EVALUATING ENVIRONMENTAL RESPONSE AND RECOVERY USING A MULTIPROXY PALEOLIMNOLOGICAL APPROACH

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ABSTRACT

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Human activity has influenced ecosystems since antiquity, with the health and function of aquatic ecosystems being significantly impacted since the industrial revolution. A better understanding of the response of modern aquatic ecosystems to natural and anthropogenic influences can be gleaned by reconstructing an environmental history of a system. An effective way to do this is to merge the record of historical perturbations with the overprint of modern stressors, which can be evaluated using lake sediment cores as they archive both historic and recent changes.

In a paleolimnological study of Muskegon Lake, Michigan, USA, multiple environmental proxies were analyzed to evaluate a system's response and recovery to human perturbation through time. The overarching hypothesis was that the disturbances from human activities (e.g., land use change) have been at an intensity that will not permit ecosystems to return to a pre-disturbance state; and because of other continuing and emerging stressors (e.g. climate change), ecosystems will not evolve to a new state of balance (characterized as steady state and/or equilibrium processes). To investigate this hypothesis, geochemical and biological proxies from a 150 cm sediment core were evaluated and compared to a chronology of human activity in the watershed. If true, the temporal patterns of biological and geochemical proxies will correspond to the chronology of natural and anthropogenic changes in the watershed.

The geochemical data from the core revealed suites of elements that corresponded to the source of the material, including terrestrial, productivity and anthropogenic related inputs. These elemental groups closely tracked the history of human activity and identified three phases of human influence. Phase one represented pre-disturbance (reference) conditions, phase two identified anthropogenic influence, and phase three suggested partial system recovery from phase two. Moreover, the profiles of anthropogenic elements were used to evaluate a geochemical reference condition for the system, and showed that the modern concentrations have not decreased to pre-historical values, thus indicating a scenario of an adapting reference condition state.

The biological reconstruction, inferred from fossil diatoms, also identified three distinct paleoecological phases corresponding to environmental change. The ecological preferences of individual diatom taxa suggest that biostratigraphic trends were driven by temperature, cultural nutrient inputs, and recently emerging stressors (e.g. climate and invasive species). Further, the benthic v. planktonic community structure of diatoms was used to reconstruct productivity regimes in the lake through time, which identified a significant shift from planktonic to benthic dominated productivity occurring in recovery phase of the core.

Finally, to better understand ecosystem function (e.g. time-lags and feedbacks), the proxies were integrated. Unique relationships among geochemical and biological proxies confirm the three phases representing ecosystem response to environmental change, supporting the overarching hypothesis. However, results also indicated apparent ecological change in response to up-and-coming stressors - of which little is known.

DEDICATION

This dissertation is dedicated to my children, parents, and sister; and the memory of my grandmother, Catherine Devenny Elwood.

ACKNOWLEDGEMENTS

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CHAPTER 1

INTRODUCTION TO EVLUATING ENVIRONMENTAL RESPONSE AND RECOVERY USING A MULTIPROXY PALEOLIMNOLOGICAL APPROACH

1.1 Introduction

The rate and magnitude of human influence has increased in recent centuries, which has resulted in greater impacts to ecological systems worldwide and added to concerns about society's reliance on natural resources as we attempt to achieve a sustainable society. Aquatic ecosystems provide particularly important natural capital for ecosystem services (e.g. potable water, fisheries, and recreation); yet they are particularly vulnerable to anthropogenic environmental change as atmospheric and terrestrial pollutants are ultimately funneled into rivers and lakes. Global increases in population and economic development, as well as land use transformations (e.g. deforestation, agriculture, and urbanization) impact the health of ecological systems, ultimately affecting the quality (and quantity) of water resources. As such, the societal relevance in understanding the full extent of anthropogenic impacts on aquatic ecosystems is of critical importance because the demand for ecological services is increasing. However, while understanding ecosystem dynamics is critical for developing sustainable societies, it also presents a challenge given the complexity of varying timescales and intensities of past anthropogenic stressors.

The broad goals of this study are to 1) interpret the response of aquatic ecosystems to human impacts and 2) untangle the complexity of ecosystem processes using archives of ecological indicators from lake sediments. This study utilizes a paleoecological approach, which is suitable because it provides a long-term chronological record of changing ecological conditions, that reveal functional dynamics and interrelationships (feedbacks) within systems and reconstructs the continued environmental response as the role of humans becomes more dominant. Identifying the signals of ecological response can help policy makers understand the past, present and future scenarios of freshwater ecosystems, thus making better decisions for sustainable management (Dearing et al. 2008).

Currently, there are gaps in knowledge about aquatic ecosystem response to human influence, and importantly, what that may indicate for water resources globally (Brock et al. 2008; Carpenter et al. 2003; Messerli et al. 2000). This disconnect exists, in large part, because while we have a detailed chronology of human activities available in historical records, we lack reliable long-term ecological history. This gap can be narrowed using a paleolimnological approach and addressing the following questions: 1) how have systems responded to multiple anthropogenic stressors?; 2) how does the current ecosystem state compare to the pre-perturbation state?; 3) is it possible to infer significant system shifts using indirect (e.g. fossil diatom) proxy methods?; 4) does using multiple proxies (e.g. chemical and biological) improve interpretations of environmental change?; and 5) can we predict future ecosystem recovery and/or a stable state scenario resulting from the advent of environmental regulations (e.g. Clean Air/Clean Water Acts)?

1.2 Literature review

Environmental system science

Given the complexity of environmental dynamics, it is necessary to carefully frame this study using a system that includes interactions between large, key 'system components', as well smaller scale internal processes embedded within those components, called 'constituents' (Kay 2008a; Kay 2008b). Figure 1.1 is a conceptual diagram showing key environmental system components and select constituents. The bi-directional arrows in the figure represent the potential interactions of modern systems using three major components: ecological, planetary and human forcings. The circular arrows within each box represent the relationships among select internal constituents of that component. The targeted system for this study, an aquatic ecosystem, is an open system, and therefore both a source and a sink for the flow of matter and energy. For example, the web of ecological processes in aquatic ecosystems is influenced by physical, chemical and biological processes operating internally through feedback mechanisms (e.g. nutrient cycling with primary producers); as well as the external relationships with other components (e.g. solar radiation influence on primary producers).

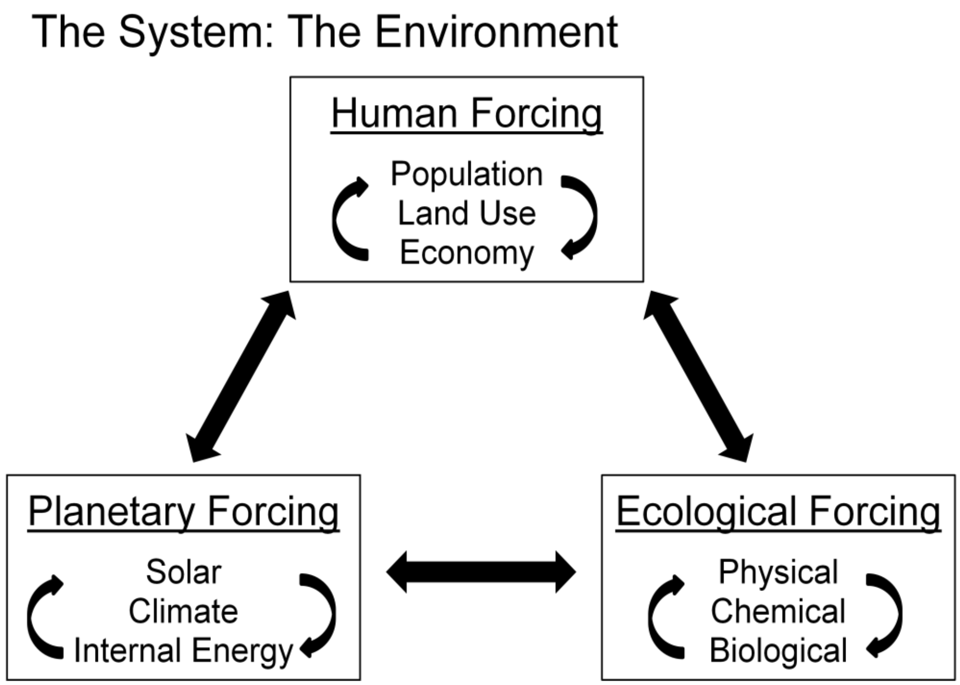


Figure 1.1: Conceptual map of potential environmental system interactions (modified from Dearing 2006)

Dynamics in environmental systems are not naturally temporally static. Most ecological systems slowly change over time (e.g. natural succession - ontology), which slowly changes the type and function of the ecosystem (Battarbee et al. 2005). Pre-historically, before major human perturbation, a two component system of ecological and planetary forces operated in a state of interconnected and successional balance. In this type of environment, the system evolves to new states, but the time scale typically allows for components and constituents to slowly adapt. Modern ecosystems, however, encounter an additional forcing from human perturbations (forcing), resulting in a three component system where the human component often dominates (Dearing et al. 2006a; Dearing et al. 2006b). Consequently, global ecosystems have changed where humans have an impact on earth systems; at times causing abrupt, catastrophic ecological change, and shifting a predictable balance between components, resulting in an unpredictable and unstable state (Scheffer and Carpenter 2003; Carpenter 2008)

A vital aspect to ensuring that ecosystem health, and thus, ecosystem services are maintained requires employing sustainable management practices. Determining what is a sustainable, or even a feasible management strategy necessitates an understanding of how complex ecological systems respond to the long term effects of anthropogenic stressors (e.g. land use changes and climate change) (Dearing et al. 2008). Figure 1.2 shows a conceptual model of environmental response to anthropogenic influence at varying temporal scales. This figure is delineated by development status; and by varying degree of management strategies (modified from Garcier 2007). Because of the inherent uncertainty, and at times, lack of documentation regarding human influence in both the Europe/Asia region and in modern developing nations, sites (systems) in North America present a good opportunity to study environmental change. Systems in this region possess a near complete recorded timeline of human manipulation of the environment. In the Great Lakes Region, this began in earnest in the early 1800s following the arrival of Europeans to the North American continent (Alexander 2006).

Correlating human events with ecological conditions since that time are used to evaluate how humans have impacted the environment (Dixit et al. 1992; 1999; Reavie et al. 1998). Reconstructing environmental response is further useful given that future ecological conditions depend on management decisions; therefore, understanding environmental response is directly linked to the feasibility of making informed recommendations that best restore and maintain sustainable systems.

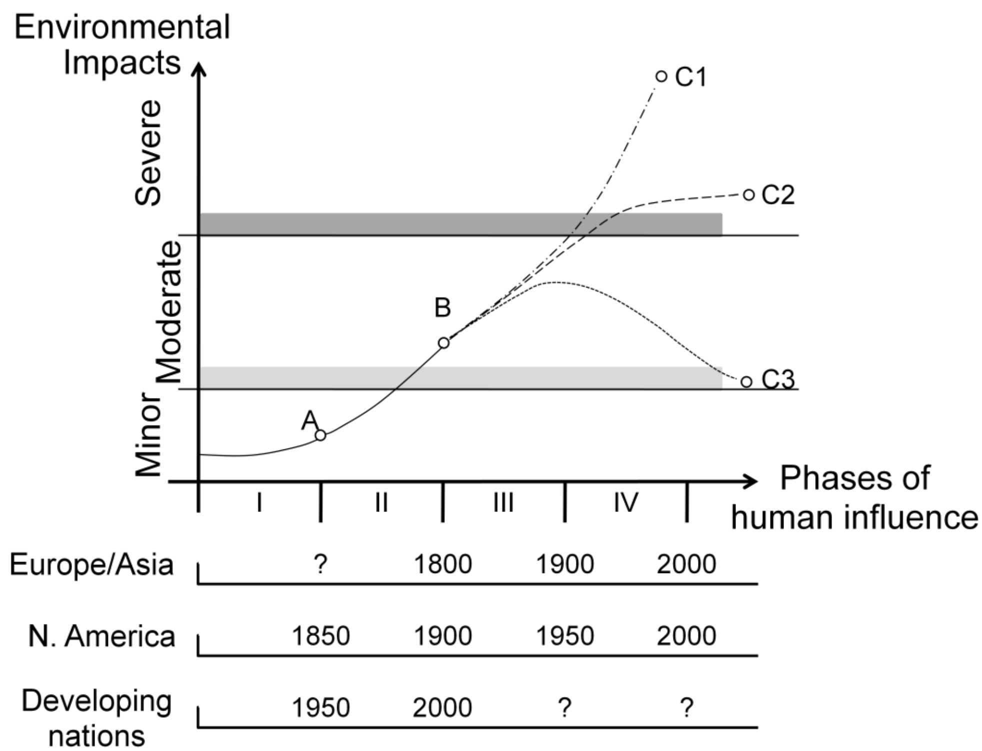


Figure 1.2: Conceptual model of environmental response to anthropogenic influence in varying regions delineated by development status; and under contrasting management strategies through time. The line to point A shows a linear relationship with population/development and environmental impact. Point A to B depicts an exponential increase of environmental impacts related to industrialization. The line from B to C1 results when no management strategies are in place to mitigate environmental degradation from human activity. The line from B to C2 illustrates the trajectory of environmental impacts with minimal management oversight. The line from B to C3 demonstrates partial environmental recovery with strict environmental regulations. (Modified from Garcier 2007)

Paleoecology

Paleolimnological archives offer a unique opportunity for understanding the broad scope of ecological change, as paleoecological reconstructions reveal the complexity of the ecological-planetary-human system (Dearing et al. 2006a; Dearing et al. 2008). This approach is useful for evaluating environmental response because assessing pre-historic ecological health from contemporary environmental measurements (e.g. water column total phosphorous) is not possible without using inferences. Furthermore, ongoing monitoring efforts since the 1970s have not sufficiently captured the dynamics of long-term environmental behavior (Bennion and Battarbee 2007).

Paleoecological studies main focus was historically on a single indicator or a limited number of proxies from lake sediments to infer climate change, to evaluate regional and watershed scale perturbation in land use and land cover change, and to reconstruct historic environmental events such as fires, logging (e.g., Cooper and Brush 1993; Cooper 1995; Stoermer et al. 1996; Bunting et al. 1997; Bradbury et al. 1997; Smol and Cummings 2000; Reavie et al. 2000; Huang and O'Connell 2000; Garrison and Wakeman 2000; Paterson et al. 2001; Yohn et al. 2002; Sorvari et al. 2002; Das 2002; Brugam et al. 2003; Cohen et al. 2005). Commonly used paleoecological proxies include: multi-element sediment geochemistry (Olsson et al. 1997; Chapman et al. 1998; Yohn et al. 2002; Siver et al. 2003; Parsons et al. 2004; O'Reilly et al. 2005); diatom population structure (Wolfe et al. 1996; Hall et al. 1997; Hall and Smol 1999; Slate and Stevenson 2000; Battarbee et al. 2001; Ramstack et al. 2003); pollen stratigraphy (Yansa and Ashworth 2005; Birks and Birks 2002; Cohen et al. 2005b; Davidson et al. 2005); ostracodes (Forester 1991, 1987, 1983; Smith 1993); chrysophycean statospores, silicoflagellates and algal pigments (Smol 2002). While these can be effective for specific research objectives (e.g., nutrient loading), a single inference proxy is not robust enough to determine the comprehensive influence of environmental change on stressor-responses within aquatic ecosystems (Dearing et al. 2008). This investigation uses multiple proxies archived in lake sediment to gain an understanding of historic reference conditions and ecological regime shifts. Deciphering the impact of multiple environmental stressors that operate at different timescales and intensities will lend insight to overall stressor-response relationships.

The key to understanding ecological behavior under the modern three component system lies in the slice of history since the human component has become dominant, in North America this is roughly the past 300 years. Lake sediment collects material deposited in lakes during this slice of time, as atmospheric and terrestrial pollutants ultimately funnel into lake basins (Smol 2002; Wetzel 2003). This process naturally records environmental change, and the subsequent internal and external feedback mechanisms driving that change (Smol 2002). Consequently, the time period focused on in this study will document the historic reference condition (before human impacts) and the human induced ecological regime shifts. These tell the broader story of ecosystem processes, and facilitate a mechanistic understanding of environmental dynamics.

1.3 Research hypothesis

Modern systems have deviated from the pre-perturbation two-system dynamic, where ecological balance slowly adapted in concert through natural succession (ontology)

(Dearing et al. 2006(a)). What remains unclear is: the degree to which modern systems have been altered, which specific human impacts (e.g. logging, industry and urbanization) have resulted in the most catastrophic changes, the environmental legacy of past watershed activity on modern systems is, and the future scenarios expected with the overprint of multiple contemporary stressors. To understand these issues, it is necessary to reconstruct the ecological history to understand mechanisms of system transformations. Therefore, a temporal perspective of ecological behavior is advantageous to understanding how systems respond (Smol 2002). Aspects of this include investigating the time-lags, the strength of system resiliency, the thresholds for irreversible change, and the "stability" of ecological regimes (Scheffer and Carpenter 2003).

Aquatic ecosystems are good ecosystems to investigate environmental change, because they are particularly sensitive to the effects of human forcing as the ecological endpoint of material flushed from the atmosphere and terrestrial landscape (Smol 2002; Andersen et al 2004; Bennion and Battarbee 2007). Consequently, aquatic ecosystem 'response' to anthropogenic inputs, as recorded in sediment, offers a reliable indication of a system's ecological history, in terms of how it changes as a consequence of human perturbations (Dearing et al. 2008).

The objective of this research is to examine environmental change through the lens of an "environmental systems approach" that will best interpret ecological processes. Specific objectives of this study include; 1) to determine how aquatic ecosystems respond to specific perturbations (e.g. logging, industry, urbanization) using individual paleoecological proxies (e.g. geochemistry and diatoms); 2) to interpret the environmental legacy of anthropogenic stressors by calculating deviation from geochemical reference concentration and by identifying ecological regime shifts; 3) to identify correlations/relationships among geochemical and diatom sediment chronologies that infer process such as lag-time and feedback mechanisms. The overarching hypothesis driving this research was that: ecosystems highly disturbed by human activity cannot be expected to return to a pre-disturbance state; and further, with the influence of climate change, and other continuing and emerging stressors, ecosystems will not obtain a new state of balance (steady state and/or equilibrium).

To evaluate this hypothesis, data from multi-element sediment geochemistry and fossil diatoms were examined independently, and then integrated to better understand the temporal dynamics of aquatic ecosystem function as a whole system. The main tools developed in this study will identify/validate relationships among proxy groups of the core geochemistry, and examine planktonic versus benthic diatom community structure to better understand productivity regime shifts. The integration chapter will contribute to how the indicators can be merged to enhance single proxy interpretations. This study is important for contributing to an understanding of long term environmental response trends related to the history of anthropogenic activity and changing climate influences in the Great Lakes region. This framework is also transferable to other regions as it will address the system response to management strategies aimed at protecting water quality in watersheds.

1.4 Study Area

Great Lakes Watershed

The Laurentian Great Lakes region has experienced exponential growth in human development due to increased trade, logging and industry since the advent of Euro-American settlement. These developments emerged from the use of the region's natural resources, including vast forests of white pine and hardwood species, as well as metal deposits, including copper, iron, gold and silver (Alexander 2006; Wells 1978). The utility of abundant tributaries connected to the massive Great Lakes corridor allowed raw materials and finished goods to be transported to ports throughout the region in the 19th and 20th centuries, and ultimately distributed around the world (Alexander 2006). Historically, these activities and related modifications to hydrology for agriculture and municipal uses (e.g., dams, diversions and reservoirs) have contributed to the degradation of aquatic ecosystems in the Great Lakes watershed (Wolin and Stoermer 2005).

Ecological change in the Great Lakes region has been coupled with the overprint of a warmer and wetter climate regime since the Little Ice Age ended (~1850), further stressing ecosystems with changing environmental conditions (Magnuson et al. 1997). A study by Magnuson et al. (2000), data indicate that ice phenologies (records of ice freeze and thaw dates) in the Great Lakes region since the Little Ice Age have averaged ~6 days later and ~6 days earlier per 100 years, which corresponds to a 1.2°C increase in the last 100 years. Ecological response to warming conditions can further complicate impacts on aquatic ecosystems; with various feedback mechanisms little understood and highly uncertain (Magnuson et al. 1997).

Muskegon River Watershed

The Muskegon River Watershed is part of the Great Lakes Watershed, located in the Lower Peninsula of western Michigan. It is a dominant drainage network in the state, fed by wetlands, groundwater springs, lakes, agricultural drains and tributaries. Figure 1.3 shows the Muskegon River Watershed which encompasses 7057 km^2 of western and central Michigan (Wahrer 1996; U'Ren 2002; Pijanowski et al. 2008). The Muskegon River flows more than 350 km from its headwaters at Houghton and Higgins Lake to its mouth at Muskegon Lake, then eventually empties into Lake Michigan through a navigation channel (De Mol ANNIS 2005; Pijanowski et al. 2008; Steinman et al. 2008). Average rain fall is increasing in the watershed, with a recent average of 90 cm per year; while from 1899 through 2001, the average was 83 cm. A lower average precipitation of 75 cm occurred from 1920-1940 (Pijanowski et al. 2008). Surficial glacial deposits dominate the Muskegon River Watershed. These deposits overlay Jurassic "red-beds" in the upper reaches, to Pennsylvanian Period Grand River – Saginaw Formations in the central portion, and Mississippian Period Parma Sandstone, Bayport Limestone, Michigan Formation and Marshall Sandstone in the lower segment of the watershed (Wahrer et al., 1996; Westjohn and Weaver, 1997; U'Ren and Conzelmann 2002; Fitzpatrick et al. 2007).

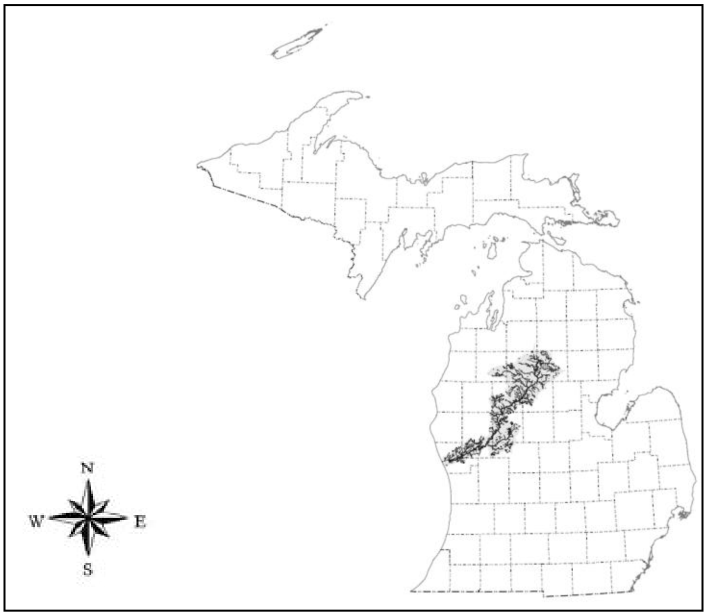


Figure 1.3: Location of the Muskegon River Watershed in Michigan, USA.

Land use varies throughout the watershed, but is dominated by forests and agricultural lands, with small scattered urban areas (Varnakovida et al. 2005). Figure 1.4 shows modern land use in the Muskegon River Watershed. In the upper watershed, nearly twenty percent of the watershed is state forest land in public ownership, with the remaining land also being predominately forested (Pijanowski et al. 2008). The middle watershed is dominated by agriculture, with increasing urbanization and reforestation as agriculture decreases in significance. The lower watershed is the most developed, and has the largest urban and industrial areas. The city of Muskegon, where the Muskegon River discharges into Muskegon Lake, is the largest urban area in the watershed with an estimated population of 40,000 people in 2000 (Pijanowski et al. 2008). Tourism and recreation on Muskegon Lake and throughout the watershed are vital ecosystem services that rely heavily on the lake's fisheries. However, it is forecasted that there will be an increase in urbanization in the watershed which may contribute to further watershed degradation and therefore decrease the economic vitality of the region, since it relies on revenue from the fisheries for a significant portion of its economy (Tang et al. 2005; Pijanowski et al. 2005).

