., and McArthur, J.V. 1988. Hydrologic and riparian influences on the import and storage of coarse particulate organic matter in a prairie stream. Canadian Journal of Fisheries and Aquatic Sciences. 45: 655-665.

Hilgeman,T.R. 2005. Environmental Protection Plan, Remediation of Ruddiman Creek Main Branch and Pond. U.S. Environmental Protection Agency, Great Lakes National Program Office, Chicago, Illinois.

Hilsenhoff, W.L. 1987. An improved biotic index of organic stream pollution. The Great Lakes Entomologist. 20(1): 31-39.

Hilsenhoff, W.L. 1988. Rapid field assessment of organic pollution with a family-level biotic index. Journal of the North American Benthological Society. 7(1): 65-68.

Janesak, J. 2006. Post Remediation Report of Ruddiman Creek Main Branch and Pond. U.S. Environmental Protection Agency, Great Lakes National Program Office, Chicago, Illinois.

Kelaher, B.P., Levinton, J.S., Oomen, J., Allen, B.J., and Wong, W.H. 2003. Changes in benthos following the clean-up of a severely metal-polluted cove in the Hudson River Estuary: Environmental restoration or ecological disturbance? Estuaries. 26(6): 1505-1516.

Lenat, D.R. 1993. A biotic index for the southeastern United States: Derivation and list of tolerance values, with criteria for assigning water-quality ratings. Journal of the North American Benthological Society. 12(3): 279-290.

Leopold, J.S., Wolman, M.G., and Miller, J.P. 1964. Fluvial processes in geomorphology. W.H. Freeman, San Francisco, California. 522 pp.

LeSage, S. and Smith, J. 2008. Water Quality and Pollution Control in Michigan 2008 Sections 303(d), 305(b), and 314 Integrated Report. MI/DEQ/WB-08/007. Michigan Department of Environmental Quality, Lansing, Michigan.

Linke, S., Bailey, R.C., and Schwindt, J. 1999. Temporal variability of stream bioassessments using benthic macroinvertebrates. Freshwater Biology. 42(3): 575-584.

Magurran, A.E. 1988. Ecological diversity and its measurement. Princeton University Press, Princeton, New Jersey. 179 pp.

Marsalek J., Diamond, M., Kok, S.K., and Watt, W.E. 2001. Urban runoff, chapter 11. In: Environment Canada, Threats to sources of drinking water and aquatic ecosystem health in Canada. National Water Research Institute, Burlington, Ontario. NWRI Scientific Assessment Report Series No. 1. pp. 47-50.

Mattingly, R.L., Cummins, K.W., and King, B.H. 1981. The influence of substrate organic content on the growth of a stream chironomid. Hydrobiologia. 77: 161-165.

Mattsson, B.J., and Cooper, R.J. 2006. Louisiana waterthrushes (*Seiurus motacilla*) and habitat assessments as cost-effective indicators of instream biotic integrity. Freshwater Biology. 51(10): 1941-1958.

Merritt, R. W., Cummins, K. W. (Editors), 1996. *An Introduction to the Aquatic Insects of North America.* Kendall/Hunt Publ. Co., Dubuque, Iowa.

McElravy, E.P., Lamberti, G.A., and Resh, V.H. 1989. Year-to-year variation in the aquatic macroinvertebrate fauna of a northern California stream. Journal of the North American Benthological Society. 8: 51-63.

Michigan Department of Environmental Quality (MDEQ). 1986. Administrative Rules, Part 4. Water Quality Standards. Lansing, Michigan.

Mockus, V. 1972. National Engineering Handbook, Section 4, Hydrology. U.S. Soil Conservation Service, Washington, D.C. 127 pp.

Mulvey, M., and Diamond, S.A. 1991. Genetic factors and tolerance acquisition in populations exposed to metals and metalloids. *In:* Ecotoxicology of metals: concepts and applications (M.C. Newman and A.W. McIntosh, Editors). Lewis Publishers, Inc., Chelsea, Michigan, USA. pp. 301-321.

Peeters, E. T. H. M., Dewitte, A., Koelmans, A. A., van der Velden, J., and den Besten, P. J. 2001. Evaluation of bioassays versus contaminant concentrations in explaining the macroinvertebrate community structure in the Rhine-Meuse Delta, The Netherlands. Environmental Toxicology and Chemistry. 20: 2883-2891.

Pielou, E.C. 1975. Ecologcial diversity. John Wiley & Sons, New York, New York. 165 pp.

Pitt, R. 2003. Receiving water impacts associated with urban wet weather flows. In: Handbook of ecotoxicology, Second Edition (Hoffman D.J., Rattner, B.A., Burton Jr., G.A., Cairns Jr., J., Editors). CRC/Lewis Publishers, Boca Raton, Florida. pp. 575-613.

Plafkin, J.L., Barbour, M.T., Porter, K.D., Gross, S.K., and Hughes, R.M. 1989. Rapid bioassessment protocols for use in streams and rivers: benthic macroinvertebrates and fish. EPA-444/4-89-001. Office of Water, U.S. Environmental Protection Agency, Washington, D.C.

Pollard, A.I., and Yuan, L.L. 2006. Community response patterns: evaluating benthic invertebrate composition in metal-polluted streams. Ecological Applications. 16: 645–655.

Quigley, M.P., and Hall, J.A. 1999. Recovery of macrobenthic communities after maintenance dredging in the Blyth estuary, north-east England. Aquatic Conservation: Marine and Freshwater Ecosystems. 9(1): 63-73.

Rediske, R.R. 2002. Preliminary investigation of the extent of sediment contamination in Muskegon Lake. Annis Water Resources Institute, Grand Valley State University, Muskegon, Michigan. 103 pp.

Rediske, R.R. 2004. Ruddiman Creek – Muskegon, MI, Technical Summary of Environmental Data and Issues. Annis Water Resources Institute, Grand Valley State University, Muskegon, Michigan. 40 pp.

Resh, V.H, Myers, M.J., and Hannaford, M.J. 1996. Macroinvertebrates as Biotic Indicators of Environmental Quality. Methods in Stream Ecology. (F.R. Hauer and G.A. Lamberti, Editors), Academic Press, Inc., San Diego, California. pp. 647-667.

Resh, V.H., and Unzicker, J.D. 1975. Water quality monitoring and aquatic organisms: The importance of species identification. Journal of the Water Pollution Control Federation. 47: 9-19.

Scullion, J., and Stinton, A. 1983. Effects of artificial freshets on substratum composition, benthic invertebrate fauna and invertebrate drift in two impounded rivers in mid-Wales. Hydrobiologia. 107: 261-269.

Scrimgeour, G.J., and Winterbourn, M.J. 1989. Effects of floods on epilithon and benthic macroinvertebrate populations in an unstable New Zealand river. Hydrobiologia. 171: 33-44.

Snell Environmental Group, Inc. 2000. Phase II File Review Report, Ruddiman Creek, Muskegon, Michigan, Prepared for United States Army Corp of Engineers, Detroit District. #9941-5494-29. Lansing, Michigan.

Sorrell, R.C. 2003. Computing flood discharges for small ungaged watersheds. Geological and Land Management Division, Michigan Department of Environmental Quality, Lansing, Michigan, 31 pp.

Stewart, P.M., Butcher, J.T., and Swinford, T.O. 2000. Land Use, habitat, and water quality effects on macroinvertebrate communities in three watersheds of a Lake Michigan associated marsh system. Aquatic Ecosystem Health and Management. 3: 179-189.

Stewart-Oaten, A., Murdoch, W.W., and Parker, K.R. 1986. Environmental impact assessment: pseudo-replication in time? Ecology. 67: 929-940.

Thorp, J.H., and Covich, A. P. (Editors) 2001. Ecology and Classification of North American Freshwater Invertebrates, Second Edition. Academic Press, San Diego, California. 1038 pp.

United States Environmental Protection Agency (USEPA). 1988. Ambient water quality criteria for chloride. EPA 440/5-88-001. Office of Water, Washington, DC.

United States Environmental Protection Agency (USEPA). 2003. National Drinking Water Standards. EPA 816-F-03-016. Office of Water, Washington, DC.

Uzarski, D.G., Burton, T.M., and Genet, J.A. 2004. Validation and performance of an invertebrate index of biotic integrity for Lakes Huron and Michigan fringing wetlands during a period of lake level decline. Aquatic Ecosystem Health & Management. 7(2): 269-288.

Vuori, K-M. 1994. Rapid behavioral and morphological responses of hydropysychid larvae (Trichoptera, Hydropsychidae) to sublethal cadmium exposure. Environmental Pollution. 84: 291-299.

Waters, T.F. 1995. Sediment in streams: sources, biological effects, and control. American Fisheries Society of Monograph. 7: 1-251.

Wiederholm, T. 1980. Use of benthos in lake monitoring. Journal of Water Pollution Control Federation. 52: 537-547.

Wuycheck, J. 1990. Biological and Sediment Contaminant Surveys of Ruddiman Creek and Unnamed Tributary, Muskegon County, Michigan. #MI/DNR/SWQ-90/101. Surface Water Quality Division, Michigan Department of Natural Resources, Lansing, Michigan.

Wuycheck, J., and Creal, W. 2002. Water Quality and Pollution Control in Michigan 2002 Sections 303(d) and 305(b) Integrated Report. Michigan Department of Environmental Quality, Lansing, Michigan.