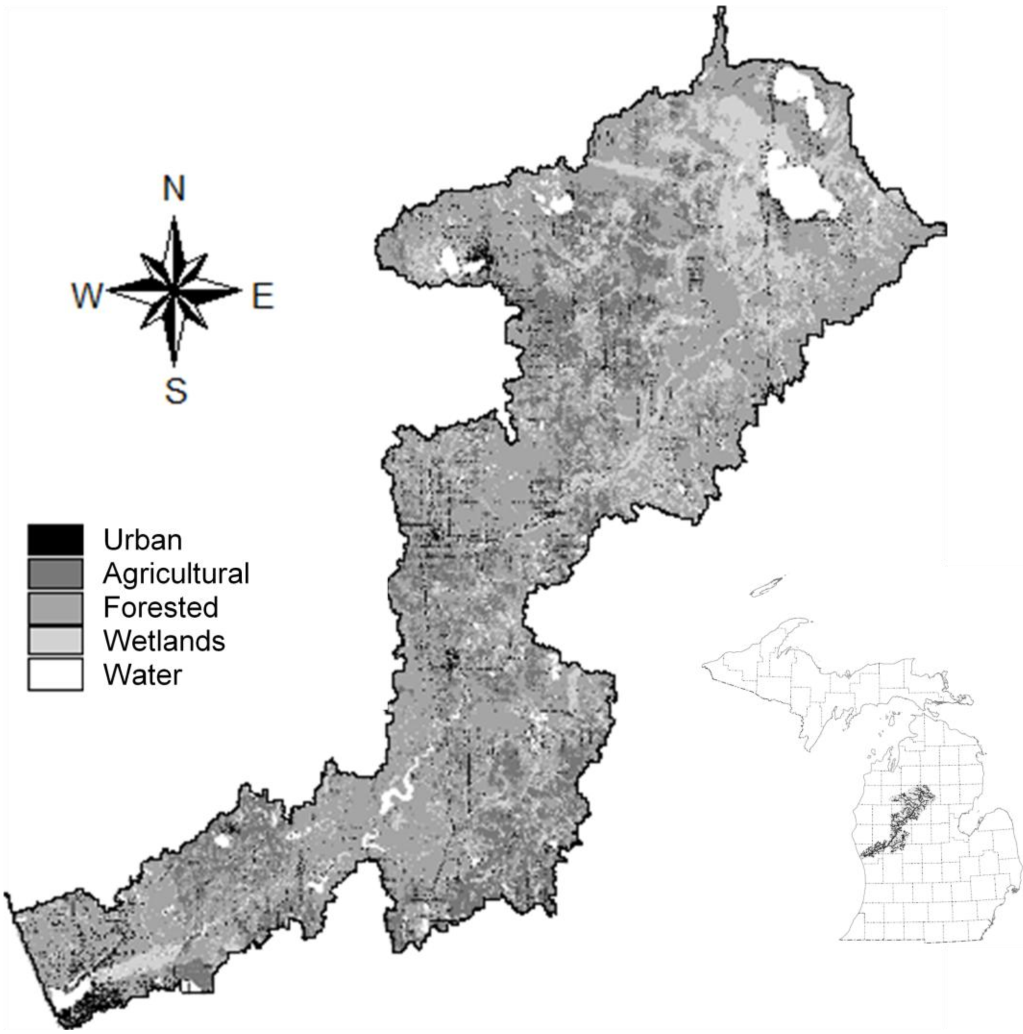


Figure 1.4: Current land use in the Muskegon River Watershed, Michigan, USA.

Historic Muskegon River Watershed Disturbances

Until the seventeenth century, the Muskegon River Watershed was not significantly altered by humans, despite human occupation for several thousands of years. Paleo-Indian hunters were the first inhabitants following the retreat of the late

Wisconsinsan glaciers (Alexander 2006). This population was succeeded by Archaic Woodland and other cultural traditions. Before Euro-American settlements appeared, various bands of Ottawa and Pottawatomi Indians were the predominant tribes in the area. The name "Muskegon" is a derivation of the Ottawa Indian word "Masquigon" meaning "river with marshes". This is supported by the fact that the Masquigon River is how the Muskegon River is identified on French maps dating from the late seventeenth century; also indicating that French fur trappers probably reached the western coast of Michigan by that time (Alexander 2006). Fur trapping in the region was followed by exploitation of the vast pine forests present in Michigan (Whitney 1987; Alexander 2006). Besides logging in the Muskegon River watershed, rivers and streams were used to transport logs to newly developing cities. At one point during the lumber peak of the 1880s, the city of Muskegon had more than 47 sawmills (Alexander 2006). By 1890, the watershed's dense White Pine forest was almost completely harvested, and logging came to a halt by 1910 (Alexander 2006; Wells 1978). Declining lumbering prior to 1900 began to be replaced by heavy industry.

Following the depletion of lumber resources, emerging 20th century developments included a growing industrial base (The Association for Iron and Steel, 1904). Among the first major factories to locate to Muskegon Lake's shoreline was the Central Paper Company, currently SAPPI Fine Paper - North America (Alexander, 1999). The paper mill, foundries, oil tank farms, and other factories that located to the City of Muskegon, City of Roosevelt Park, City of Norton Shores, and City of Muskegon Heights, contributed to heavy metals and toxic chemicals inputs to Muskegon Lake. Some of this development was undermined by the Great Depression in the 1930s;

however, the economy rebounded during the World War II era, with Muskegon having a role in arsenal production, with oil and chemical industries having a prominent presence (Alexander 2006). In the mid-1900s, foundries, metal finishing plants, a paper mill, and petrochemical storage facilities were on the shore of Muskegon Lake (Steinman et al. 2008; Alexander 2006). Subsequently, industrial pollution concerns dominated Muskegon environmental issues and effected human health for decades (Carter et al. 2006; Rediske et al. 2002). Expansion of heavy industry and shipping in 1960s and 1970s contributed to over 100,000m^3/day of wastewater discharged from industrial and municipal sources (Great Lakes Commission 2000, Evans 1992). The initial logging and later effects of the automobile (leaded gasoline), coupled with other industrial activities, have permanently altered the natural environmental landscape and degraded habitat quality in the lake and its tributaries. The legacy of those impacts resulted in two superfund sites in the Muskegon Lake watershed area, and multiple advisories on fish and wildlife consumption due to mercury and PCB contamination (MDEQ 2006). Additionally, invasive species such as zebra mussel and blooms of the exotic marine microalgae, *Enteromorpha flexuosa*, have plagued Muskegon Lake, thereby having degraded water quality, aesthetics, and fishery habitats (Lougheed et al. 2004). See Figure 1.5 for a timeline of relevant events in the history of the Muskegon River Watershed.

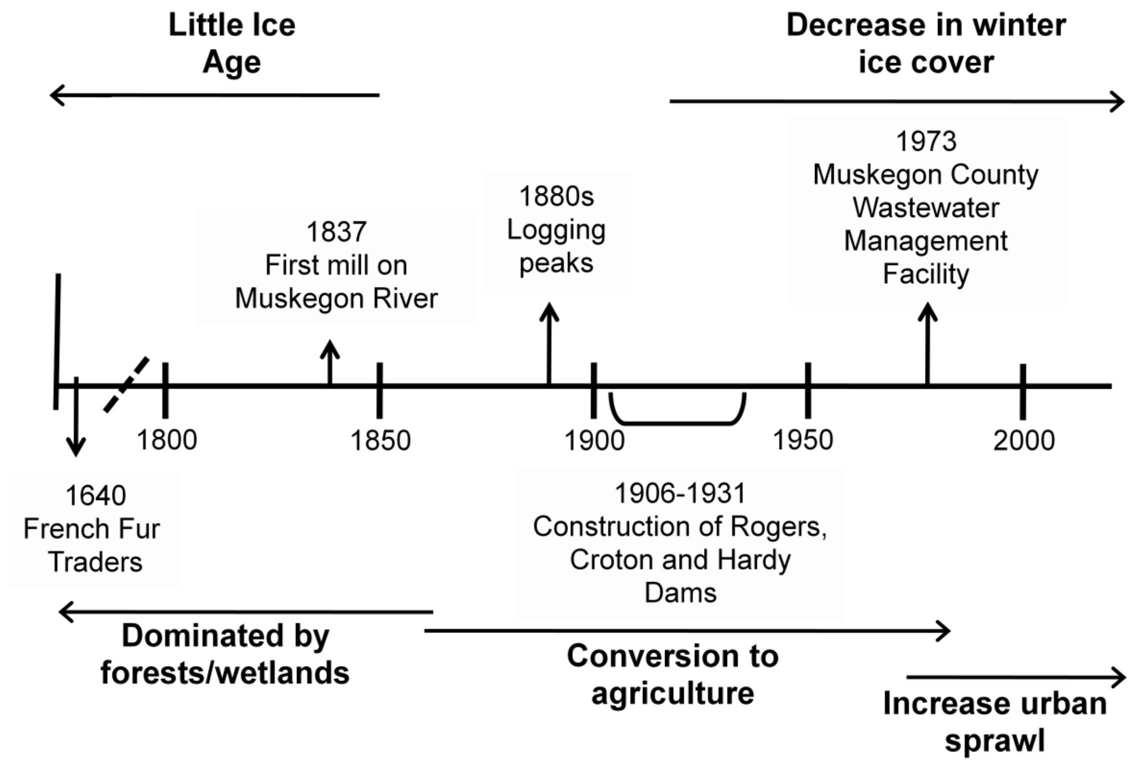


Figure 1.5: Timeline of relevant historical events in the Muskegon River Watershed.

As part of remediation efforts in the early 1970s, one directive set to minimize human effects on the Muskegon River Watershed, including Muskegon Lake, was the installation of a waste water treatment plant (Steinman et al. 2008; Carter et al. 2006). An 11,000 acre tertiary facility was constructed in 1973 to divert industrial and municipal wastewater from surface and groundwater discharges. This land application system is capable of treating 42 million gallons a day using aeration, lagoon impoundment, slow-rate irrigation and rapid-sand filtration (http://co.muskegon.mi.us/wwtf.htm). The Muskegon River receives the treated wastewater at a point nearly 26 miles upstream of

Muskegon Lake, near Mosquito Creek. As a result, there have been substantial improvements in the water quality of Muskegon Lake.

Muskegon Lake

Muskegon Lake is a 16.91 km^2 inland water body located in the Laurentian Great Lakes region (W 86° 17′ 25.42″, N 43°13′59.45″). Mean depth in the lake is 7 m with a maximum depth of 23 m. Muskegon Lake itself is the end point of drowned river mouth system that connects the Muskegon River Watershed to the coastal zone of Lake Michigan through a navigation channel (Steinman et al. 2008). Residence time is ~23 days, with the primary inflow being the Muskegon River, and the primary outlet flowing to Lake Michigan (Freedman et al. 1979). The lakes proximity to the Great Lakes and general ecological setting make it an important fishery, though invasive species, habitat loss and degradation continue to be factors of concern (Clapp et al. 2001; LaMP 2004).

Muskegon Lake has a history of intense anthropogenic activity on its shoreline since early 1800s. At one point during the lumber peak in the 1880s, the city of Muskegon had more than 47 sawmills (Alexander 2006). Following the depletion of lumber resources, turn of the century development in Muskegon involved industrial corporations; but this was quickly undermined by the Great Depression in the 1930s. The economy rebounded during the World War II era, with Muskegon having a role in arsenal production, with oil and chemical industries having a prominent presence (Alexander 2006). In the mid-1900s, foundries, metal finishing plants, a paper mill and petrochemical storage facilities were on the shore of Muskegon Lake (Steinman et al. 2008). Subsequently, industrial pollution concerns dominated Muskegon environmental issues and effected human and ecological health concerns for decades (Rediske et al. 2002; Carter et al. 2006; Steinman et al. 2008). Expansion of heavy industry and shipping in 1960s and 1970s contributed to over 100,000m^3/day of wastewater discharged from industrial and municipal sources until a tertiary waste water treatment plant was installed in 1973 (Freedman et al. 1979; Evans 1992; Great Lakes Commission 2000, Steinman et al. 2008).

Historical land use around Muskegon Lake is available for a 500 m buffer from 1900 to present. Figure 1.6 shows Muskegon Lake with a defined 500 m buffer, and the dominant modern land uses from MiDGL. Figure 1.7 shows the historical percentage of water, wetlands, grasslands, agriculture, forests and urban land use (Ray and Pijanowski 2010).

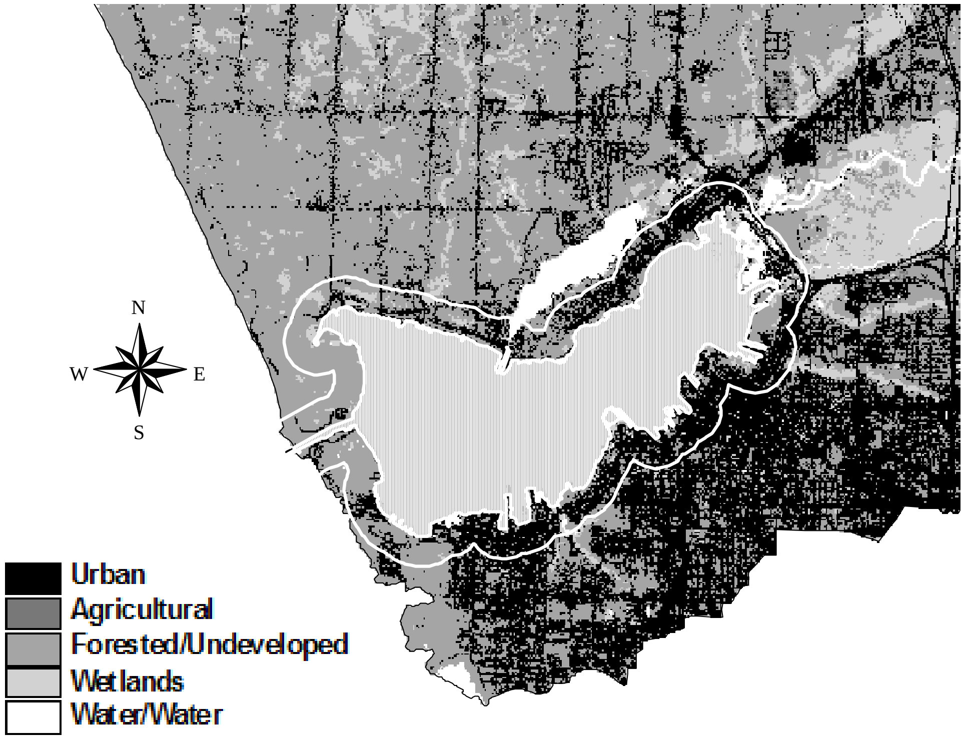
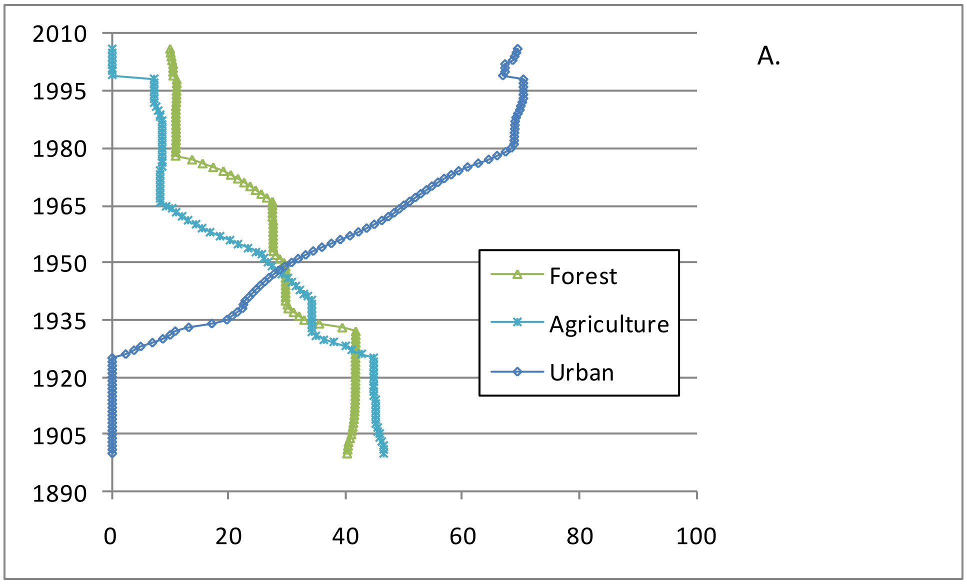


Figure 1.6: Modern land use near Muskegon Lake, with a 500 m buffer delineated.



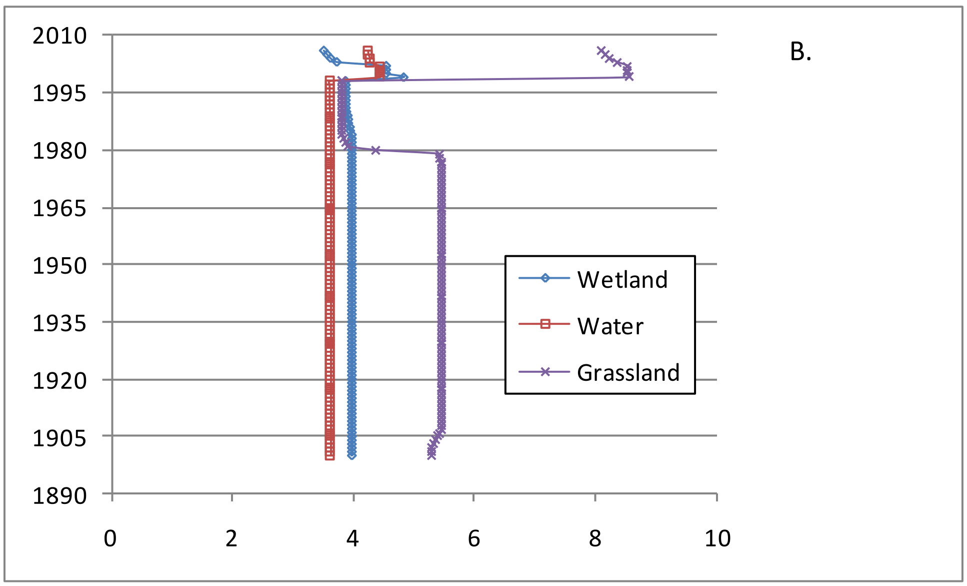


Figure 1.7a and b: Graphs showing historic land uses of a 500m buffer of Muskegon Lake (Ray and Pijanowski 2010)

Shoreline development has furthered the declining ecological status of Muskegon Lake by altering natural buffers and habitats, and by introducing metal, organic and nutrient pollutants. Many of these pollutants impact biota in the water column before being buried in the sediment of the lake (Carter et al. 2006; Redieski et al. 2002). A recent inventory of 49.99 km of Muskegon Lake shoreline shows that 64.93% (32.46 km) is hardened and 35.07% (16.99 km) is natural. Of the hardened shoreline, 10.31 km is metal seawall, 9.98 km concrete riprap, and 3.85 km rock riprap. The natural shoreline is predominately located on the lake's north side; while remaining industrial activity is concentrated on the south shore (Steinman et al. 2008). Figure 1.8 shows the change in the shoreline of Muskegon Lake (Steinman, personal communication, 2008).

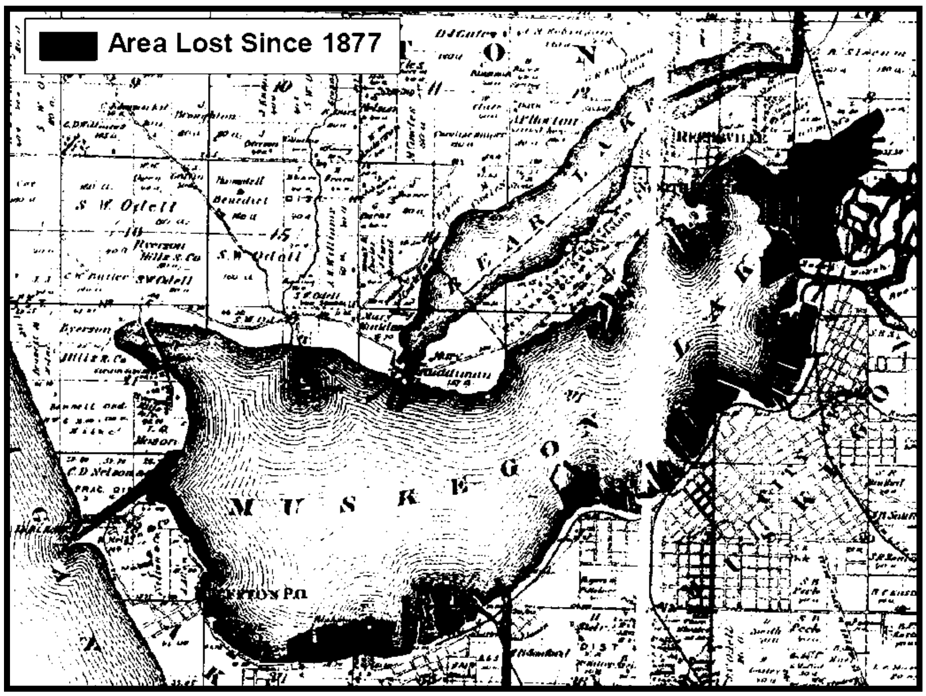


Figure 1.8: Map of Muskegon Lake showing loss in shoreline since 1877. (Steinman, personal communication, 2008)

As mentioned, substantial water quality improvements in Muskegon Lake resulted from the installation of the Wastewater Management System (WMS) in 1973. Total phosphorus and soluble reactive phosphorus decreased from 68 to 27 µg/L, and from 20 to 5 µg/L, respectively, from installation to 2006 (Steinman et al. 2008). Secchi disk depth also increased from 1.5 to 2.2 m. Conversely, nitrate concentrations have increased from 70 to 270 μg/L over the same time period (Steinman et al. 2008; Freedman et al. 1979).

Overall, the Muskegon Lake ecosystem improved as a result of this particular environmental directive. However, non-point source stressors and a history of contaminated sediments have plagued the lake and may be a factor in the observed modern increases in water column nitrate concentrations (Steinman et al. 2008; Carter et al. 2006; Rediske et al. 2002).

Though there is great awareness through stakeholder involvement, and policy initiatives in place to restore ecological integrity to Muskegon Lake (e.g. Great Lakes Fisheries Trust funding). However, there remains a need to understand how the system will respond to these efforts, particularly with the overprint of modern stressors such as land use change and climate change. Understanding historic ecological response in Muskegon Lake will clarify future scenarios of ecological health, not only for the Muskegon River Watershed, but for the larger Great Lakes Watershed and similar environments globally.

1.5 Methods

Sampling

The Michigan Department of Environmental Quality M/V *Nibi* and a modified Mooring Systems Model 2172 Piston Corer were used to collect a 149 cm core from the depositional basin of Muskegon Lake. The core was collected as close as possible to the latitude and longitude of the 2003 coring location reported in Parsons et al. (2004)(43º14.042'N, 86º17.006'W) as determined using a bathymetric map, and GPS electronic depth finder (to facilitate core chronology). On shore immediately following sampling, the core was sectioned at 1 cm resolution using a hydraulic extruder. Sediment were placed in acid washed polypropylene jars and quickly put on ice to maintain the chemical and biological integrity of the sample. Subsamples were double homogenized, split for sediment geochemical and diatom analyses, placed in whirl-pak® bags and frozen.

Sediment description

Descriptions of sediment color and texture were recorded for each increment of the 149 cm core at the time of sampling. Throughout, lithographic units were not easily discernable, with mostly dark, thick mud. Visual inspection showed that: sediment were mostly organic and watery from 0 to 9 cm; consisted of brown mud from 10 to 20 cm; became moderately thicker with fragments of shells and petroleum odor intermittent from 21 to 59 cm; darker and thicker mud was present with continued shells and other organic fragments from 60 to 98 cm; more dark brown and thick sediment from 99 to 149 cm, with lenses of wood fragments present from 95 to 115 cm (presumably from logging). An additional, more detailed visual inspection of sediment performed in search for ^{14}C dating material revealed that wood fragments and saw dust was prominent between 94 and 115 cm depth. This is believed to indicate the time period of 1870 through 1910 when logging was the dominant human activity in the watershed. Descriptions of the sediment core characteristics are given in Appendix I.

Porosity

Porosity was determined by placing a separate subsample in a pre-weighed whirlpak® bag and then reweighing after freeze-drying. Results of porosity from the Muskegon Lake core are given in Appendix II. Porosity was determined from a separate sub-sample using the calculation:

$$\varphi = (\text{Mass of wet} - \text{Mass of dry}) / \text{Mass total}$$

1.6 Research Approach

This chapter has discussed the overall concept and objectives for this research. While other research has explored multiple paleoecological proxies, the perspective of this study that links theory to observation using a highly disturbed system is unique. Particularly, integrating the use of high resolution geochemistry and biological indicators, coupled with the detailed record of human activity in the Muskegon River Watershed will further the understanding of environmental response to successive waves of human influence. Furthermore, the nature of perturbations in the Great Lakes region is unique in that there has been significant environmental change in a relatively short span of time, allowing paleolimnology to capture insight to the pre-disturbance 'state of the system'. The analyses presented in each of the chapters demonstrate an approach for interpretation of environmental change using ecological indicators.

Chapter 2 focuses on how contemporary geochemical concentrations can be assessed by investigating their deviation from 'reference' concentrations (with consideration of underlying natural variability). A suite of elements was employed to best define elemental classes and relationships among classes through time in response to the multiple anthropogenic stressors of Muskegon Lake.

Chapter 3 investigates the application of regime shifts to ecological systems using diatom community structure. Shifts in benthic versus planktonic taxa from the lake sediment will be the focus of this chapter, as they have been shown to effectively record productivity and corresponding nutrient conditions as inferred through community stratigraphy.

Chapter 4 integrates chemical and biological proxies to determine interrelationships and feedback mechanisms. This final chapter seeks to understand irreversible and non-linear changes in complex environmental systems following human perturbations that resulted in abrupt and unpredictable catastrophic environmental change.

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CHAPTER 2

EVALUATING REFERENCE CONDITIONS USING GEOCHEMICAL CHRONOLOGIES FROM A LACUSTRINE SEDIMENT CORE

Abstract

In recent centuries, human impacts such deforestation, chemical pollution and invasive species have stressed freshwater ecosystems in the Great Lakes Region, altered the rate and function of biogeochemical processes, and resulted in surface water impairment. Efforts to mitigate environmental degradation increasingly refer to an ecological reference condition as a target for remediation goals. However, questions and uncertainty remain as to whether this is an appropriate or feasible target since it is difficult to determine a system's prehistorical reference condition using contemporary monitoring techniques.

This study used a lacustrine sediment core from a highly perturbed watershed to investigate the magnitude and intensity of environmental change, with an emphasis on identifying the geochemical reference condition; or the system's geochemistry prior to Euro-American logging. The hypothesis for this study is that, 1) disturbance of ecosystems from prehistoric reference conditions can be identified by changes in temporal trends of individual elementals as well as the relationships among proxy groups, and 2) systems have not returned to prehistoric reference conditions but rather are evolving to new states of balance due to the legacy of human perturbation. To test this

hypothesis, sediment core geochemical profiles were evaluated to determine how the system responded to secular and human perturbations.

Results identified three temporal phases of geochemical response; 1) pre-perturbation, or “reference conditions” (core bottom), 2) human dominated phase with increased anthropogenic element concentrations (core middle), and 3) policy response phase with decreased anthropogenic element concentrations (top 25 cm of core). Modern elemental concentrations at the top of the core did not match the ‘reference values’ from the bottom of the core. The environmental ‘recovery’ phase, following the installation of a tertiary waste water treatment plant upstream from Muskegon Lake, did reflect decreasing anthropogenic loading of metals and nutrients; however, effects of terrestrial run-off and increased urbanization continue to be observed in the certain sediment geochemical profiles. While the hypothesis was supported by the results, as both individual elements and proxy groups tracked temporal geochemical changes and identified the prehistorical reference condition, this investigation suggests greater consideration needs to be directed at the environmental legacy of human behavior when interpreting modern sediment geochemical trends. Additionally, conclusions from this study suggest a reassessment of reasonable ecological recovery expectations may be necessary, given the influence of contemporary human disturbances and the superimposed predicted changes in climate.

2.1 Introduction

Problem

In recent centuries, human impacts have been a major force stressing global freshwater ecosystems (Smol 2002). In the Great Lakes region, major human perturbation began with rapid and catastrophic deforestation, as Euro-American settlers leveled forests to clear land for agriculture; and later, industrialization and urban development (Tang et al. 2005; Anderson 2004; Pijanowski et al. 2007). Since the 1800s, these activities have resulted in altered hydrology, disrupted biological communities and chemical contamination of aquatic ecosystems (Freedman et al. 1979; Hall and Smol 1995; Anderson 2004; Steinman et al. 2008). Environmental regulations (e.g. Clean Water/Clean Air Acts) initiated in the early 1970s were designed to mitigate the effects of catastrophic ecological degradation, and to maintain critical ecosystem services such as potable water, fisheries, and tourism. Efforts to restore ecological health of degraded systems often recommend a prehistorical "reference condition" (also referred to as background, pre-perturbation or pre-settlement) as a remediation target (Andersen et al. 2004; Battarbee et al. 2005; Bennion and Battarbee 2007). A challenge with this recommendation is that quantifying a genuine reference condition, or state of the environmental system prior to disturbance, is not possible using modern empirical data. Additionally, it has been suggested that the reference condition adapts through time as a function of secular changes and/or the legacy of anthropogenic impacts (Battarbee 1999; Battarbee et al. 2005; Long et al. 2010). This chapter presents lake sediment geochemical data to explore aspects of historic system response to environmental change, evaluate the reference condition concept, and determine modern trends to lend insight about what can be expected from system recovery.

Reference conditions

Reference conditions are defined as a baseline, or background measure of an ecosystem variable (including biological, chemical, or physical attributes) representative of a system having no (or minimal) human influence (Karr 1981, Bennion et al. 2004; Bennion and Battarbee 2007). In the absence of long-term monitoring data that identify an ecological reference condition, paleolimnology is a tool that provides the temporal framework; since sediments in aquatic ecosystems are excellent recorders of environmental change and thus offer a solution for establishing reference conditions (Leira et al. 2006, Ekdahl et al. 2007). Using paleolimnology, pre-perturbation conditions can be reconstructed using a single inference proxy, or combination of proxies (Anderson 1993; Last and Smol 2001). Proxies often include fossilized indicators representing the flora (e.g. planktonic and benthic diatoms, aquatic plant macrofossils, pollen) and fauna (e.g. chironomids, ostracods, cladocerans, fish scales) of aquatic systems (Bennion and Batterbee 2007) and select geochemistry (Smol 1992; Bailey et al. 2004).

Sediment cores reveal the legacy of naturally occurring and human induced changes through geochemical signals that provide information about atmospheric deposition, watershed scale terrestrial inputs, and internal processes of lacustrine systems (Olsson et al. 1997; Dean 2002; Yohn et al. 2002; O’Reilly et al. 2005). Thus, sediment cores can help answer questions such as: how stable were systems before anthropogenic impacts, how have lake systems responded to various human activities, including logging, industrialization, etc., and are systems showing recovery with the advent of stricter management? Paleolimnological studies that focus on core geochemisty to investigate reference conditions have mainly employed major elements (e.g. Al, Fe that can indicate landscape stability or erosion) or anthropogenic elements (e.g. Hg, Pb) to evaluate chemical deviation from pre-perturbation, mostly in tandem with biological proxies such as diatoms (Ekdahl et al. 2007). While using select major elements can be effective for specific objectives, including geochemical pollution inventories, these alone are not sufficient for understanding the dynamics of system response (e.g. productivity, toxicity) to specific environmental disturbances (Smol 2002; Battarbee et al. 2005; Bennion and Battarbee 2007).

Interpretation of chemical profiles is at times difficult, and can be additionally challenging due to post-depositional environments where diagenetic processes and bioturbation may convolute the record (Berner 1980; Engstrom and Wright 1984; Long et al. 2010). Nevertheless, advances in analytical techniques, such as inductively coupled plasma mass spectroscopy (ICPMS) encourage multi-element data analysis by providing trace element data that can untangle many of the interpretation complexities. Recent work by Yohn et al. (2004) and Long et al. (2010) demonstrate that 'classes' of elements (e.g. terrestrial, carbonate, diagenetic and anthropogenic) reliably track system response because groups of like elements tend to have similar source and in-lake biogeochemical behavior, and thus behave in similar ways.

Categorizing classes of elements in a multi-element approach can give another line of evidence to validate biogeochemical behavior and help to understand overall system response to change more clearly. For example, in pristine environments where ecological and planetary components are the only components (see Fig. 1.1), certain elements may be coupled (demonstrated by having similar profile trajectories); then, with the influence of a third, human component (whether it be from a terrestrial disturbance or atmospheric deposition), the elemental response changes, reflecting a different trajectory with new trends forming in concert with changing biogeochemical dynamics.

The reference condition concept, as discussed here, has been applied to most freshwater ecosystems, including rivers, lakes and wetlands, and is used to identify conditions of low impacted environments (Karr 1991, Bennion and Battarbee 2007). Furthermore, it has been incorporated into national and international programs for monitoring and assessing environmental conditions (Dodds et al. 2004; Dodds et al. 2006, Leira et al. 2006). The United States Environmental Protection Agency's Clean Water Act (CWA) uses the reference condition as a restoration target, while the European Union (EU) has the reference condition concept as an explicitly stated feature of the Water Framework Directive (WFD) (Stoddard et al. 2006). The pre-industralization condition *c* AD 1850 is commonly used in both Europe and North America to define reference conditions (Andersen et al. 2004; Bailey et al. 2004; Bennion and Battarbee 2007). Examining reference conditions through the lens of geochemical classes untangles some of the complexity of multi-element analyses, and improves insight to ecological response, including biogeochemical processes. Ultimately, this may lead to the development of a cost-effective toolbox of techniques for assessing water quality conditions and setting realistic remediation targets.

Generally, a comparison of the top and bottom portions of a sediment core is used to establish the 'before' (reference) and 'after' (present) condition of the water body, indicating the magnitude of change from reference (Dixit et al. 1999; Garrison et al. 2005). However, examining only the top and bottom of an undated core does not ensure reaching 'reference', nor does it lend insight to the biogeochemical dynamics of system response to anthropogenic disturbances occurring in the rest of the core. Therefore, an examination of the geochemistry from a complete core, particularly at high resolution, offers the best potential for identifying the reference state and for understanding how the system adapted through time.

Role of geochemistry to reference conditions

Investigating sediment chemical chronologies is a useful approach for comparing contemporary and pre-disturbance environmental conditions, thus providing insight to the ecological response of past, present, and future anthropogenic influence (Engstrom and Wright, 1984; Smol 1992; Dean 2002; Smol 2002; Andersen et al. 2004; Reimann and Garrett 2005). Establishing an elemental baseline (e.g. the chemical reference condition) can differentiate natural inputs from anthropogenic inputs (by subtracting the anthropogenic concentration from the pre-anthropogenic value) and thus establish the reference condition concentration for an element. A firm understanding of how systems temporally and mechanistically respond to stressors is important to assess environmental health, and to target restoration efforts (Bennion et al. 1996; Engstrom et al. 2006). To this end, profiles of sediment geochemistry lend insight about environmental response that can facilitate an understanding of the trajectory of future environmental conditions. The goal of this study is to examine the variability of system recovery as shown in historic vs. modern geochemical enrichments, and how systems respond and adapt through time, to identify geochemical response signals to specific events (e.g. logging), as well as the influence of multiple, simultaneous anthropogenic stressors. Important to note for this study is that both ecological degradation and recovery efforts are considered human induced environmental change, and are equally significant under the umbrella of understanding environmental dynamics in this study.

A conceptual example of systems with adapting reference conditions is shown in Figure 2.1. This figure conceptually depicts environmental change in response to multiple stressors, and corresponding ecosystem recovery resulting from implementation of management strategies. The trends depicted on this figure can be described using a generalized example of anthropogenic disturbance in aquatic ecosystems in the Great Lakes region. The first major perturbation was the catastrophic impact of logging (stress 1), which disturbed the terrestrial environment and increased erosion (response 1). As forest resources became depleted, the landscape began to "recover" from logging and erosion decreased. However, afforestation takes decades or centuries, and the legacy of catastrophic clear-cutting had an effect on succeeding land cover. For instance, in a later span of time, increases in agricultural and urban land uses (stress 2) has led to increased nutrient inputs (response 2), which resulted in cultural eutrophication. This second stress compounds on the first stress, and the system is unable to return to a pre-perturbation state. Though environmental legislation in the 1970s mandated a nutrient flux decrease to aquatic ecosystems from point-sources, non-point sources are more difficult to mitigate and continue to influence the overall system. This results in the reference condition adapting through time.

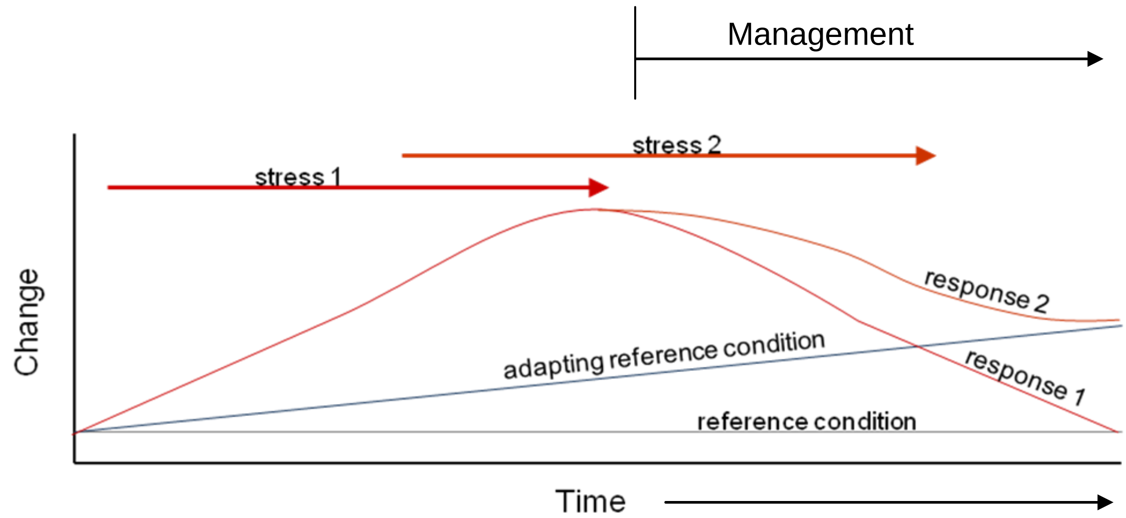


Figure 2.1: Diagram depicting ecosystem response to various stressors and the change in reference condition targets as a function of environmental legacy to anthropogenic stress. Modified from Battarbee (1999) and Battarbee et al. (2005).

The stressor/response relationship is the foundation of the adapting reference condition concept. This is particularly relevant in light of the magnitude and duration of human disturbance, and the superimposed predictions of climate change. While it is a challenge to interpret or predict what the long term recovery from environmental stressors may look like, it is possible to define the system's prehistoric reference condition, identify deviation from reference to determine ecosystem response to stressors, and subsequently assess current ecological health. This study utilizes geochemical profiles to determine if: 1) is it possible to identify stressor/response relationships when multiple stressors are present at once? 2) are scenarios of modern ecosystem equilibrium likely to resemble historical reference conditions due to persistent perturbations?, and 3) is the adapted reference condition stable?

Approach

This study uses a multi-element geochemical approach to gain insight into the temporal dynamics of the system. The role of geochemical reference conditions are explored as a tool for understanding long-term ecosystem response to anthropogenic activity. The state of the system before significant human perturbation will be examined for trends that identify the environmental condition of the system under a two component system (planetary and ecological – see chapter 1). Then, these conditions will be compared to the subsequent state(s) of the three component system where the human influence is considered. From this, change from reference will be determined to assess system response and magnitude of change through time.

The hypotheses for this study are that, 1) in disturbed ecosystems the sediment geochemistry has changed from a historic geochemical reference condition; and 2) that due to the legacy of human perturbation, elemental and proxy group trends have not obtained a new state of balance. To test these hypotheses, a sediment core from Muskegon Lake, Michigan was analyzed for a suite of 23 elements. If true, then we would expect the geochemical profiles to reflect changing trajectories during periods of specific secular/human influence. Grouping elements allowed an examination of ecosystem response using inter-elemental class relationships (e.g. terrestrial, anthropogenic and carbonates) that identify 'fingerprint' signals of specific human perturbations (e.g. logging = increase in terrestrial sourced material) (Long et al. 2010). In this way, the study could explore the effect of multiple stressors/influences, including recent environmental regulations. Evaluating environmental response to perturbation events improves the understanding of relationships among elements, among classes of elements, and infers geochemical processes associated with a range of anthropogenic activity (Smol, 2002; Bennion and Battarbee 2007). Importantly, the advanced insight developed in this study can be exported to other systems influenced by human activity, and are particularly beneficial to developing areas where similar issues are now emerging.

2.2 Methods

A description of the study area, and core collection methods are described in Chapter 1.

Laboratory Analyses

After collection, sediments were stored on ice and transported to MSU's Aqueous and Environmental Geochemistry Laboratories where they were double homogenized, placed in whirl-pak® bags and frozen. In preparation for chemical analysis, sediment were freeze-dried in a lyophilizer, digested (U.S. EPA Method 3051) using nitric acid in a CEM-MDS-81D microwave (Hewitt and Renyolds, 1990) and then filtered through an acid-washed, E-pure (Barnstead International, Dubuque, IA) water rinsed 0.40 µm polycarbonate filter (Nuclepore™). Sediments were analyzed for a suite of metals and metalloids including Mg, Al, Si, P, K, Ca, Sc, Ti, V, Cr, Mn, Fe, Co, Ni, Cu, Zn, As, Se, Sr, Mo, Cd, Sn, Ba, Hg, Pb, U. All samples were analyzed on inductively coupled, plasma, mass spectrometer (ICP-MS) with hexapole technology (Micromass). Separate analyses of freeze-dried sediments for total mercury were performed on a Lumex R-915+

Zeeman corrected atomic absorption spectrometer with an R-915 pyrolyzer attachment (www.ohiolumex.com). This instrument and method is approved for mercury analysis under EPA Method 7473. For all chemical analyses, analytical blanks and standard reference material (NIST SRM 8407 Buffalo River Sediment) were utilized to maintain quality control and quality assurance. Vanadium data were not collected for all samples in this core. Results below the detection limits for each element were eliminated from the data.

Chronology

Establishing core chronology for Muskegon Lake presented unique challenges since Muskegon Lake is a large fluvial lake with a mean annual inflow of 55.5 m^3 s^{-1} (Carter et al. 2006), and the endpoint of the second largest watershed in the state of Michigan (Steinman et al. 2008). Variable inputs make Muskegon Lake a complicated system and therefore difficult to date, since the rate of supply of sediment was likely not constant. As a consequence, several dating techniques were integrated to determine core chronology.

The first approach was to use accelerator mass spectrometry (AMS) radiocarbon technique to obtain a chronology. Two wood samples from the Muskegon Lake core, from depths 110 cm and 101 cm, were sent for analysis to Beta Analytic Inc. in Miami, Florida, USA. Samples below (older) these depths could not be included because they did not contain viable material for ^{14}C testing. The resulting ^{14}C dates from 101 cm and

110 cm depths were converted to calendar dates (AD) through calibration with the IntCal04 calibration curve using CALIB 5.0 software (http://calib.qub.ac.uk/calib).

The second chronological technique employed was a quantitative comparison of anthropogenic elements from a ^{210}Pb dated short cores collected in the same depositional basin; one collected in 2006 (43 cm) and one collected in 2003 (52 cm). The short cores were collected as part of the Michigan State University/Michigan Department of Environmental Quality Inland Lakes project (Parsons et al 2004; 2007). The ^{210}Pb analyses took place at the Freshwater Institute, in Manitoba, Canada. The data were validated using ^{137}Cs peak, derived from atomic-bomb testing that peaked in about 1963. Elemental concentrations of anthropogenic elements in ^{210}Pb dated short cores (e.g. stable Pb, Mo, Cd, Cr, and Hg) were used as 'indicator elements' to correlate peaks/trends in the long core for corroborating chronology. However, because the shorter ^{210}Pb dated cores did not reach prior to 1955, this method is only reliable in the top section of the core. Merging these approaches, combined with an appraisal of event based indicators from the watershed, such as logging, allowed for a qualitative approximation of chronology.

Sediment chemical concentration analysis

Elemental concentration data from the Muskegon Lake core were manipulated in several ways. First, geochemical data were divided into four classes based on source and known environmental behavior (e.g., export, redox, anthropogenic and productivity). Categorizing elements in this manner has been a useful approach to infer physical, chemical and biological processes in other studies examining environmental response to anthropogenic activity (e.g. Yohn et al. 2004; Long et al. 2010). In this study, terrestrial export proxies include Al, Mg, Ti and K that are influenced by physical processes, such as erosion from the landscape (Dean et al. 2002; Yohn et al. 2002; Long et al. 2010). Redox proxies include Fe, As and Mn that are influenced by post-depositional diagenetic chemical processes (Mortimer, 1942; Berner 1980). Productivity proxies include Ca and P that are influenced by biological processes, and are often used to infer primary productivity dynamics (Dean et al., 2002; Engstrom et al., 2006). Anthropogenic proxies encompass a wide variety of trace elements, including Cu, Pb, Hg, Cr and Zn, among others, which are influenced by human activities (Yohn et al. 2002; 2004). Anthropogenic proxies can also facilitate an understanding of the source and transport of pollutants (e.g. Yohn et al. 2002; Parsons et al. 2007).

Second, sediment geochemical profiles were analyzed to determine prehistoric (geochemical) reference concentrations. Based on the historical timeline of human activity in the Muskegon River Watershed (see Chapter 1) and the established chronology of the core (see below), it was determined that samples below the 129 cm depth conservatively preceded significant human influence for the ecosystem. Concentrations of all samples below that depth were averaged to establish a reference concentration for each element. Then, the deviations from those concentrations were followed throughout other phases of the core. Specifically, these values were compared to peak concentrations that, for many elements, occurred during a phase of significant human influence. The reference values are also compared to modern concentrations that were averaged from the top 8 cm of the core. Geochemical inventories were calculated to using the following formula:

$$\text{Inventory } (\mu g/cm^2) = \left(\Sigma \left[C * ((1 - \varphi) * \rho * d)\right]\right)$$

Where:

C = concentration of chemical at an increment (mg/kg)

Φ = porosity (%)

ρ = bulk density (g/L)

d = thickness of increment (cm)

These comparisons determined not only the magnitude of human influence, but allowed the current state of the system to be assessed within the context of pre historical conditions to better frame an understanding of modern trends.

Geochemical profiles of specific, mainly anthropogenic elements (e.g. Pb, Hg, Cd, Sn, and Mo) were considered in detail to evaluate the trajectory of human influence, and the environmental legacy of human activity in the Muskegon area. Using the profiles of known contaminants to Muskegon Lake allowed the analysis of the system response to, and recovery from, particular perturbation events (e.g. logging, industry and urbanization). From this, it was possible to determine how modern conditions deviate from what is considered reference.

2.3 Results

Sediment core characteristics are discussed in Chapter 1, with Appendix I giving a 1 cm resolution sediment description. Porosity results are given in Appendix II. A timeline of human activity in the Muskegon River Watershed is also given in Chapter 1.

Chronology

Results from two ^{14}C AMS radiocarbon analyses confirm the difficulty in establishing a chronology for the Muskegon Lake core. The 110 cm depth sample reported a conventional radiocarbon age of 280 years BP +/- 40 years; while the 101 cm depth sample reported as 161.5 +/- 0.5 pMC, indicating that the sample had more ^{14}C than did the modern (AD 1950) reference standard. The unit "pMC" stands for "percent modern carbon" and is used when samples are part of a system respiring carbon after the on-set of thermo-nuclear bomb testing (AD 1950s).

A comparison of the cores indicates apparent differences in sedimentation rates. The analysis of ^{210}Pb dated short cores from Muskegon Lake reveal a sediment accumulation rate of 1711 g/m^2/y (or 1.08 cm/y) for the 2003 Muskegon Lake short core (Parsons et al. 2004) and 1607 g/m^2/y (or 1.02 cm/y) for the 2006 short core (Parsons et al. 2007). Assuming a similar sedimentation rate in the 149 cm core, the sediment at the bottom of the long core would be extrapolated to ~ AD 1868. However, it is evident through event based dating that the core is older than AD 1868, since the bottom of the core has ~30 cm of pre-logging sediment (identified by the presence of wood fragments, indicating the logging era, between 98 – 118 cm depth). This suggests that the long core was compacted during sampling, decreasing the sedimentation rate reflected by the core, and thus capturing a longer temporal record. Therefore, the sedimentation rate is believed to be nonlinear throughout the core, due to compaction and the multiple human disturbances which altered the amount of material washed into the lake, such as erosion from logging. For that reason, geochemical trends and peaks of select elements among the cores were used to compare chronology of the long core.

Comparison of concentrations of Cd and Mo from the ^{210}Pb dated short cores from Muskegon Lake in 2003 and 2006 show a correlation in trends and peak values to the long core that could be used to establish chronology. Figures 2.2 and 2.3 show the correlation comparison for trends and peaks of Mo, Cd, Pb, and Cr. Figure 2.2 highlights the correlations using Mo and Cd. In both the 2003 short core and 2006 long core, Mo has a peak that equates to ~1980 (shown by point A). Point B is represented in all cores by a peak in Cd which corresponds to dates 1964 in 2003 and 1966 in 2006 short cores. The 2003 short core reflects Mo decreasing dramatically at the ~34 cm depth in the long core, corresponding to ~1955 AD, shown by point C. To support these correlations, Figure 2.3 illustrates a comparison using Cr and Pb. Point A shows where Cr has a slight decrease in concentration at ~4 cm depth in the long core, which is dated to 2005 in the 2006 long core (though this is too shallow in the core to be exempt from bioturbation effects). All cores show a decoupling of Cr and Pb (at point B) that equates to ~1978 in the 2003, and ~1974 in the 2006 short cores. Point C is positioned at ~32 cm depth in the long core, and corresponds to 1962 and 1966 using a Cr peak in the 2003 and 2006 short cores, respectively. Using these profiles, it was possible to establish a qualitative chronology for the 2006 Muskegon Lake long core that is shown in Table 2.1.

Importantly, the long core does appear to reach a chemically ‘stable’ background concentration with regards to anthropogenic elements, whereas the shorter cores do not. Additionally, pollen analysis from the 104 cm depth and below show that *Pinus strobus*-type (white pine) is the dominant pollen type, which indicate that this level was prior to logging. This interpretation is supported by the near absence of *Ambrosia*-type (ragweed) pollen, which is a clear marker of disturbance that followed logging (Grimm, 1983). These two pollen signatures in the core suggest that the bottom of the core captured the reference condition, the pre-logging time prior to 1840. This indicates that the 2006 long core reaches system reference conditions (assumed to be pre-logging, or pre-1840).

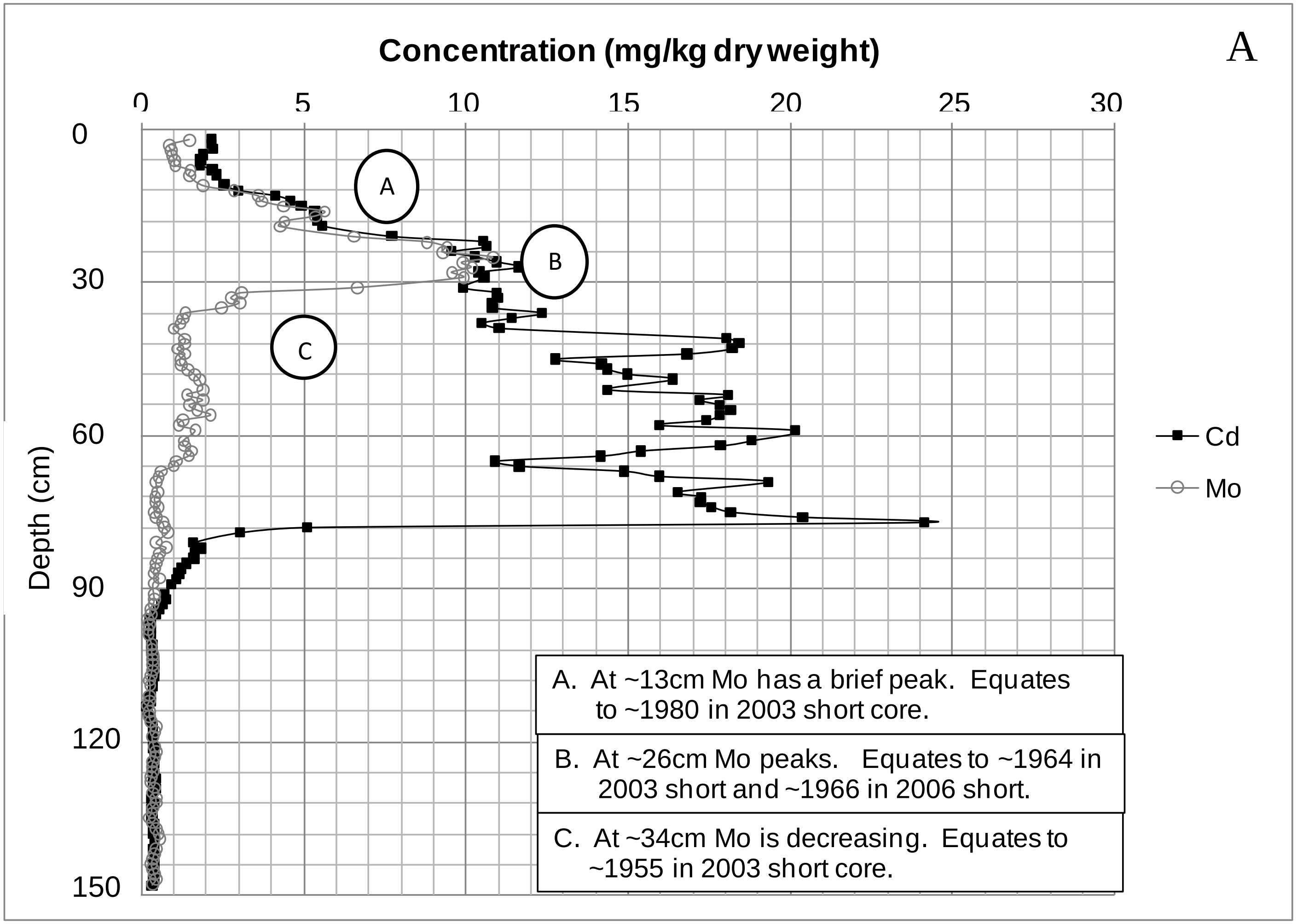


Figure 2.2a: Chronology comparision for Cd and Mo in Muskegon Lake 2006 long core.

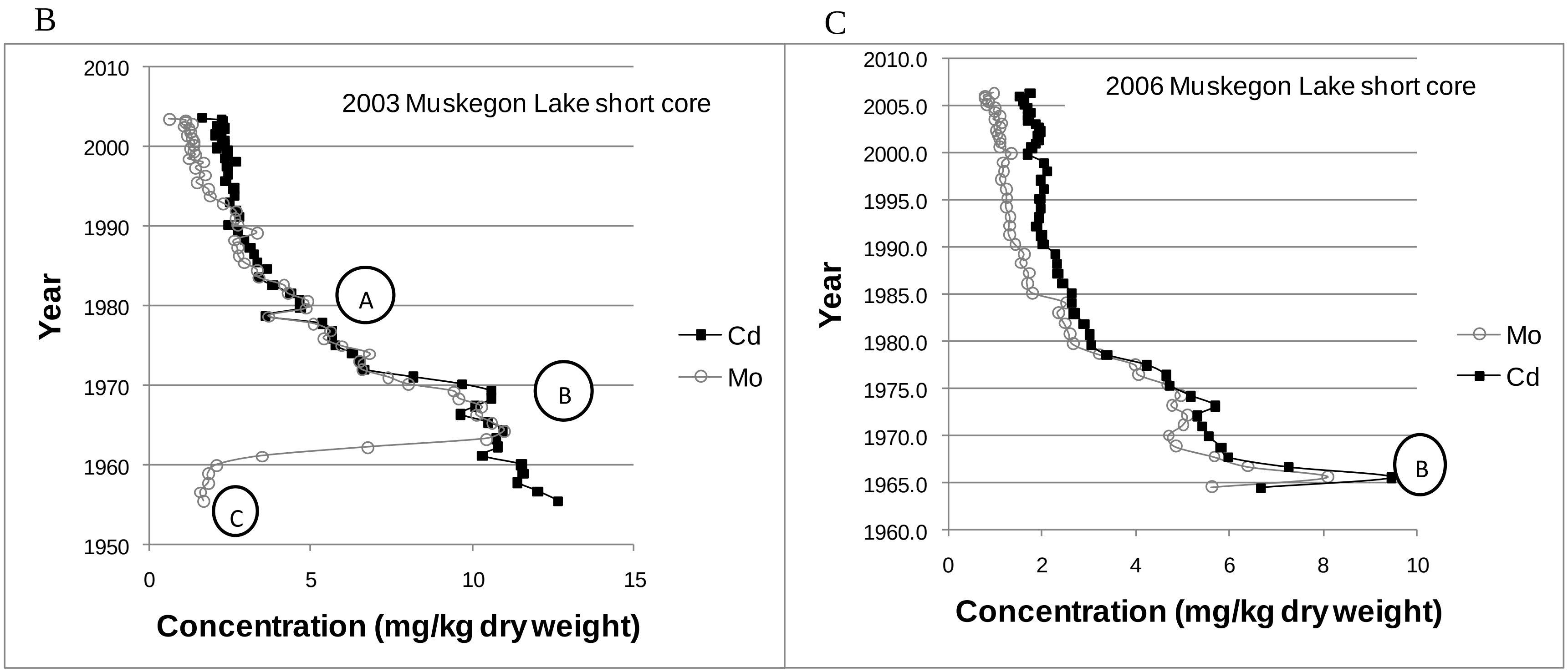


Figure 2.2b-c: Chronology comparision for Muskegon Lake for 2003 and 2006 short cores (see page 58 for comparison with short cores).

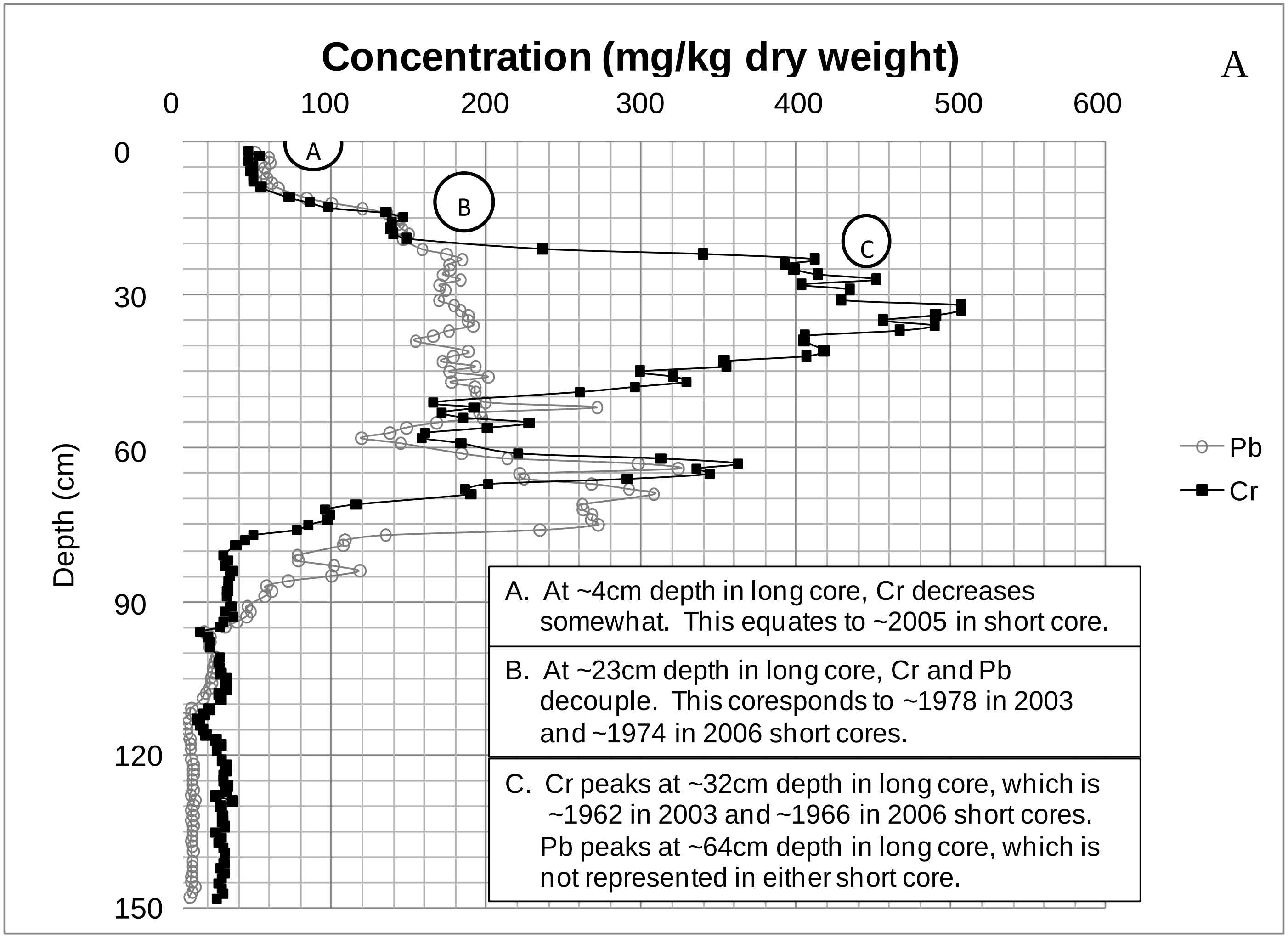


Figure 2.2a: Chronology comparision for Cr and Pb in Muskegon Lake 2006 long core.

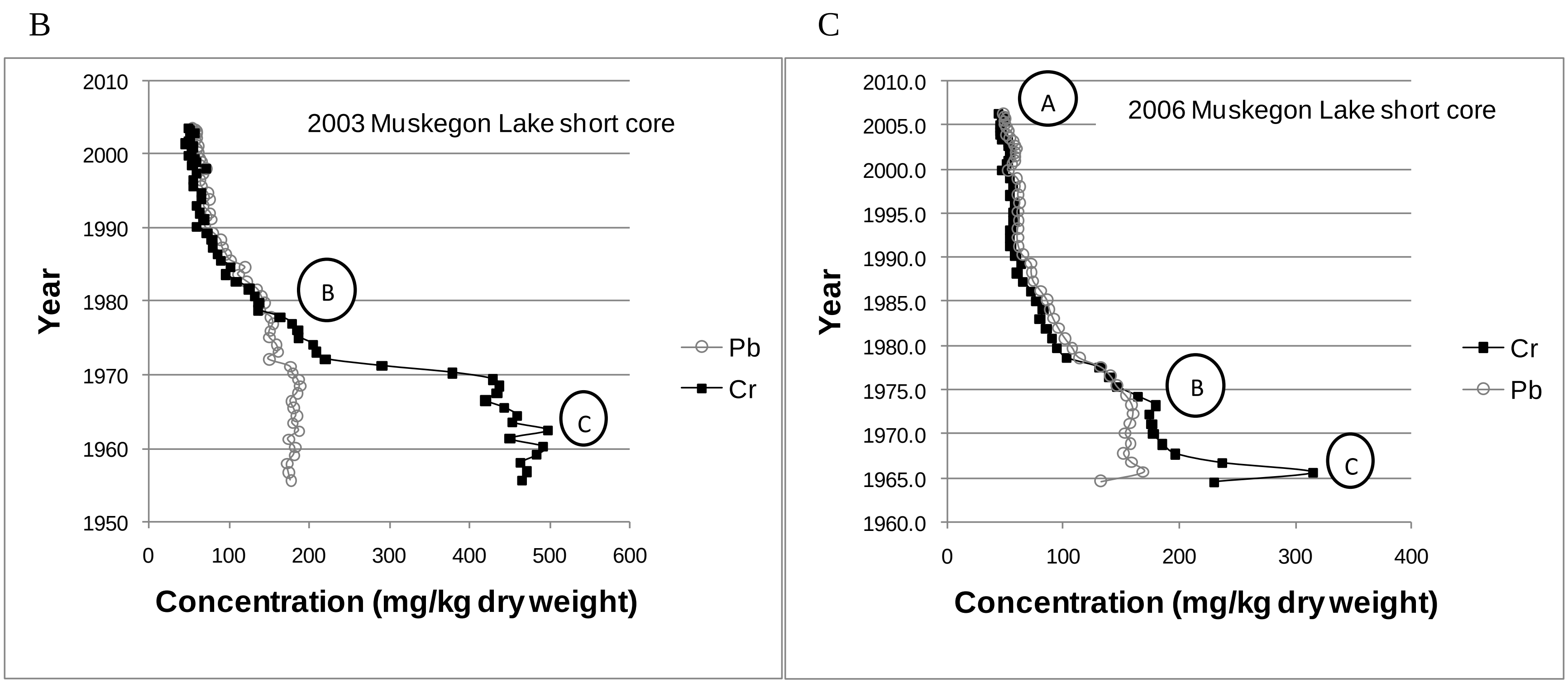


Figure 2.3b-c: Chronology comparision for Cr and Pb sediment geochemistry in Muskegon Lake 2006 long core.

Table 2.1: Chronology summary for Muskegon Lake 2006 long core.

Depth (cm)	Chronology Date (AD)	Method	Watershed event year/decade
0	2006		
4	2005	Cr	2004 flood
10			
13	1980	Mo, Cd	1986 flood
15			
20			WWTP installation
23	1976	Pb, Cr	Newaygo dam out (1969)
25	1965	Mo, Cd	US131 opens(1964)
30	1960	Cr	
35	1955	Mo	
40	1951	extrapolation	
45	1947	extrapolation	
50	1943	extrapolation	WWII
55	1939	extrapolation	
60	1935	extrapolation	Dust Bowl
65	1931	extrapolation	Great Depression
70	1927	extrapolation	
75	1923	extrapolation	
80	1919	extrapolation	WWI
85	1915	extrapolation	
90	1911	no wood	Start of industry/
95	1897.5	Wood	logging
100	1885	Wood/charcoal	logging
105	1872.5	Wood/charcoal	logging/1874 fire
110	1860	Wood	logging
115	1847.5	Wood	logging
120	1835	no wood	Pre-logging
125			
130			
135			
140			
145			
150			

Sediment geochemistry

The full dataset of sediment geochemical concentrations for the Muskegon Lake core is listed in Appendix III. Specific geochemical patterns are reported below, with trends detailed by proxy group, and individually profiles as relevant. The elemental concentrations reveal multiple anthropogenic perturbations (e.g. including logging, hydrological perturbations, and industrialization in the watershed). The core also captures geochemical response to management efforts aimed at mitigating human disturbance, including the Clean Air and Clean Water Acts in 1970. Specifically to the MRW, the 1973 installation of the WWTP shows notable influence on the geochemistry of the core. Recovery due to mitigation practices indicates a shift to a new geochemical state which is different from the apparent reference concentrations. Results of the chemical trends are reported and discussed below.

Indicator proxy groups

Categorizing elements to indicator proxy groups allowed different geochemical relationships and processes to be discerned. The redox indicator group, consisting of As, Fe and Mn shown in Figure 2.4 indicate variability in concentrations throughout the core. The profile trends indicate that the bottom of the core has more variable concentrations than in the middle and top. From 30-90 cm depth, the proxies appear the most coupled, which may indicate a change in oxygen conditions in the lake. Iron and Mn have the most similar trajectories for much of the core as compared to As, with the exception of the very bottom. Important to note is the offset peaks of Mn and Fe at the top of the core which could indicate a modern redox boundary.

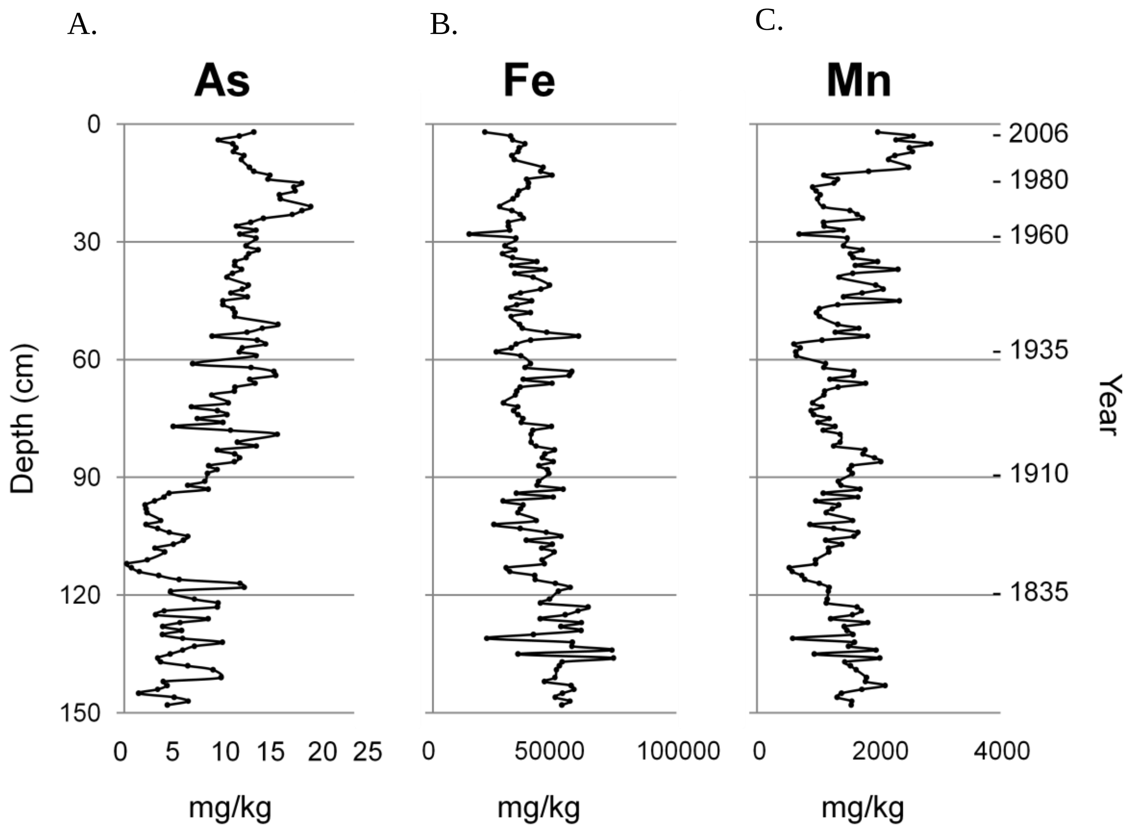


Figure 2.4a-c: Concentration profiles of the redox proxies.

Elements under no or low influence of redox dynamics include the terrestrial, productivity, and anthropogenic groups. The trends of Al, K, Mg and Ti, shown in Figure 2.5, were used to infer terrestrial export from the landscape. Overall, their profiles show temporal variability in concentrations, which may be due to the flow through hydrology of Muskegon Lake. Magnesium and Al are coupled through most of the core (suggesting a similar source and/or similar transport mechanism). Potassium also appears coupled with Mg and Al for much of the core, but becomes decoupled in the top 5 cm as K decreases and Mg and Al increase. Titanium is often used to infer terrestrial export, though in Muskegon Lake the relationship with other terrestrial elements is not strong. All terrestrial elements reflect a decrease in normalized concentration beginning at 118 cm depth. This ‘dip’ is followed by increasing concentrations for ~10 cm.

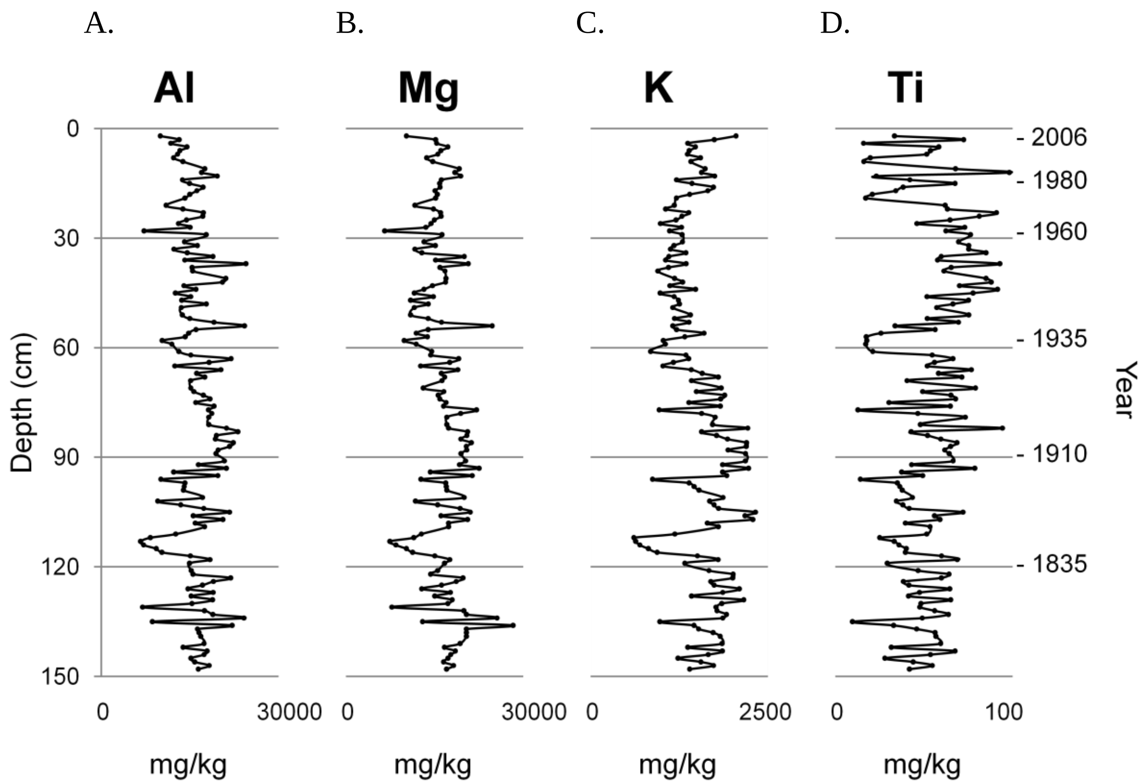


Figure 2.5a-d: Concentration profiles of terrestrial proxies.

Productivity indicators, P and Ca, shown in Figure 2.6 have normalized concentrations that are similar through most of the core. Between 120 and 95 cm depth, they behave similarly, much the same as the terrestrial and redox indicator proxies. The elements appear to behave in a similar manner between 90 – 120 cm depth as they begin to increase in concentration. These two elements continue to move toward peak concentrations, and track one another until ~30 cm depth. A distinct decoupling occurs at ~18 cm, which continues until about the top 3 cm of the core when they have similar trajectories again.

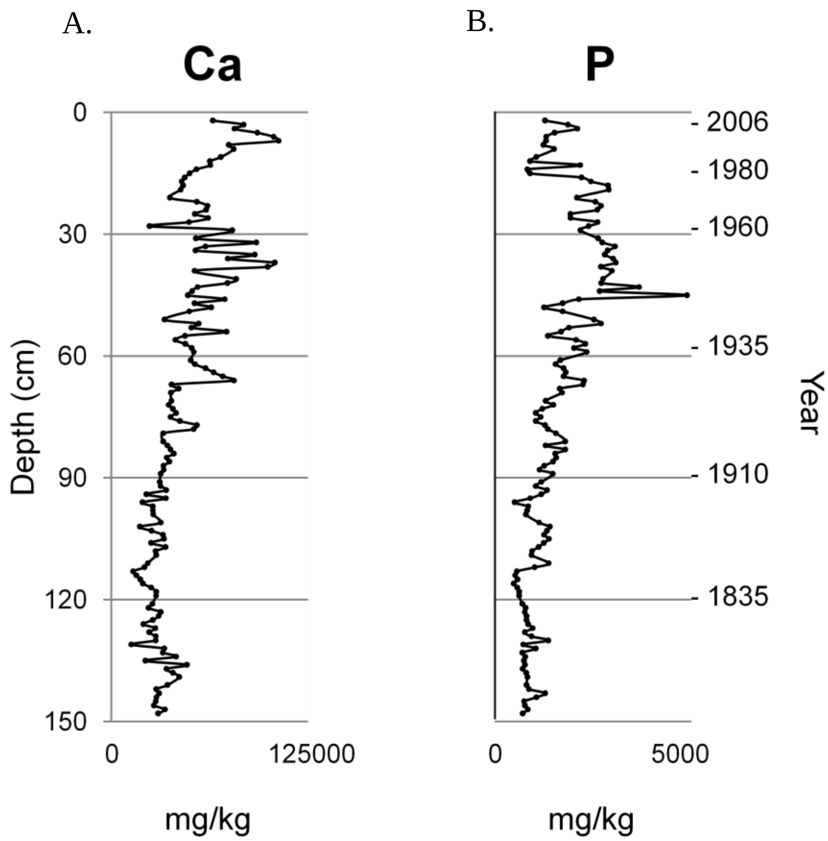


Figure 2.6a-b: Concentration profiles of productivity proxies a) calcium b) phosphorus.

Geochemical reference concentrations

The non-redox group with the most distinct trends is the anthropogenic proxy group. Select elements from this group, Cd, Cr, Hg, Pb and Sn, are shown in Figure 2.7. The stratigraphy of these elements are divisible into three zones, identified as “G” zones, for geochemical stratigraphy, which are illustrated by grey envelopes (Figure 2.7). Zone GI is below the 120 cm depth where the geochemical profiles show relativley aligned and parallel trends with near constant low concentrations. These are considered to be the reference concentrations because they are relatively parallel and low concentrations. The beginning of zone GII, identified by the first observed deviation from the reference concentrations, occurs at about the 118 cm depth. At this depth, the normalized concentrations of anthropogenic proxies decrease slightly, then increase to a small peak at about the 105 cm depth, and decrease again at 97 cm. From the 97 cm depth, the profiles show a generally increasing trend, with many elements having peak concentrations before the 60 cm level. For example, Cd and Hg peak at ~75 cm, and Pb and Sn peak ~ 64 cm. All elements decrease at 60 cm, but generally increase again between 45 and 35 cm. The exception of this is Cd, which continues to decrease slightly, and Cr that peaks at 33 cm. Between ~33 cm to 8 cm, concentrations of Cd, Cr and Sn decrease significantly; though Pb and Hg do not consistently decrease until the 23 cm depth. Mercury increases again with peaks at 19 cm and 3 cm, but the general trend is of decreasing concentrations. Zone G3 is between 25 cm to the top of the core. In this zone, elements are observed to have the same decreasing trajectory. Notably, all elements also show a small increase to a peak ~20 cm. Though concentrations are varied in the top 8 cm of the core, there is an overall increasing trend in concetnrations of most anthropogenic elements to the surface.

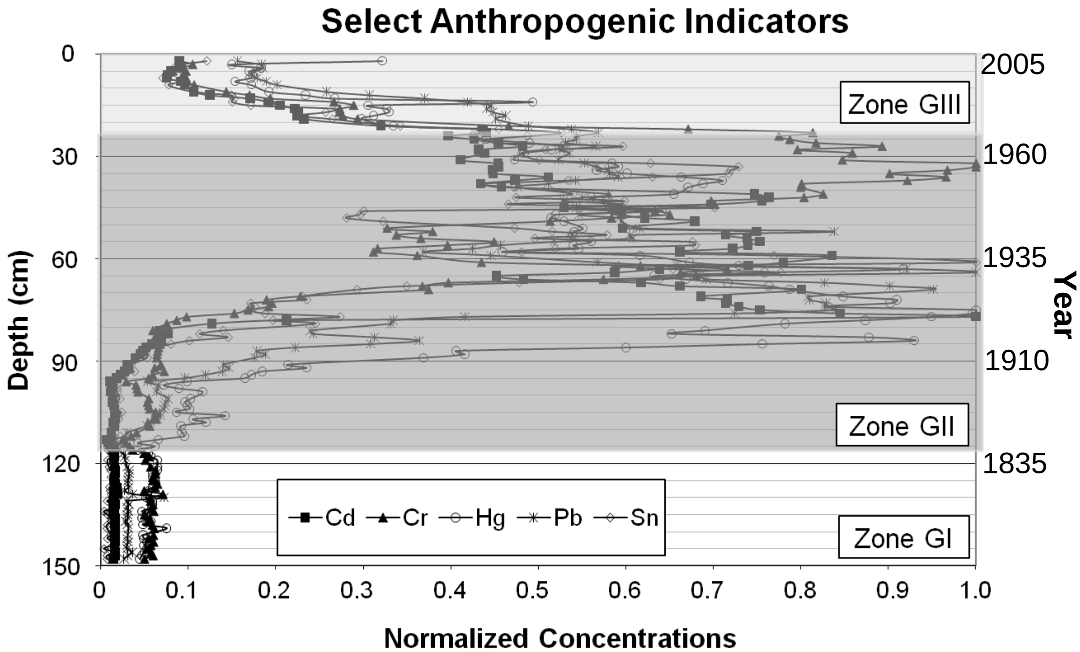
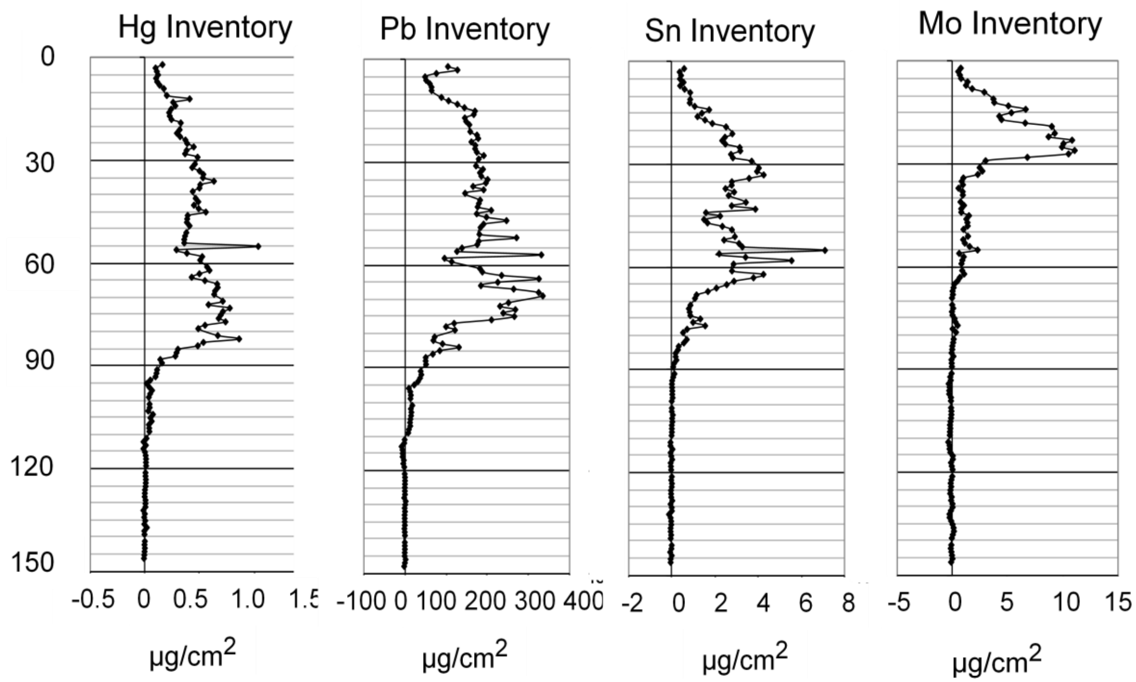


Figure 2.7: Concentration profiles of normalized anthropogenic proxies, delineated into geochemical zones.

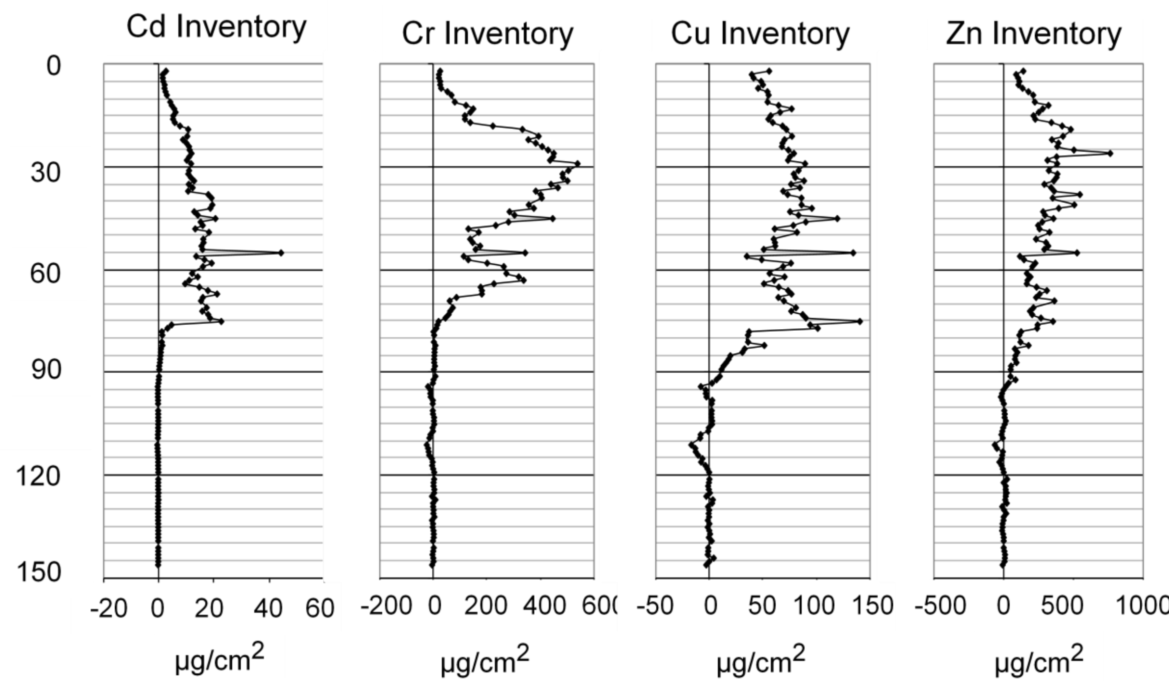
Geochemical inventories associated with human activity

Anthropogenic inventories were calculated for select elements from Muskegon Lake. The values are in µg/cm², and convey the mass of the chemical that has been added to the lake as the result of human activity. Results for anthropogenic inventories of Hg, Pb, Sn, Mo, Cd, Cr, Cu and Zn are shown in Figures 2.8a and 2.8b, and are plotted next to an abbreviated list of dated MRW events. With the exception of Mo, it is observed that the largest proportion of the chemical inventories occur in the top 90 cm of the core after the initial departure from reference concentrations at ~118 cm depth, which approximately dates to the 1840s and 1850s, when logging began and the Newaygo Dam was built (Alexander 2006). Molybdenum inventory increases later in the core, at ~65 cm depth, with a sharp increase and peak between 23-28 cm depth, corresponding to dates in the late 1960s and early 1970s. All elements show a decreasing trend from their zone G3 peaks, with slight episodic increases at 15 cm and 3 cm.



Year	**Watershed Event**
2004	Flood
1986	Flood
1973	Tertiary treatment plant
1960s	US 31 opens and Newaygo Dam out
1940s	WWII
1930s	Great Depression and Dust Bowl
1910s	WWI
1911	Last log run on Muskegon River
1900s	Rogers and Croton Dams built
1901	Muskegon Industrial Age begins
1874	City of Muskegon fire
1854	Newaygo Dam built
1830s	MRW logging begins

Figure 2.8a: Geochemical inventories of anthropogenic elements.



Year	**Watershed Event**
2004	Flood
1986	Flood
1973	Tertiary treatment plant
1960s	US 31 opens and Newaygo Dam out
1940s	WWII
1930s	Great Depression and Dust Bowl
1910s	WWI
1911	Last log run on Muskegon River
1900s	Rogers and Croton Dams built
1901	Muskegon Industrial Age begins
1874	City of Muskegon fire
1854	Newaygo Dam built
1830s	MRW logging begins

Figure 2.8b: Geochemical inventories of anthropogenic elements.

Variability of recovery demonstrated by historic v. modern enrichment

Sediment concentrations of all elements were evaluated to determine historic geochemical reference concentrations to compare with modern concentrations. Not all groups showed indications of geochemical reference, as many redox and terrestrial elements show high variability in the bottom of the core, indicating natural variability not conducive to determining reference concentrations. However, as reported above, productivity proxies, and especially anthropogenic proxies, reveal reference concentrations that do indicate pre-perturbation conditions were reflected in this sediment core. Therefore, profiles of these proxy groups are particularly useful to evaluate environmental change from pre-perturbation conditions, and to assess the state of the modern system conditions.

Geochemical concentrations of sediment deeper than 129 cm were considered pre-perturbation. This sediment conservatively precedes significant logging, settlement or other significant human influences in the watershed. Table 2.2 summarizes geochemical results by illustrating: a reference concentration average for select elements; the depth where trajectories of elements may change; the peak concentration; the total geochemical inventories (ug/cm^2) of anthropogenic elements (Sn, Cd, Pb, Mo, Cr, Zn, Cu, and Hg); and nutrient indicator P, since the advent of human influence. Results shown include all elements, and are grouped by proxy classes slightly different than those reported above in that redox *related* indicators are grouped between anthropogenic and productivity, because of duel influences; and productivity elements include Sr and Ba as they can be influenced by productivity dynamics. A group named ‘other’ includes elements without a clear association to any group. Though there is an overlapping nature of many of the elements (e.g. Mo and As are redox sensitive, but also have anthropogenic sources, and P, which is most commonly a nutrient, is also redox sensitive), this table puts elements in their dominant group.

Results for select elements in table 2.2 (e.g. Fe, U, Mn, Ca, Ba, Sr, Rb, Co, Mg, K, Al and Ti) are not complete due to significant variability and unclear trends in concentration profiles. With the exception of P, inventories for these elements and nickel are not complete for the same reason. Excluding the geochemical change that occurs between 115 and 120 cm depth due to logging, the onset of increasing concentrations varies by element. As seen in figures 2.8a and b, concentrations of many elements (e.g. Cu, Hg, Pb, Sn and Zn) begin to increase between 90 and 96 cm, while Cr and Mo are the last at 80 cm and 65 cm respectively. This variability is assumed to be attributed to differences in source that will be discussed later in this chapter. The depth of the peak of anthropogenic elements varies greatly in the range of 22 – 76 cm. Again, a function of source that will be later discussed. Total elemental inventories are also reported, which lists the total mass of the element deposited to the lake to date. The greatest inventory is for P (101690 μg/cm^2) and the smallest is Hg at 35 μg/cm^2. Peak enrichments were calculated by dividing the peak concentration by the reference concentration. Results show that Sn has the most significant enrichment at 92 times the reference concentration, while Cd, Pb, and Mo are also >30 times enriched over reference. Peak enrichments were compared to modern sediment enrichments, calculated from averaging concentrations from the top 8 cm and dividing by the reference concentration. It is observed that these values are much less than peak enrichments, thus indicating some degree of system response/recovery. However, the modern enrichments do not return to 1, a value which would indicate concentrations similar to reference. This indicates that modern recovery, although significant, does not resemble reference.

Table 2.2: Geochemical Compilation

Proxy class	Element	Reference average (mg/kg)	Depth of first increase (cm)	depth of peak (cm)	Peak conc. (mg/kg)	Peak Enrich-ment	Total inventory (µg/cm2)	Modern average (mg/kg)	Modern Enrich-ment	Status of modern concentrations
Anthropogenic	Zn	119	95	76	1275	10.66	21638	161	1.35	Semi-stable
	Cu	15.23	94	77	161	10.59	5486	52.38	3.44	Semi-stable
	Cr	29.18	80	33	506	17.36	16043	48.89	1.68	Stable
	Cd	0.38	90	77	24.12	62.85	870	2.05	5.34	Stable
	Hg	0.04	95	75-76	0.26	6.07	35.59	0.15	3.56	Semi-stable
	Pb	10.67	96	64	324	30.37	13304	57.47	5.38	Semi-stable
	Sn	0.06	90	61	5.36	92.38	165	0.48	8.34	Semi-stable
	Ni	21.37	96	22	44.58	2.09	--	20.57	0.96	Semi-stable
	Mo	0.37	65	25	10.84	29.03	181	1.10	2.95	Semi-stable
Redox related	Fe	52556	--	--	--	--	--	32286	--	Not stable
	U	0.776	--	--	--	--	--	0.790	--	Semi-stable
	Mn	1542	--	--	--	--	--	2426	--	Not stable
	As	5.24	96	21	19.13	3.65	--	11.53	2.20	Not stable
Productivity	P	860	95	45	4885	5.68	101690	1514	1.76	Not stable
	Ca	31427	--	7	106254	3.38	--	86178	2.74	Not stable
	Ba	156	96	74	443	2.83	--	181	1.16	Not stable
	Sr	37.10	96	7	118	3.18	--	103	2.79	Not stable
Other	Rb	18.41	--	93	25.89	1.41	--	14.44	0.78	Semi-stable
	Co	9.53	--	93	11.62	1.22	--	7.15	0.75	Not stable
Terrestrial	Mg	18371	--	54	24736	1.35	--	14742	0.80	Not stable
	K	1645	--	105	2323	1.41	--	1563	0.95	Not stable
	Al	16466	--	54	24222	1.47	--	12540	0.76	Not stable
	Ti	47.63	--	12	98.18	2.06	--	43.55	0.91	Semi-stable

2.4 Discussion

Chronology

It is possible to reconcile discrepancies with the results of two AMS analyses, though this method is ultimately not that useful for establishing the chronology of this sediment core. The results indicated a 200 year difference in 9 cm of sediment, which is unlikely given the documented high sedimentation rate in Muskegon Lake (Parsons et al. 2006). Instead, it is likely that the 101 cm depth sample was material that got pulled down from shallower in the core during sampling (Darden Hood, personal communication, 2008). It was also concluded that the 110 cm sample, which reported a 280 ±40 year age, was in fact that old - despite it being deposited approximately 145 years ago. Conceivably, this occurs if a wood sample from the inner ring of a tree harvested from an old growth forest was deposited during the logging era in the Muskegon River Watershed. To further support the event based dating, abundant "wood chip", "saw dust", and "charcoal" type material was identified between 94 and 115 cm depth, and was interpreted as deposited during the logging era. According to Wells (1978) and Alexander (2006), clear cut logging is event dated to AD 1830s – 1900 in the Muskegon River Watershed. Overall, chronology established using geochemical correlations from the ^{210}Pb dated cores, as well as watershed events (e.g. onset of industry, etc.) offered good indicators for qualitative dating.

Proxy group patterns represent a geochemical reference state

Grouping elemental geochemistry into indicator proxy groups proved useful in interpreting the Muskegon Lake core. Though the redox and terrestrial proxies were not as valuable as productivity and anthropogenic proxies at identifying geochemical reference conditions, they did provide evidence for perturbation events and specific system dynamics. The redox proxies did reflect a coupling/decoupling dynamic, seemingly in response to human influences; while the terrestrial proxies were variable throughout the core, likely the result of flow-through dynamics of the lake system. Taken together, the patterns of the proxy groups represented the reference state of Muskegon Lake. Specific results of how the groups tracked perturbation response are discussed below.

Overall, the terrestrial proxies were unclear in identifying reference conditions, as the concentrations of the elements in the bottom of the core (identified to be reference) were highly variable (refer to Figure 2.5). This is likely due to Muskegon Lake being a large drowned river mouth system that has such significant inputs that the record becomes convoluted. However, these is a similarity to the trends for Al and Mg, indicating that they have similar source or transport mechanism (e.g. input from Muskegon River and other tributaries). Potassium and Ti were not as closely coupled to Al and Mg, though the K profile did show a similar trajectory to Al and Mg for most of the core, with the exception of the top 5 cm. Titanium was least associated with the terrestrial proxies overall, instead showing an increase in concentration during the industrial phase (see Figure 2.11) of the core, as well as high variability in the top 25 cm of the core, suggesting a significant anthropogenic influence in this system. Due to variability, it may be useful to consider the mean concentration of these elements can be used to establish a reference state.

Important to assessing geochemical response to perturbation events in this system were the trajectories observed during the logging era. Terrestrial elements followed similar concentration changes during this time (~90-120 cm depth), suggesting that their concentrations were influenced by similar process. An unexpected decrease in concentrations occured before and after significant logging. Possible explanations of this decrease include the following. First, it is possible that decreased sediment influx resulting from altered hydrology, and thus the flow of material to Muskegon Lake, from the installation of Newaygo Dam may have decreased the relative concentrations of terrestrial elements. The Newaygo Dam was constructed in 1854 to power the Big Red Mill, and was the first major dam on the Muskegon River (Alexander 2006). Moreover, Rogers Dam, near Big Rapids was installed in 1906, and Croton Dam, also near Newaygo, was built in 1907; both correspond to points in the core where there decrease in concentrations of the terrestrial proxies. The latter two dams are farther up in the watershed, but could still have a substantial influence on sediment flux.

It is a possiblility that terrestrial elements were diluted due to an increase in the proportion of leached (metal deficient) upper mineral soil horizon layer following above-normal erosion due to initial logging (Davis et al. 2006). The subsequent increase in concentrations would then be due to a new stage of erosion reaching the deeper mineral rich soil, followed by another decrease when large scale logging ceases in the late 1890s and early 1900s. The timing of the installation of the dams correspond closely to the onset of logging in the watershed, making it difficult to untangle the stressor/response trajectory.

Coupling/decoupling trends of productivity proxies may indicate a regime shift

Human activity appears to be correlated with productivity proxies in the Muskegon Lake core, as depicted by coupling and decoupling of the elements. Figure 2.9 shows phases of human activity that correspond to zones of geochemical trends delineated above for anthropogenic elements (see Figure 2.7). Phase GI reflects the pre-perturbation conditions that were identified by P and Ca concentrations that fluctuate around a mean concentration in the bottom 30 cm. Core chronology confirms this depth precedes significant human influence and is considered the system's reference state; and the observed variability in concentrations indicative of Muskegon Lake's inherent flow through dynamics. The pre-perturbation segment of the core does not reflect a clear relationship between Ca and P until ~118 cm depth, which coincides approximately with the start of logging in the watershed; this depth also marks the early part of Phase GII, the anthropogenic phase. Both elements show a decreasing trend near this depth, though there is an offset with the trend as P begins to decrease first at 125 cm depth, then Ca later at 118 cm. The decrease in P that precedes Ca may indicate a process related to nutrient dyanamics or terrestrial inputs. While other studies (e.g. Long et al. 2010) have reported a relationship between Mg and P that indicates a terrestrial source, this core does not reflect a coupling of this pair. As such, this is unlikely the result of variable terrestrial inputs as discussed above and more likely reflects in-lake processes play a greater role in driving P biogeochemical dynamics. Productivity is a major factor of in-lake processes, which can be examined in the patterns of Ca and P after logging; and through the correlation in the trends of Ca and Mg through most of the core.

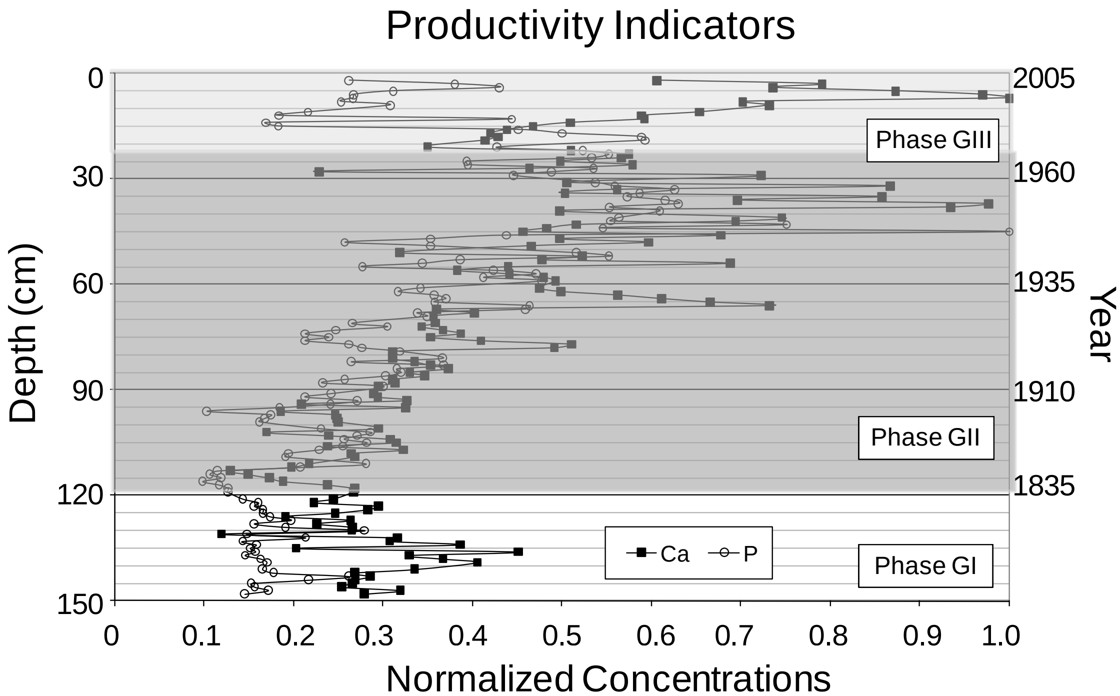


Figure 2.9: Geochemical phases of productivity proxies.

From a core depth of 90 to 25 cm, the productivity proxies track one another and seemingly reflect the response of changing nutrient conditions in Muskegon Lake, as concentrations increase in this portion and are considerd consistent with increasing productivity. Phosphorus and Ca have aligned trajecories in this segment (also part of phase GII) attributed to increased photosynthesis resulting from increasing nutrient inputs. This occurs because increasing water column photosynthesis will consume CO_2 and increase pH. Higher pH values will cause more $CaCO_3$ to precipitate, resulting in increased Ca concentrations in the sediment profiles. Moreover, increased organic matter due to increased photosynthesis will increase the particulate P, which is why these elements together are considered proxies for productivity. An important caveat for interpreting P sediment profiles is that in some systems, anoxic bottom water can release P from sediments to overlying water. While this is not believed to be a factor in Muskegon Lake due to short residence time, historical and seasonal anoxia have been reported and may influence the sediment accumulation of P (Anderson et al. 1993). Based on the relationship observed between P and Ca in this segment of the core, the increases are more likely related to productivity than redox processes. However, the decoupling of Ca and P at 18 cm depth deviates from this assumption.

At the 20 cm depth, Ca and P are inversely coupled, as P concentrations decrease and Ca concentrations increase. This depth is the start of Phase GIII, which is considered to be the "recovery phase" since it corresponds to environmental regulatory efforts. These efforts were designed to reduce nutrient and contaminant loading to Muskegon Lake.

The Muskegon Waste Water Treatment facility was installed in 1973 (at ~23 cm depth), which is near the time that the productivity elements shift to inversely coupling. Steinman et al. (2008) reported that this facility was successful in decreasing nutrient imputs as water column TP values declined from 68 - 27 µg/L from 1972 to 2005. Phosphorus concentrations would be expected to decrease with reduced inputs. However, P concentrations begin to slowly increase at 10cm, and may reflect an increase in non-point source nutrients and/or inputs from increased urbanization (see Figure 1.4). The P concentrations in the top 5 cm decrease, but caution should be noted in interpreting this due to the before mentioned redox sensitivity of P; though Ca decreases in the top 5-7 cm as well, so it is possible this indicates the productivity elements are again coupling and re-establishing a relationship following 'remediation' and an oligiotrophication processes (Anderson et al. 2005). Biological proxies in the next chapter will help resolve that unclear interpretation. A comparison of Ca with Ba and Sr, other productivity related elements often correlated with Ca, show related trends throughout the core. However, there is no relationship between them at the top of the core; though Sr is coupled with Ca, indicating similar processes/behavior.

Freedman et al. (1972) measured surface and bottom TP values, and reported that they were ~60 and ~80 µg/L, respectively. These measurements correspond to decreases in water column P documented by Steinmann et al. (2008). The segment between 3 and 18 cm where the productivity proxies are decoupled may reflect a period where the system is responding to changing nutrient conditions.

Geochemical sub-phases reflect human activities

A distinct response to the effects of human stressors was reflected by the anthropogenic proxy group. Sediment geochemical profiles for the elements in this group show consistent low concentrations among all elements in the bottom 30 cm of the core. These concentrations are indicative of the "reference state" before the human component significantly influenced biogeochemical processes (see Chapter 1). Consequently, these concentrations are the baseline against which later concentrations are assessed. This phase is represented by Phase GI in Figure 2.10.

The next phase is GII that begins with GIIA at the 118 cm depth, when concentrations of many elements, not just the anthropogenic, respond to logging. Phase GIIB further illustrates human induced environmental degradation as high concentrations of anthropogenic elements reflect the response to intense industrialization dominate. Many of the contaminants depicted here came from point source discharges or surface runoff as industry replaced logging. This activity continued through WWI and until the Great Depression. During the Great Depression, the area surrounding Muskegon Lake experienced a decline in industrial production which is marked in the sediment core by decreasing concentrations of anthropogenic elements near the 60 cm depth. Following the Great Depression, demand for industrial materials again increased due to WWII, which is reflected by an increase in anthropogenic elements in phase GIIC. Many of the anthropognic concentrations are variable, though remained high until a slight decrease in the late 1960s when a decline in economic conditions decreased much of the industrial productity of the region. Further decreases in inputs of anthropogenic elements are observed in the 1970s after environmental mitigation was legislated and reduced inputs of certain pollutants, indicating the start of the next phase beginning at ~25 cm depth.

The most recent phase is GIII that occurs in the top 25 cm and represents an era of environmental regulation. Concentrations of anthropogenic indicators greatly decrease in this phase. Isolated increases at ~15 and 3 cm depths may represent major floods in the watershed in 1986 and 2004, respectively. Or, the slight increasing trend in the top 8 cm may indicate an increase in non-point sources, or the environmental legacy from intense pollution (being flushed in as it erodes from the landscape). Consequently, this segment of the core is important to understanding system response under management efforts.

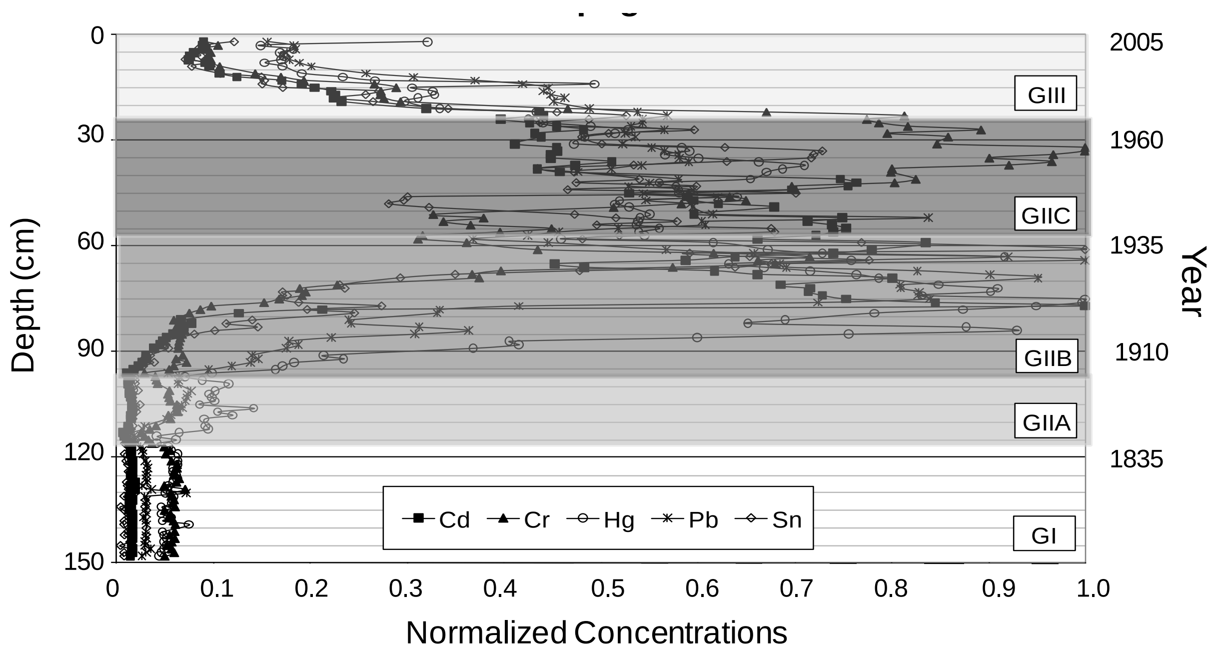


Figure 2.10: Concentration profiles of normalized anthropogenic proxy group elements.

Specific elements lend insight to specific watershed activity

Human influence in the Muskegon River Watershed began with the onset of massive deforestation in the watershed. The human settlements established from the logging era set the stage for growth of industry, as many lumbermen began to foresee the need to diversify as the forest became depleted. Before the turn of the 20th century, several companies began to open mills and foundries as lumbermen in Muskegon exerted their interest in diversification. In the early 1880s, the Lumbermen's Exchange, a trade and pricing group, reorganized as the Muskegon Board of Trade. The board inaugurated a newspaper advertising campaign to promote Muskegon as a place for industry. They brought two new railroads into the area and stressed the need to establish nonlumber-related industries. This strategy worked and several firms were started with capital from lumbermen or with support from the Board of Trade. These companies included Chemical Fire Company, Muskegon Cracker Company, Sargent Manufacturing Company, Chase Brothers Piano, and the Muskegon Rolling Mills (Kilar 1990). The accumulation of many anthropogenic elements can be compared directly to these companies. Notably, enrichments for different elements occur at different depths, indicating different sources and pathways for different elements.

According to a book on industrial activity in 1900 (American Iron and Steel Institute, 1904), industry was becoming established in the Great Lakes region at the turn of the century. For example, as of 1 November 1901, Michigan had a total of 16 Rolling Mills, Steel Works, Tinplate Works etc., while Ohio and Pennsylvania had 855 and 1478, respectively. This trend illustrates the movement of industrialization from east to west, similar to the logging movement decades earlier. The American Tin Company, also owned by the United States Steel Corporation, had the first black plate mills in the Muskegon Works complex, situated on the shores of Muskegon Lake. Added in 1899, this facility housed 6 double heating furnaces, 2 annealing furnaces, 6 single pair furnaces, and six 26-inch hot and five 24-inch cold mills. Annual capacity in 1902 was 12,000 gross tons of iron, steel black plates for tinning, tin and terne plates (terne plates are similar, but the ‘bath’ is not tin, but lead, and therefore not shiny but dull, hence *terne*). Fueling the facility was coal and manufactured gas (American Iron and Steel Institute, 1902). Figure 2.8a shows the profile for Tin, with early increases being attributed to the American Tin Plate Company.

Continental Motors was another early company that contributed to industrial base of Muskegon. Opening in 1905, it is plausible that this company contributed to a significant portion of the metals in Muskegon Lake sediment. They built industrial engines, and bus and truck motors and increased production with military vehicles in WWI. At one time, Continental Motors produced nearly 90% of US aircraft engines with less than 75 horsepower, though the Great Depression of 1930s greatly decreased the demand for this industry. After the Great Depression, Continental Motors built engines for tanks, landing craft and other military vehicles for WWII. Also, beginning in 1943, they made Rolls Royce Merlin engines used in the P-51 Mustang. In the late 1960s, Continental was sold twice, and the downtown Muskegon Plant was removed in 1993. Chromium (Figure 2.8b) remains a common element used in Muskegon industry, and continues today in aspects of manufacturing. Specifically, the Sealed Power Division of the SPX Corporation which is a metal finishing facility in Muskegon that reports its hard chromium electroplating plates had to comply with compliance standards by January 1997 (http://www.nmfrc.org/epadocs/int1996f.htm).

Cannon Muskegon opened in 1969 and is currently still operating. This company is a metallurgical specialist of premium-grade alloys for medical and aerospace casting applications. The website for this company lists Ni, Co, Cr, Ti, Al, Cu and Mo as elements used in their industry. It is evident from the profile of Mo (Figure 2.8a) that it did not have significant accumulations until later, which is different than most other anthropogenic elements (e.g. Pb, Sn) and suggests it was not heavily used until the late 1950s and peaked in the late 1960s/early 1970s.

Evidence of adapting reference conditions

A goal of this study was to shed light on how a system responds and adapts to various human perturbations. The geochemical trends within indicator groups lend insight to the degree and trajectory of change. Though inferring if a system is recovering from anthropogenic perturbation is challenging with a single proxy type (e.g. geochemistry), the geochemical patterns for many anthropogenic elements in Muskegon Lake sediment do become lower in concentration again near the top of the core. This may be interpreted to indicate a return/trend towards a historic reference state; however elemental concentrations are, in many cases, far from pre-disturbance concentrations.

The modern concentrations and trends for many elements (e.g. Cd, Sn, Hg, Pb, Cu, and Mo) are generally much lower than peak enrichments, though still above the reference concentration (Table 2.2). This suggests an adapting reference condition, where the modern condition, because of the magnitude of human perturbation, does not resemble the reference state. Profiles for Sn, Cr and Mo specifically illustrate the trajectory of change for anthropogenic inventories, and depict concentrations near the top of the core that show the geochemical response to regional and local management efforts to reduced loading of pollutants. While Figure 2.11 (below) depicts the adapting reference condition as related to the normalized profiles of Pb, Cr, and Sn, that each have a specific role in reconstructing Muskegon Lake influences, but reinforce identification of multiple stressors. This figure is meant to represent, using real data, the concept presented in Figure 2.1, which was modified from Battarbee et al. (2005). The arrow for stress 1 reflects system response to logging perturbation; the concentration of Pb, Cr and Sn in the figure increase then decrease, though the decrease does not reach the reference conditions baseline. The next stress (2) is the advent of industrial activity and WWI engine production in the Muskegon area, and shows a very significant increase of concentrations in the core. Again, the concentrations decrease when the Great Depression halts much of the industrial activity – though, the concentrations do not reach reference, nor do they decrease to the post-stressor 1 concentration, possibly because the next stressor occurred too quickly for a more substantial/complete recovery. In response to stress 3, concentrations increase again, corresponding to WWII military industry and later, continued automotive related production in the region. Tin and Cr have a greater enrichment with this stress, likely the result of direct industrial activity in the Muskegon area. The recovery segment observed is due in part to an economic downturn in the region, as well as environmental legislation that mandated a decrease in atmospheric and terrestrial inputs.

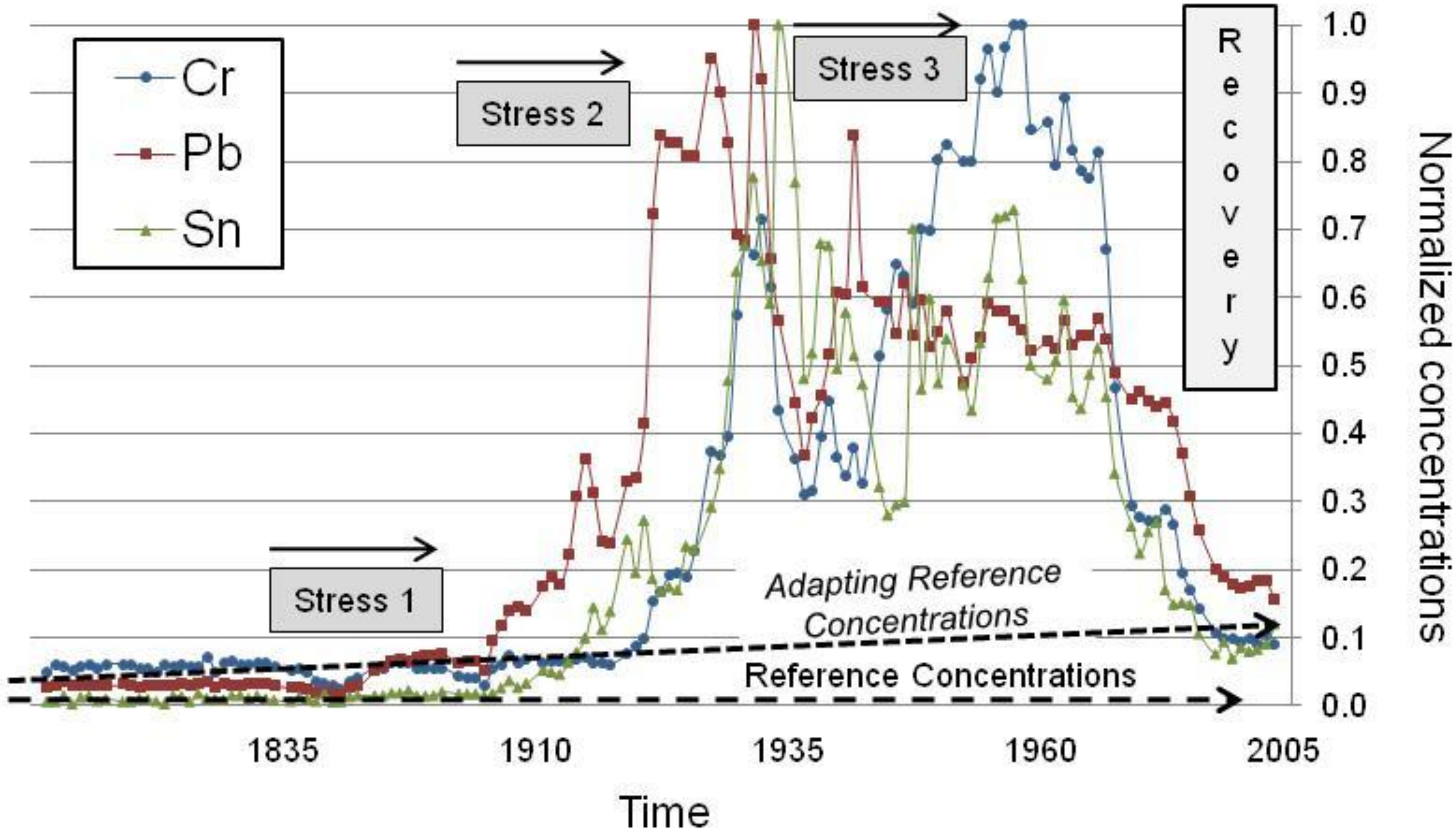


Figure 2.11: Adapting reference conditions using Muskegon Lake geochemical data.

2.5 Conclusions

This chapter applied the reference condition concept to a paleolimnological study using geochemical profiles and trends of individual elements and proxy groups from a highly perturbed lake system. The investigation predicted that modern ecosystem equilibrium was not likely to resemble historical reference conditions. An evaluation of the post-disturbance state of the system relative to pre-disturbance (with consideration of the influence of management strategies) supported the hypothesis. The results showed that, for many anthropogenic proxies, what may be considered the reference concentration has adapted through time with the overprint of multiple stressors.

Policy mandates have dramatically reduced the loading of nutrients and metals and improved the water quality of Muskegon Lake. The current geochemical conditions in the lake core reflect significant recovery from high concentrations of many elements. However, concentrations have not returned to the identified “reference” values but reflect an adapted reference state as evidenced by geochemical profiles and the trends within proxy groups. Specific differences in geochemical profiles within the proxy groups (e.g. anthropogenic and terrestrial) reveal watershed scale sources and are linked to historic human activity in Muskegon, MI, USA.

Due to the adapting nature of the geochemical reference condition evidenced by this study, it is suggested that modern remediation targets need to consider the legacy of human impacts and further be cognizant of emerging stressors. While addition proxy groups (e.g. terrestrial and productivity) were useful in untangling biogeochemical response, particularly in the top of the core when productivity proxies become decoupled, further lines of evidence (e.g. biological proxies) would be useful to validate those potential regime shift interpretations. Therefore, based on these results, there appears to be a need to investigate additional constituents of ecological system components to evaluate system response. Select elements reliably track anthropogenic activity and/or are able to indicate a specific biogeochemical process. Therefore, integrating biological proxies could help gain a better understanding of environmental processes resulting from anthropogenic influence.

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CHAPTER 3

IDENTIFYING PALEOECOLOGICAL REGIME SHIFTS USING FRESHWATER DIATOMS AS PRIMARY PRODUCTIVITY INDICATORS

Abstract

Understanding primary productivity in a system can lend insight to overall biogeochemical dynamics, as it is an essential part of aquatic ecosystem function. This chapter uses fossil diatoms to investigate shifts in primary productivity habitat to better understand the role of the aquatic biological response to environmental change. Based in part on the geochemical results presented in Chapter 2, and continuing with the hypothesis that disturbances from human activities have been so intense that ecosystems cannot fully return to a pre-disturbance state (particularly with the influence of emerging stressors), this chapter addressed the following questions: 1) how do specific environmental changes affect diatom assemblages, 2) what dictates the dominant primary productivity habitat ratio (planktonic:benthic), 3) what is the mechanism within systems that may shift that ratio? Fossil diatoms in a sediment core from Muskegon Lake, MI were used to assess these questions. Each taxon was evaluated individually for specific ecological preference, then, taxa were grouped as benthic or planktonic to investigate broader changes in the productivity regime of the system. The Muskegon Lake core showed variability with the 27 "common" taxa, revealing distinct changes in water quality, and a recent shift in the dominant habitat of primary production. The hypotheis was supported with three temporal phases identified using fossil diatom data: 1) pre-disturbance, 2) anthropogenic disturbance, and 3) recovery. Specific taxa indicated increasing nutrient conditions (e.g. *Fragilaria* spp.), with additional restructuring

occurring in response to the Waste Water Treatment Facility installation, indicating that monitoring of primary productivity trends is critical to ecological health assessments.

3.1 Introduction

Problem

The intensity of global environmental change has been extraordinary in recent centuries, adversely affecting ecological systems worldwide (Ricciardi and Rasmussen 1999; Murray et al. 2004). The status of global freshwater resources in particular has become a prominent environmental concern; with the demand for clean/potable water forecasted to increase. An understanding of how aquatic ecosystems respond to natural and anthropogenic stressors is critical to ensuring both a sustainable supply of potable water, as well as associated ecosystem services (Bennett et al. 2003). However, due to variability of, and within, aquatic ecosystems, questions remain about their structure and function. Specifically, how are systems driven by natural factors (e.g. climate and succession), how do they react to anthropogenic impacts (e.g. cultural eutrophication), and does employing management/mitigation strategies result in system recovery – and what that recovery can be expected to be? Answers to these questions require a long term evaluation of system dynamics, which is not available using modern empirical data, or even decadal scale monitoring strategies (Battarbee et al. 2005).

Paleolimnology is a reliable tool for inferring historical environmental conditions (Smol 2002; Bennion and Battarbee 2007; Ekdahl et al. 2007). In particular, diatoms can be used as paleoecological indicators for reconstructing water quality conditions, such as nutrient status, pH, temperature, and have been shown to successfully reflect anthropogenic impacts (Smol 2002; Wolin and Stoermer 2005; Chen et al. 2008). In this study, the biostratigraphic record of subfossil diatoms from a lake core is used to reconstruct the effects of human activity in a highly impacted watershed in the Great Lakes region.

Diatoms

Freshwater algae are photosynthesizing organisms that react rapidly to changes in solar radiation (light), temperature, and nutrient conditions. In particular, diatoms play a prominent role in paleoecological studies because their valves are opaline silica, which are preserved well in lacustrine sediment compared to soft algae species (Hall and Smol 1999). Thus, diatoms recovered from sediment cores reliably reflect changing environmental conditions (Wolfe et al. 1996; Hall et al. 1997; Hall and Smol 1999; Slate and Stevenson 2000; Battarbee et al. 2001; Smol 2002; Ramstack et al. 2003). Moreover, the biostratigraphic record of diatoms can reconstruct historical (pre-disturbance) water quality conditions (e.g. nutrient loading) because of distinct preferences of individual diatom taxa. Consequently, diatoms have been used to identify ecological reference conditions (Leira et al. 2006; Bennion and Battarbee 2007) and productivity regime shifts (Vadeboncoeur et al. 2001; 2002), both of which have applications to aquatic management and recovery strategies (Smol 1992; Hall and Smol 1999; Smol and Cumming 2000; Smol 2002; Bennion et al. 2004; Batterbee et al. 2005; Leira et al. 2006; Lepisto et al. 2006; Taylor et al. 2006; Carpenter et al. 2008).

Algae, as the base of the food web, play a significant role in the biogeochemical cycling of nutrients and are sensitive to nutrient loading (Wetzel 2003). Shifts in biomass or species composition indicate not only a change in external environmental conditions (e.g. nutrient inputs), but also lend insight to internal lake process (e.g. feedback mechanisms) (Vadeboncoeur et al. 2001). Notably, shifts in primary productivity habitat, including benthic and planktonic, reveal shifts in ecological dynamics; since primary productivity is an important factor in trophic transfer, and thus, the flow of energy through a lake system (Vadeboncoeur et al. 2002).

A systems internal response from an external stressor, such as cultural eutrophication can be complex (Scheffer and Carpenter 2006). Using diatoms as an ecological indicator, the analysis of individual taxa dynamics and dominant habitat lends insight to complexity of the system (Vadeboncoeur et al. 2001). For example, it is known that increasing nutrient inputs increases the productivity of the water column (the pelagic zone) (Vollenveider 1976, Schindler 1978). Cultural eutrophication often leads to a higher abundance of planktonic diatoms, which sequesters nutrients and blocks light from reaching the benthic habitat. This decreases the relative benthic productivity; and thus, changes the productivity regime of the system (Vadeboncoeur et al. 2001; 2002).

An examination of the complex dynamics with biological interactions provide important insight to both the diatom assemblage composition at any given time, as well as lend insight to primary productivity habitats (Dokulil and Teubner 2005). During times when a system is transitioning between states (the spectrum of oligotrophic to eutrophic regimes), phytoplankton composition does not always track the typical succession expected using the chemically determined nutrient status, as phytoplankton response – particularly to oligotrophication – is often delayed; this apparent hysteresis is not well understood (Sas 1989; Dokulil and Teubner 2005). Furthermore, phytoplankton community structure shows a lag time response to oligotrophication even when phosphorus and Secchi depth parameters reflect a concomitant change during restoration efforts (Dokulil and Teubner 2005). Further understanding whole-ecosystem dynamics is important to identify primary productivity regime shifts, as triggered by natural or anthropogenic activities.

Regime shift approach

In ecological systems, a regime is considered to be a set of conditions that remain relatively stable and persistent over time (Carpenter 2003; Scheffer and Carpenter 2003; Carpenter et al. 2008). These can be identified by examining the temporal composition and diversity of the biological community (Das et al. 2009). The concept of ecological regime shifts includes consideration of thresholds and resiliency, often driven by internal feedback mechanisms (Carpenter 2003; Scheffer and Carpenter 2003; Walker and Meyers 2004). Figure 3.1 shows a hypothetical system response given successive perturbations. This figure illustrates how systems have the capacity to recover from a single perturbation event, but subsequent perturbations may push a system past a threshold, and kick it into a new regime (Scheffer et al. 2001; Carpenter 2003; Dearing et al 2006). In other words, a system has a certain resiliency to perturbations and can resist major change to a degree, but under persistent or multiple stressors, the system reaches a threshold which tips the ecosystem to a different state. Ecological variables, as demonstrated in this figure, can act as indicators of ecological regime shifts, and their temporal dynamics offer clues to system function (Scheffer et al. 2001; Carpenter and Brock 2006).

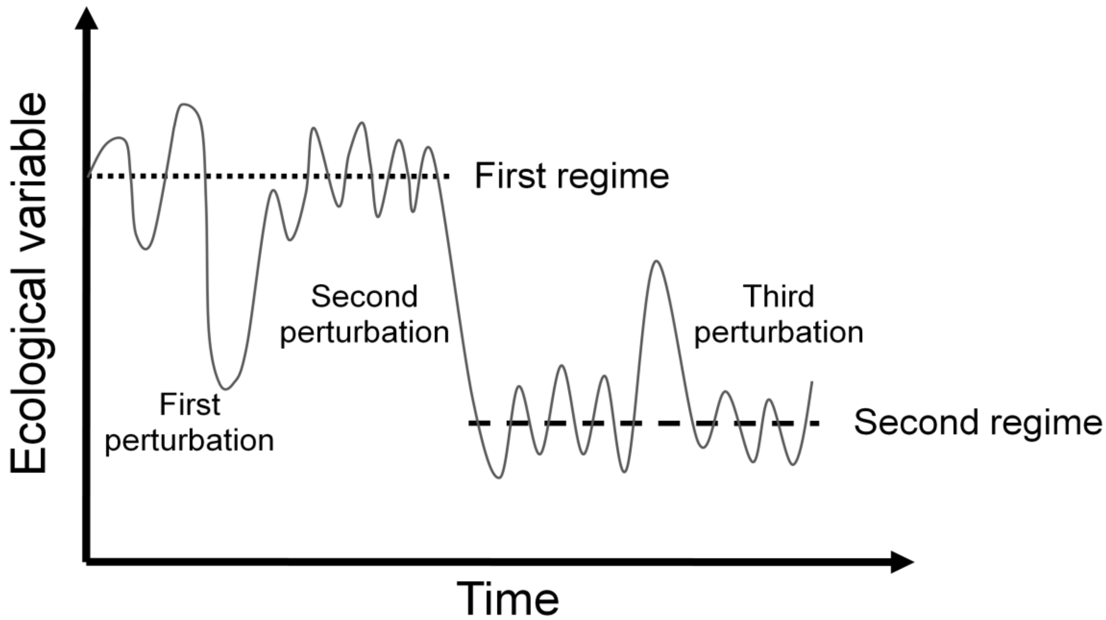

Figure 3.1: Hypothetical conceptual diagram of regime shifts (modified from Carpenter 2003).

This chapter considers shifts in ecological regimes through the lens of primary productivity habitats as the ecological variable. Specifically, this study examines the ecological response to successive waves of environmental perturbation, and the effectiveness of attempt(s) to mitigate human perturbations. Two distinct ecosystem regimes (states), including benthic dominated production and planktonic dominated production, are examined as they lend insight to system response (Vadeboncoeur et al. 2001; McGowen et al. 2005). Because of the relationships described above, lakes dominated by benthic production are typically reported as clear water systems associated with oligotrophic to mesotrophic conditions (Vadeboncoeur et al. 2002). Conversely, planktonic production dominated lakes are associated with increased nutrient inputs, turbid water, and potential anoxic conditions resulting from cultural stressors of urban development and agriculture (Vadeboncoeur et al. 2002; Wetzel 2003). Another factor important to the habitat of productivity is lake morphology, with the proportion of planktonic dominated productivity typically higher in large deep lakes (Wetzel 2003).

Table 3.1 depicts the relative contribution of primary producers under eutrophic and oligotrophic conditions. Conceptually, primary productivity habitat can change the flow of energy in a system (e.g. which begins with and where algal production is greatest), and may influence the overall trophic structure (Scheffer and Carpenter 2003; Scheffer et al. 2007). For example, decreased algae productivity in the water column may result in more herbivores, less planktivores and more piscivores in open water (Carpenter et al. 1985; Wetzel 2003). This chapter considers these associations with the intent of understanding ecological dynamics and the consequential significance of internal feedbacks in systems influenced by human stressors.

Table 3.1: Relative contribution of primary production in lakes

	Oligotrophic	Eutrophic
Pelagic productivity	**Low**	**High**
Benthic productivity	**High**	**Low**

This study used the record of fossil diatoms to reconstruct the rate and magnitude of human influence in a highly impacted watershed in the Great Lakes region. This chapter regarding a system's biological component is intended to compliment results of the geochemical proxies presented in Chapter 2 which identified phases of human influence. Based on the conclusions of that investigation, it is expected that fossil diatoms will have a similar temporal response, with any significant change in species composition occurring along the same phase boundaries as anthropogenic geochemical proxies. The hypothesis driving this study was that fossil diatoms archived in lacustrine sediment reflect environmental change through changes in species assemblages; and further, that productivity regime shifts that occur in response to anthropogenic activity can be discerned by examining the dominant productivity habitat structure. Fossil diatoms archived in sediment from Muskegon Lake (Michigan, USA), an ecosystem influenced by human activity since the early 1800s, were used to investigate this hypothesis. If true, the composition of diatom species assemblages will reflect shifts in primary productivity dominance that correspond to a known historical timeline of anthropogenic influence in the lake (see Chapter 1). Understanding these relationships is important, particularly as the ecological response to the overprint of warming conditions in the Great Lakes Region further complicates impacts on aquatic ecosystems; and consequently, various feedback mechanisms which remain little understood and highly uncertain (Magnuson et al. 1997). While many studies use diatoms to reconstruct historical human perturbation through nutrient and pH inferences, few use them to understand the processes influencing the behavior of the whole ecosystem (Bennion et al., 2004; Ekdahl et al. 2007; Chen et al. 2008). Though diatom proxies are less quantitative than chemical proxies, (e.g. the variability and range of acceptable conditions for diatoms versus the actual accumulation/concentration of elements); interpreting ecological change with diatom stratigraphy does indicate change in environmental conditions as diatoms also have the potential to integrate hydrological and chemical changes. Therefore, this study contributes to an understanding of long term environmental response trends (pre and post-disturbance), and also addresses the effectiveness of current management strategies aimed to protect water quality in the larger Great Lakes watershed.

3.2 Methods

Productivity regimes in Muskegon Lake were investigated using data and information previously reported, and with additional fossil diatom assemblage data. The history of the study area, sampling methods, and description of sediment core characteristics are (Appendix I) from Chapter 1. Core chronology and sediment geochemistry are detailed in Chapter 2.

Diatom processing

Diatom processing and taxonomical identification was performed by Dr. Nadja Ognjanova-Rumenova at Michigan State University's Algal Ecology Laboratory. Analysis was at 4 cm resolution throughout the core, using standard methods after Battarbee (1986). A total of 38 stratigraphic levels were analyzed from the 149 cm core. Slide preparation began by digesting approximately 0.5 g of homogenized sediment with a strong acid and heating for an hour at 80°C. Digested slurries were rinsed with distilled water until acid-free, and then the siliceous material was mounted on slides. At least 600 valves from each sample were counted and identified using oil immersion objectives at a 1000x magnification. The data were reported as counts per total, and converted to percent relative abundance. Common species were defined as those with at least 2% relative abundance in at least 2 strata levels.

Zone delineation

Ecological zones were determined for the Muskegon Lake core using trends of individual phytoplankton taxa, the composition of overall community structure, and statistical analysis. For the Muskegon Lake core, each common species was first categorized by habitat. Then, the sum of the abundances of all taxa for each habitat was calculated. Notably, the sum for most strata levels does not reach 100%, due to the proportion of "uncommon" taxa which are not accounted for in the calculation. Using these methods, the transitions from benthic to planktonic dominated primary production were clearly established by identifying the percent change in dominant habitat, or by identifying significant change in the trajectories of habitat. Also, where applicable, the distinct preferences of certain taxa (e.g. *Diatoma tenuis* relationship to chloride) were used to delineate zones. To capture less distinct changes in diatom community structure and verify identified zones, a stratigraphically constrained cluster analysis by incremental sum of squares (CONISS) using the program ZONE Version 1.2 was performed on common diatom taxa data (Juggins 1992).

3.3 Results

Diatom abundances

The algal community in the Muskegon Lake sediment core showed high variability in both species composition and relative abundance. A total of 177 taxa were identified in the 38 strata levels sampled from the Muskegon Lake sediment core. Appendix IV lists the 177 diatom species, with authorities, for all taxa present in the Muskegon Lake core. Of the 177 taxa, 27 taxa were considered common, defined as species with at least 2% relative abundance in at least 2 strata levels; these are listed in Appendix V. Results of diatom stratigraphy are reported below.

Throughout the core, both *Aulacoseira ambigua* (Grunow) Simonsen and *Aulacoseira granulata* (Ehrenberg) Simonsen were dominant, ranging from 5.7 - 60.0% and 5.1 - 57.3% minimum and maximum relative abundance, respectively. These species are typical of large, open water deep lakes; therefore, it is expected they would dominate the diatom community (Chen et al. 2008). *Aulacoseira ambigua* is further associated with turbulent water, fitting the flow-through conditions of Muskegon Lake (Wolin and Stoermer 2005). *Aulacoseira ambigua* varies throughout the core, with peaks at 148, 132, 104 and 32 cm depths. *Aulacoseira granulata* is most prevalent in the bottom two-thirds of the core, with peaks between 121 and 128 cm depth at nearly 57%; then, this taxa shows a decreasing trend toward the surface, with <10% relative abundance in top two samples. Both species share the same classifications as ecological indicators, including eutrophic conditions, moderate (~50%) oxygen saturation, and alkaliphilous pH (occurring mainly >7), detailed by van Dam et al. (2004). The relative abundances of these two taxa vary in the Muskegon Lake core, however, because of their similarities with water quality requirements and apparent inverse relationship throughout the core, they are considered to indicate the overall ecological environment (e.g. large, flow through, high nutrient) of Muskegon Lake. Also notable is that both taxa are at their lowest abundances in the top 8cm of the core (<20%).

Additional taxa comprising >20% abundance in at least one level include *Cavinula scutelloides* (W Smith), *Fragilaria capucina* Desmazieres, and *Stephanodiscus minutulus* (Kütz) and are shown in Figure 3.2. *Cavinula scutelloides* is particularly notable because it is associated with cooler climate conditions (N. Ognianova, personal communication 2007). Furthermore, this species is abundant in the bottom (oldest) section of the core, likely associated with the end of the Little Ice Age (1850 AD), before significant changes in ice phonologies (Magnuson et al. 1997). The abundance of *C. scutelloides* abruptly increases in relative abundance to 20%, then decreases at the 112 cm level and fluctuates between 2.5% and the peak between 96 and 148 cm. From the 88cm depth to the top of the core, *Cavinula scutelloides* is much less abundant, with typically only 0.25% abundance or less. *Fragilaria capucina*, and *S. minutulus* have peak abundances in the upper and mid sections of the core, respectively. *Fragilaria capucina* is a small, tychoplanktonic (present in littoral and other benthic habitats), requires high light penetration (indicating shallow water) that is typically indicative of higher nutrient conditions (Ramstack et al. 2003; Panizzo et al. 2008). *Fragilaria capucina* first appears at 108 cm in very low numbers, until reaching 3% abundance at 64 cm; then it increases significantly from 4 to 23% between 12 and 8 cm depth. The peak for *F. capucina* is 30% at 4 cm, while the top sample (2 cm) decreases by half to ~15%. *Stephanodiscus minutulus* is also a planktonic species, associated with hypereutrophic conditions (van Dam et al. 1994). This taxa is not present in the bottom of the core, but shows an abrupt maximum abundance of 24% at 64 cm (it is1.6% or less before that). *Stephanodiscus minutulus* increases again between 44 and 52 cm, and at 28 cm, but then decreases significantly to the surface.

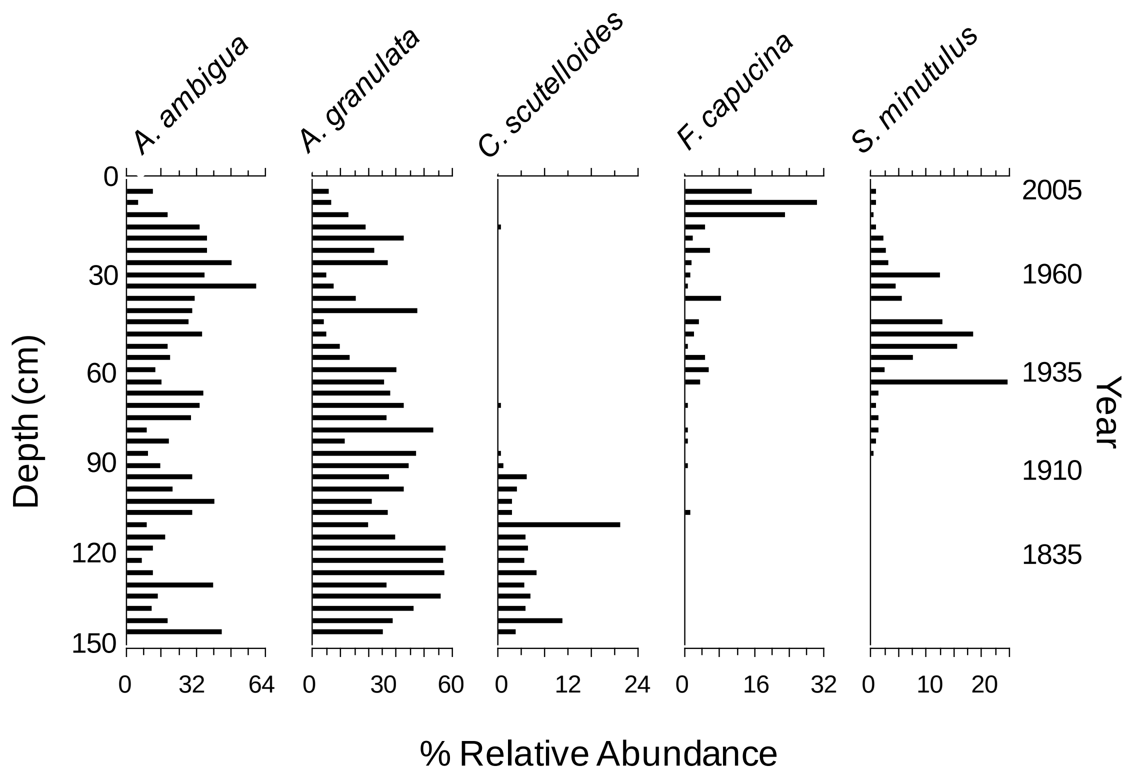

Figure 3.2: Muskegon Lake diatom taxa which are at least 20% abundant in one level.

The next group consists of those taxa that were at least 10% abundant in one sample or more. These include *Aulacoseira granulata fo curvata*, *Cocconeis placentula* Ehrenberg, *Pseudostaurosira brevistriata* (Grun in VH) Williams & Round, *Stephanodiscus hantzschii* Grun in Figure 3.3. *Aulacoseira granulata fo curvata* has a distinct presence throughout the core, and is most common between 36-76 cm. This planktonic species shares an ecological indicator preference similar to other *Aulacoseira* species (van Dam 1994). *Cocconeis placentula* is considered an epiphytic, alkaliphilous, cosmopolitan diatom, with maximum development in nutrient rich and very well oxygenated waters (Ehrlich 1973; Foged 1980; Steinberg and Schiefele 1988; van Dam et al. 1994; Bennion and Appleby 1999). The abundance of this taxon in the Muskegon Lake core is highest between 81 and 88 cm, with a maximum of 19.6% relative abundance at 84 cm; and increases toward the surface to 2-4%. *Pseudostaurosira brevistriata* is classified a benthic species (Kenney et al. 2002; Taylor et al. 2006). This species is present throughout the core, and fluctuates between 1 and 3% abundance in the bottom third (>96 cm), then increases in significance through the upper 12 cm, peaking in the top sample at 14.5%. *Pseudostaurosira brevistriata* is inverse to the trend of *S. hantzschii* which significantly decreases in abundance in the top 16 cm to <1%. *S. hantzschii* is small, centric, planktonic taxa, present in highly eutrophic conditions (Bennion and Appleby 1999; Taylor et al. 2006). Further, *S. hantzschii* has been shown to be resistant to human impacts (Witkowski et at. 2004). Overall, this group shows more distinct trends versus the cosmopolitan taxa of the >20% abundant group, and can provide better indications of ecological conditions.

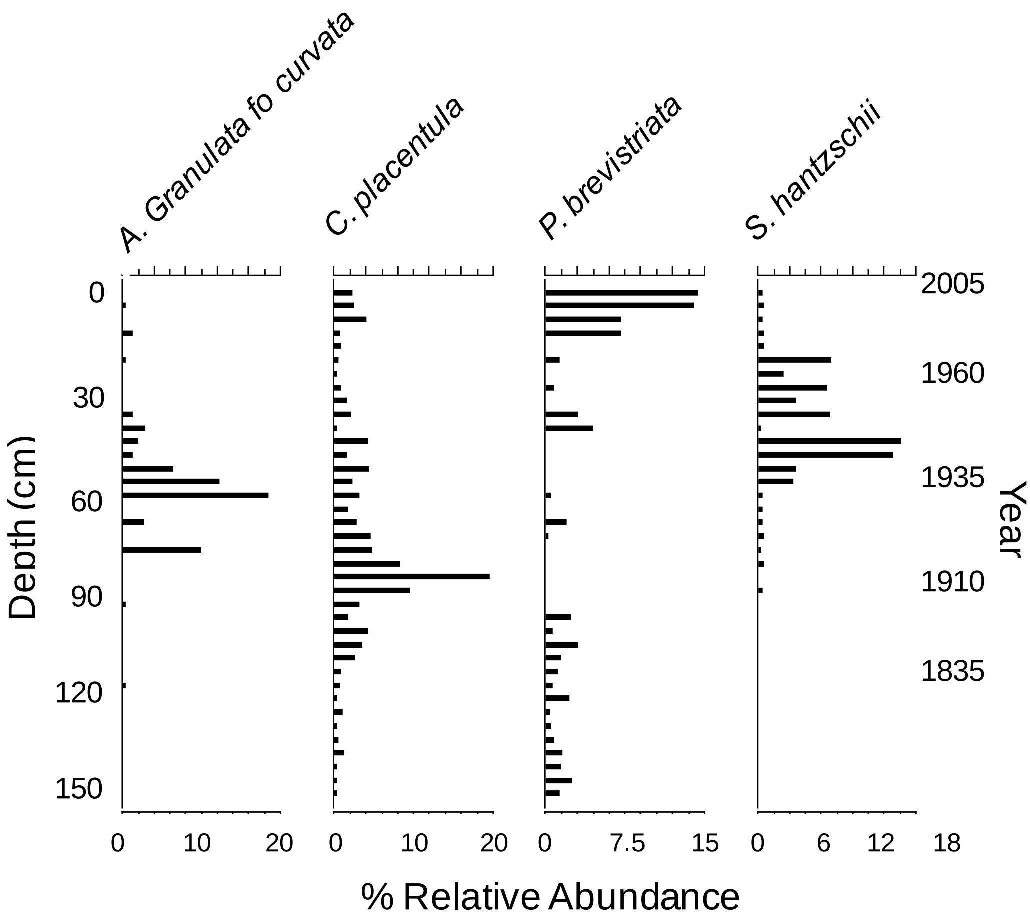

Figure 3.3: Muskegon Lake diatom taxa which are at least 10% abundant in one level.

Diatom taxa with at least 7% abundance are shown in Figure 3.4 and include *Amphora pediculus* (Kützing) Grun, *Cocconeis placentula var euglypta* (Ehrenberg) Cleve, *Fragilaria capucina var gracilis* (Østr) Hust, *Fragilaria crotonensis* Kitton, *Opephora martyi* Hérib, *Staurosira construens* (Ehr) Williams & Round. *Amphora pediculus* is a benthic epiphyte species classified as eutrophic and alkaliphilous (van Dam et al. 1994; Bennion and Appleby 1999; Taylor et al. 2006; Chen et al. 2008). *Amphora pediculus* is present in nearly all samples of this core, and peaks at ~10% at 92 cm depth, but shows an increasing trend in the top 12 cm as well. *Cocconeis placentula var euglypta* is most common in the top two-thirds of the Muskegon Lake core; this benthic taxon peak abundance is at 84 cm, and is also increasing in the top 8 cm. *Fragilaria capucina var gracilis* is a small, tychoplanktonic (adaptive to thermal changes) taxa present mainly in the top sample (2 cm depth) (Panizzo et al. 2008). Van Dam et al. (1994) classifies *F. capucina var gracilis* as associated with oligio-mesotrophic conditions in Netherland lakes, and corresponds to decreasing P concentrations in the water column of Muskegon Lake (Steinman et al. 2008). *Fragilaria crotonensis* is another small, tychoplanktonic taxa which has been observed to respond to changing nutrient conditions and rising nitrate conditions (van Dam et al. 1994; Wolin and Stoermer 2005; Panizzo et al. 2008). *Fragilaria crotonensis* is classified as mesotrophic by van Dam et al. (1994). The presence of this taxon in the Muskegon Lake core is mainly in the top half, with up to 10.4% abundance at 52 cm depth, yet it decreases to 0% in the top sample. *Opephora martyi* is a benthic species mainly present in the bottom half of the core, with maximum abundance of 9.8% at 144 cm depth. *Staurosira construens* varies throughout the core; with the most dominant appearance in the top two samples (2 and 4 cm). *Staurosira construens* is a competitive meso-eutrophic planktonic species documented as growing well in low nutrient arctic lakes (Cremer et al. 2001). Notably, this species is absent from depths 21-48 cm, which is a phase of high P concentrations in Muskegon Lake's water column.

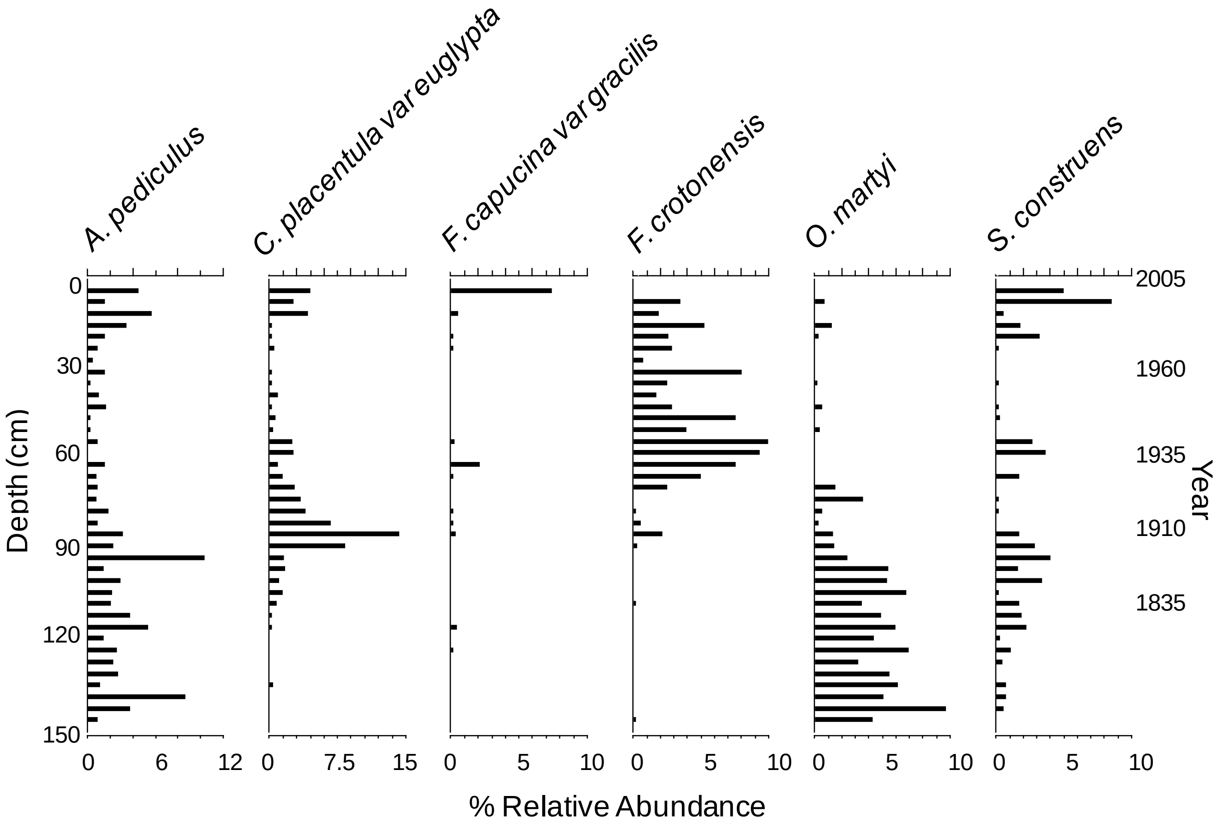

Figure 3.4: Muskegon Lake diatom taxa which are at least 7% abundant in one level.

Taxa of 5% abundance consist of *Achnanthidium minutissimum* (Kützing) Czarnecki, *Asterionella formosa* Hassal, *Aulacoseira granulata var angustissima* (O Müller) Simonsen, *Cocconeis neodiminuta* Krammer, *Cocconeis neothumensis* Krammer, *Stephanodiscus niagarae* Ehr., shown in Figure 3.5. *Achnanthidium minutissimum* is a benthic (epiphytic) generalist species present in circumneutral pH and oligio-eutrophic conditions (van Dam et al. 1994; Anderson 1997; Chen et al. 2008; Panizzo et al. 2008). *Achnanthidium minutissimum* is present in most of the core, but in very low overall relative abundances; however, this species becomes more abundant in the top 8cm of the core, reaching a peak abundance of ~5% in the very top sample. *Asterionella formosa* is present from 24 to 64 cm depth, with a peak of 5.6% at 52 cm depth. *Asterionella*

Formosa is an open water planktonic species sensitive to light changes, documented to respond to changing nutrient conditions, and found to be seasonally dominant in eutrophic lakes (Leitao and Leglize, 2000; Wolin and Stoermer 2005; Chen et al. 2008; Vanormelignen et al. 2008). The presence of this taxon corresponds to a period of significant human influence in the study area. *Aulacoseira granulata var angustissima* has one sample above 5% at 81 cm depth and is otherwise intermittent throughout the core. *Aulacoseira granulata var angustissima* is similar to other *Aulacoseira* spp. and indicative of eutrophic, alkaliphilic conditions (van Dam et al. 1994). Both *Cocconeis* spp. are benthic and show highest abundance in the bottom third of the core. They also both peak at 112 cm, then decrease substantially until the very top of the core again. *Stephanodiscus niagarae* is a planktonic species present throughout the Muskegon Lake core. The maximum abundance for *Stephanodiscus niagarae* is 5.4% at depth 108 cm. Overall, this group of algae (the 5% or more abundant) show distinct variability in both preferred habitat, and temporal presence helping to better define environmental conditions.

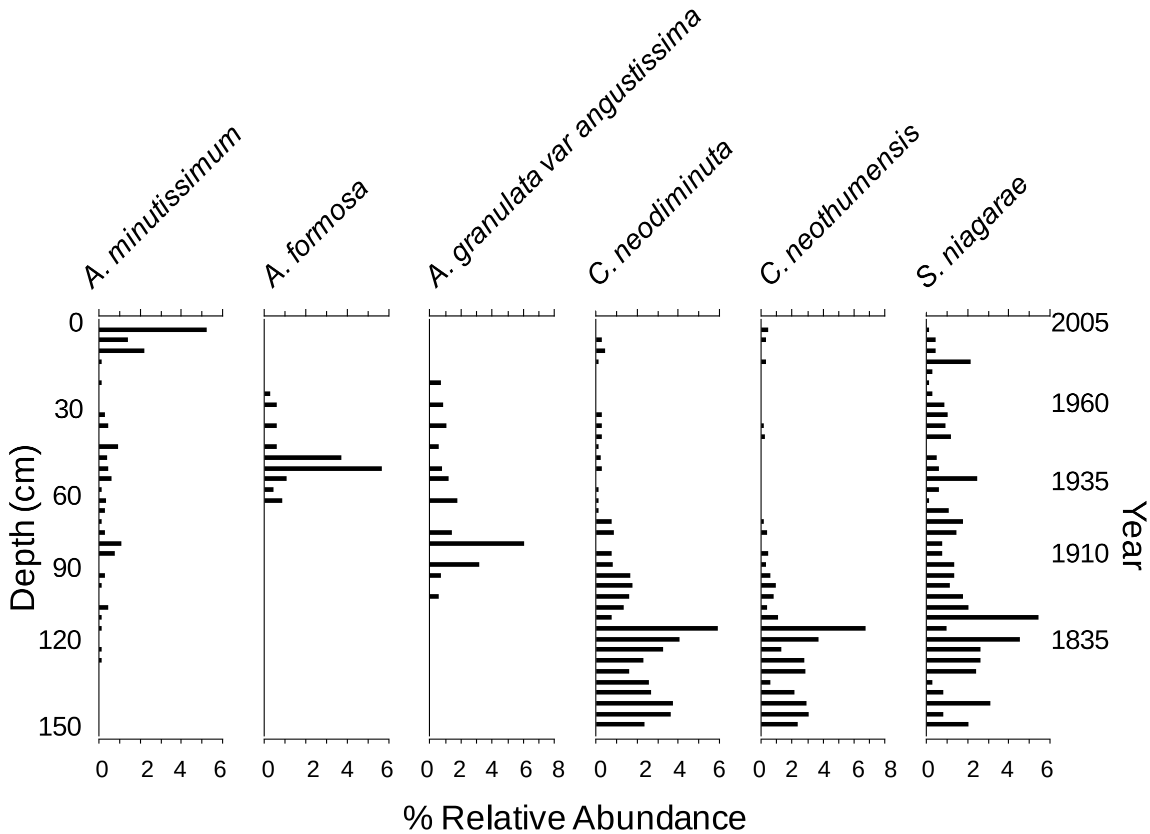

Figure 3.5: Muskegon Lake diatom taxa which are at least 5% abundant in one level.

The least abundant of common taxa, those consisting of >2% in the Muskegon Lake core are shown in Figure 3.6. This group includes *Actinocyclus normanii fo subsalsus* (Juhlin-Dannfelt) Hustedt, *Aulacoseira subartica* (O Müller) Haworth, *Cocconeis pediculus* Ehrenberg, *Diatoma tenuis* Ag, *Planothidium frequentissimum* (Lange-Bertalot) Lange-Bertalot, *Staurosirella pinnata* (Ehr) Williams & Round. *Actinocyclus normanii fo subsalsus* is a large, centric, planktonic species which has been associated with human disturbance, including eutrophication in modern European rivers, but has also been found in much older riverine sediment (van Dam et al. 1994; Leitao and Leglize 2000; Witkowski et al. 2004). In the Muskegon core, *A. normanii fo subsalsus* is present from 48 cm and above, with a peak at 44 cm. This species is not present in the top of the core. *Aulacoseira subartica* is a planktonic species with intermittent presence through most of the core, except the top and bottom samples, with a maximum of 2.7% at 42 and 72 cm depths (Taylor et al. 2006). It is again observed that this taxon is represented in a zone of significant human disturbance which agrees with its association with oligio-mesotrophic conditions (van Dam et al. 1994) and tolerance to increasing nutrient conditions (Dokulil and Teubner 2005; and Wolin and Stoermer 2005). Next is *C. pediculus*, another hypereutrophic, alkaliphic and benthic taxa (van Dam et al. 1994; Chen et al. 2008). *Cocconeis pediculus* has two zones of peak abundance, one between 68 and 88 cm, and another between 4 and 12 cm; and additional, though small, presence through much of the top two-thirds of the core. *Diatoma tenuis* has a distinct presence in the core in that it is more clustered in the top half of the core; the peak for *D. tenuis* is ~6% at depth 28 cm. Then, this hypereutrophic, alkaliphic benthic species decreases to less than 1% from 16 cm to the top (van Dam et al. 1994; Chen et al. 2008). *Planothidium frequentissimum* and *S. pinnata* are both present through most strata levels in the core. *Planothidium frequentissimum* is a benthic species, though not much has been published on it; while *S. pinnata* is benthic (epiphytic) and alkaliphic (Taylor et al. 2006; Chen et al. 2008).

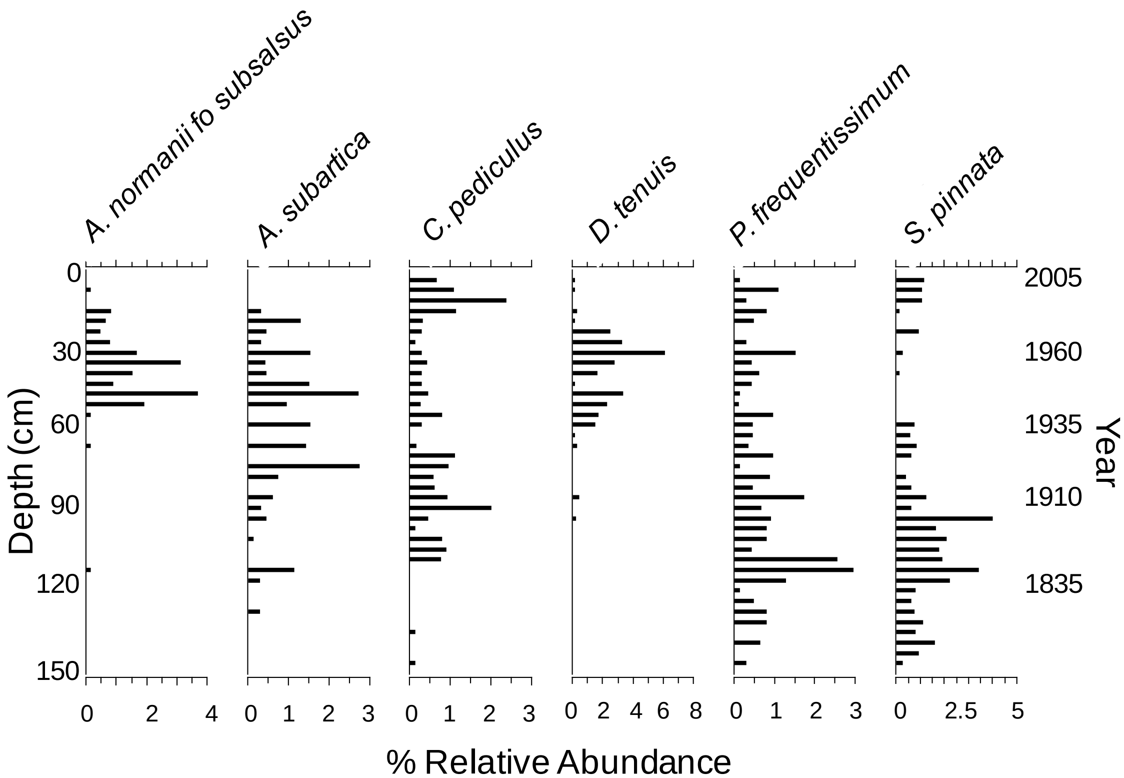

Figure 3.6: Muskegon Lake diatom taxa which are at least 2% abundant in one level.

Though all species in the Muskegon Lake core contribute to an understanding of the evolving ecological status of the lake, some are better than others at lending insight to changing environmental conditions. As reported above, there is a general consensus among many taxa regarding the dominant conditions of Muskegon Lake (e.g. *A. granulata and A. ambigua*). Moreover, a majority of common species in Muskegon Lake prefer alkaliphilic conditions, and are classified as meso-eutrophic to eutrophic. Results shown above also demonstrate that the diatom community assemblage in Muskegon Lake responded to environmental change due to human influence. For the purpose of understanding the collective response of diatoms to stressors, the remainder of this section will focus on deciphering differences in the diatom community structure.

Specific attention will be given to the dominant habitat of taxa to better identify system dynamics, and shifts in primary productivity regimes.

Dominant habitat of diatom paleoproductivity

Diatom taxa from the Muskegon Lake core were classified by habitat (benthic or planktonic) to identify primary productivity regimes. Appendix VI shows a list of the common taxa, with habitat and ecological preferences. Figure 3.7 illustrates stratigraphic habitat preferences for the common taxa, with a marked change shown for benthic and planktonic habitats, over the period of time represented by the core. The bottom of the core reflects between 56 and 77% as planktonic dominated taxa. The sum of planktonic diatom abundance peaks at 88% abundant in the 24 cm depth sample, then decrease to 26- 37% in the top 8 cm of segment from ~8 cm to the top is significant because it reflects a shift in the dominant habitat of diatoms, with the benthic taxa becoming 54 – 62% abundant.

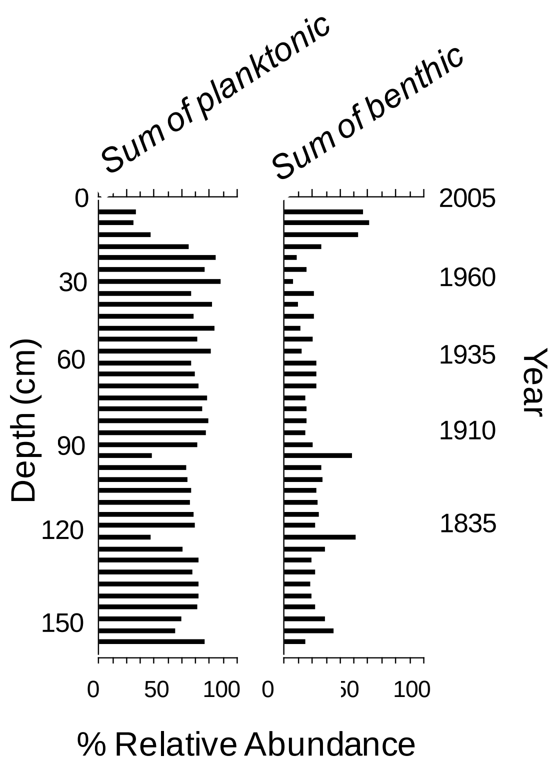

Figure 3.7: The sum of dominant diatom productivity habitats in Muskegon Lake

Diatom zones

The diatom stratigraphy of Muskegon Lake was divided into three major zones according to habitat, ecological preferences of individual taxa, and results of the Zone analysis. Figure 3.8a (benthic) and 3.8b (planktonic) shows zone specific relative abundance of all common Muskegon Lake diatoms. Delineating zones distinguished pre-disturbance and post-disturbance phases, as well as the post-regulation ecosystem state (e.g. Clean Air and Clean Water Acts). Importantly, this method clarified the response of diatoms to multiple stressors, and delineated *phases* of anthropogenic influence.

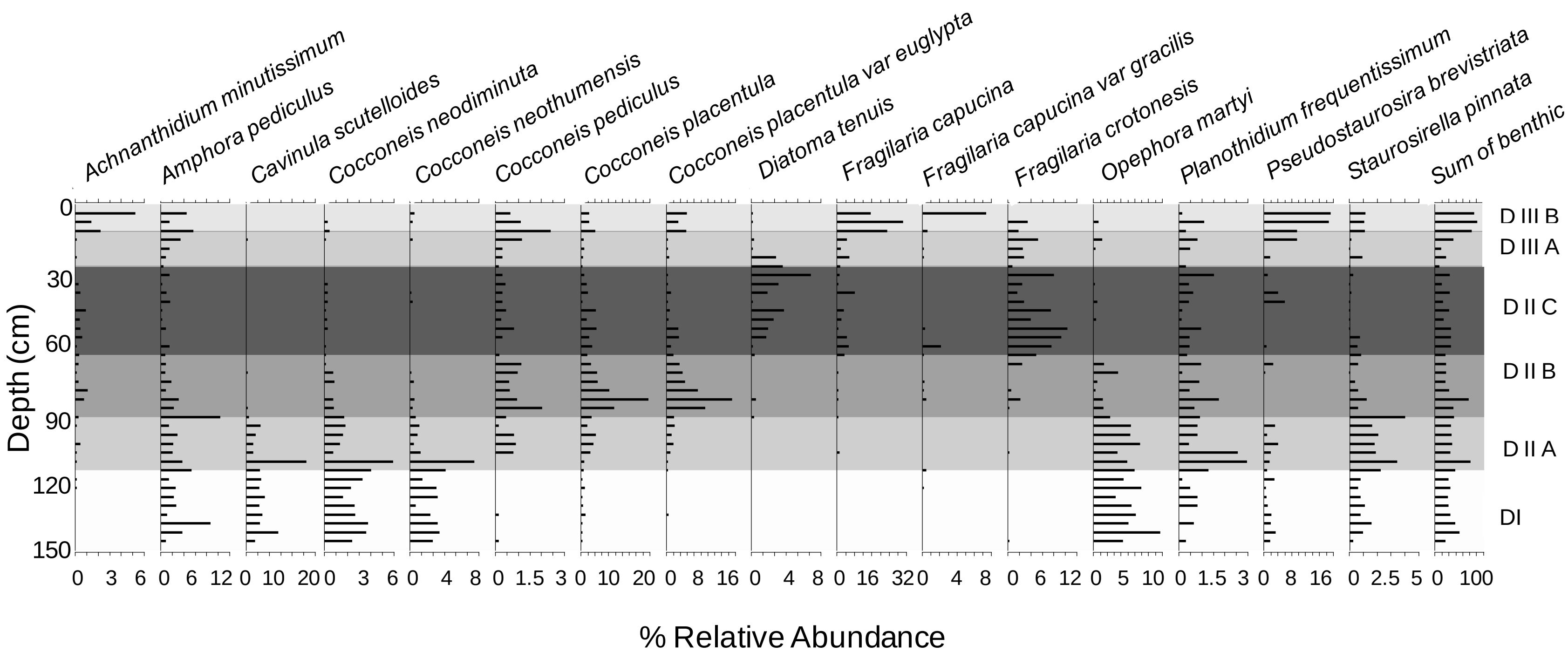

Figure 3.8a: Fossil diatoms from Muskegon Lake associated with benthic preferences.

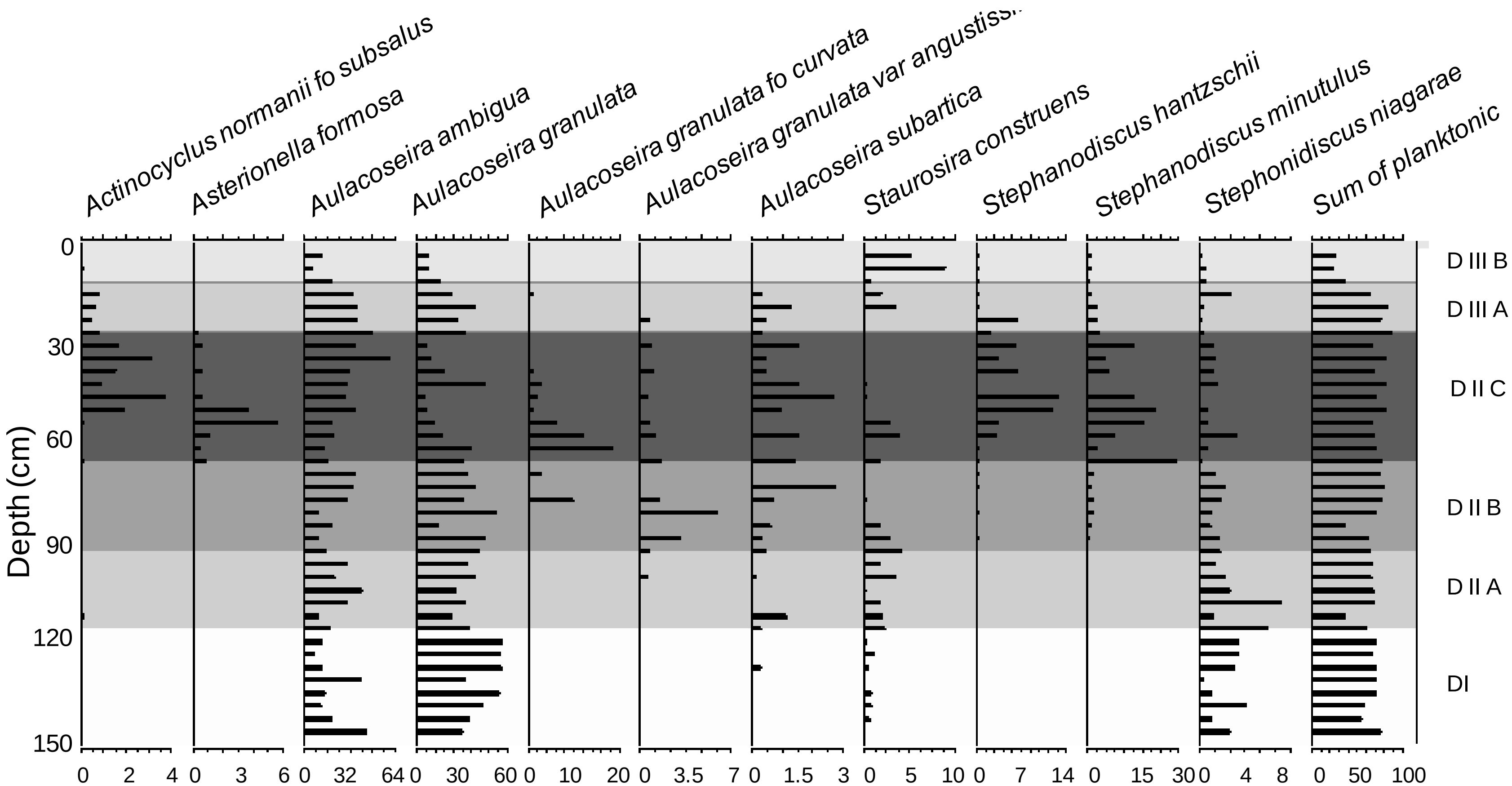

Figure 3.8b: Fossil diatoms from Muskegon Lake associated with planktonic preferences.

Zone DI: Pre-disturbance (149-116cm depth)

The bottom of the Muskegon Lake core reflects conditions prior to significant anthropogenic disturbance. Though there are still noted changes in the overall diatom community structure, the dominant assemblages at the bottom of the core were *Aulacoseira ambigua* and *A. granulata,* who show interchangeable dominance in the bottom zone, fluctuating between 54 and 75% of the total abundance. Additional taxonomic changes include a peak of *Amphora pediculus* at 139 cm depth, and both *Cavinula scutelloides* and *Opephora martyi* at depth 144 cm depth. Even though these taxanomic changes were observed, the diatom community structure is stable for the most part, and the bottom of the core is dominated by planktonic algal production (from ~60 to 77% from bottom to 112 cm depth). The 116 cm depth, where benthic productivity becomes dominant, marks the boundary between zone I and zone II. Though this switch from planktonic to benthic is just for one level, it marks the first significant change in overall composition of diatoms in the core.

Zone DII: Disturbance (116 - 21cm depth)

The second zone delineated in the Muskegon Lake core is that of anthropogenic influence. This zone reflects shifts in individual taxa abundances, as well transitions in planktonic versus benthic dominated habitats. Zone DII begins at ~116 cm depth (~1840 AD) and end at the 24 cm depth (1973 AD). In this zone, there is biostratigraphical evidence (e.g. appearance and disappearance of indicator taxa) of considerable human impacts from land use changes (e.g. industry, agriculture and urbanization). To make clear the response to human perturbation, this zone is further divided into three sub-zones that correspond to distinct floristic change.

Sub-zone DIIA (116 – 92 cm depth) is the initial disturbance zone revealed in the disturbance zone. As throughout the core, *Aulacoseira granulata* and *A. ambigua* are dominant in this sub-zone. *Stephanodiscus niagarae* increases somewhat in abundance at 108 cm depth (accounting for >5%). *Cavinula scutelloides* has an outlying peak at 21% at 112 cm depth, then returns to the 2-5% range in the younger samples. *Opephora martyi* maintains zone I abundance, while *Staurosirella pinnata* increases slightly. *Cocconeis neodiminuta* and *C. neothumensis* show peaks in this sub-zone that correlate to the *Cavinula scutelloides* peak at 112cm depth. Both then sharply decline to abundances lower than in zone DI. *Cocconeis pediculus*, *C. placentula,* and *C. placentula var euglypta* all slightly increase. In addition, *Amphora pedediculus* declines in this zone, and *Staurosirella Pinnata* have higher abundances at the upper and lower boundary of DIIA; *Cocconeis pediculus, C. placentula* and *C. placentula v. euglypta* increase, so define this zone with first significant presence (up to 5%).

The next sub-zone (DIIB) in the Muskegon Lake core is from 92 – 64 cm depth. In this sub-zone, the Muskegon Lake's cosmopolitan species, *Aulacoseira granulata* and *A. ambigua* remain dominant, though there are changes for several taxa that are good ecological indicators. The benthic and planktonic taxa percentages are relatively stable, with the exception of the 84cm depth where benthic taxa comprise 49% of the total and become dominant for one level. However, despite the stability, specific taxa are observed to be responding to the environmental change. For example, the other *Aulacoseria* spp. show high abundances in this sub-zone as *Aulacoseira granulate* fo *curvata* and *Aulacoseira granulata var angustissima* have increases. Also, *Cocconeis placentula* and *C. placentula var euglypta* both peak in this sub-zone to nearly 20 and 15%, respectively, that together comprise nearly 35% of the total at 84 cm depth. *Stephanodiscus minutulus* is dominant in this zone, with 24% abundance at 64 cm depth; marking the boundary with sub-zone DIIC. Additionally, the first noteworthy appearance of *Fragilaria capucina*, *Fragilaria capucina var gracilis* and *Fragilaria crotonensis* occurs at the boundary, illustrating an important compositional change which becomes more significant in the next zones.

The youngest of the disturbance sub-zones is DIIC, delineated between 64 – 24 cm depth. In this disturbance sub-zone, planktonic diatom taxa *Actinocyclus normanii fo subsalsus*, *Asterionella formosa*, *Stephanodiscus hantzschii*, and *S. minutulus* peak in abundance. Two common benthic species also become more abundant in this sub-zone as *Diatoma tenuis* and *Fragilaria crotonensis* both reach sustained high relative abundances at these depths (accounting for 2-7% and 3-10%, respectively). Conversely, benthic taxa *Amphora pediculus*, *Cocconeis pediculus*, *C. placentula* decline in this zone to their lowest abundances; while *Cavinula scutelloides*, *Cocconeis neodimiminuta*, *C. neothumensis* and *Opephora martyi* are very low or zero abundance in this zone.

Zone DIII: Mitigation

Beginning at the 24 cm depth, the Muskegon Lake core reflects a shift from planktonic to benthic dominated diatom habitat, with a separate, stronger shift from 8cm to the top. This zone lends insight to ecological response following a mitigation strategy; the installation of waste water treatment plant that diverted municipal and industrial waste away from Muskegon Lake. Consequently, the diatom community structure reflects a change in composition in this zone.

Several taxonomic changes occur in the first mitigationsub-zone, DIIIA. For example, *Diatoma tenuis* is observed to decline from ~3% total abundance, to nearly zero. The environmentally resistant taxa *Stephanodiscus hantzschii* and *S. minutulus*, also decline toward the surface; while *Asterionella formosa* disappears entirely for the remainder of the core. Notably, the core's most dominant planktonic species, *Aulacoseira ambigua* and *A. granulata*, both display a decreasing trend in abundance to the surface, beginning at the 24 cm depth, then more significantly in the top 8 cm, accounting for 12.6 and 7.6%, respectively of the total abundance in the most modern sample. Taxa that increase in this sub-zone include the benthic *Fragilaria crotonensis* and *Pseudostaurosira brevistriata*.

Further floristic change persists in the top 8 cm (sub-zone DIIIB). Benthic taxa become dominant in this subzone, and maintain that dominance to the modern sample (at 2 cm depth). The benthic *Achnanthidium minutissimum*, *Amphora pediculus*, *Cocconeis pediculus*, *C. placentula*, *C. placentula var euglypta*, *Fragilaria capucina*, *F. capucina var gracilis*, *Pseudostaurosira brevistriata* and *Staurosirella pinnata* all increase in relative abundance in this sub-zone. *Staurosira construens* is the sole planktonic taxon to experience an increase in abundance in this sub-zone, and accounts for 8.6 and 5% in the 8 cm and 4 cm depth samples, respectively.

Dominant diatom habitats

A summary of results for benthic versus planktonic primary productivity are shown in Figure 3.7. Described by zones, it is apparent that zone DIIIB reflects a shift in the dominant habitat from planktonic to benthic. The relative percentage of benthic taxa in this zone peaks at 61.5% at the 4 cm depth, compared to a 23% average in zone DI. Also notable are levels 112 and 84 cm, where the benthic habitat has significant increases.

3.4 Discussion

The fossil diatom based interpretation of the ecological state of Muskegon Lake suggests that there have been significant environmental change since major human settlement in the late 1800s. Diatom stratigraphy from this core reveals abrupt shifts in community structure, and paleoproductivity habitats indicate altered nutrient dynamics. Feedback mechanisms driving ecological dynamics can be indentified in these shifts, based on the delineation of zones, and coupled with the documented historical watershed activity. Using the three major zones identified by fossil diatom assemblages, it is possible to correlate ecological dynamics to phases of human influence in the watershed.

Phase I

The first phase represented by the core is that of pre-disturbance, corresponding to zone DI described in the results. *Aulacoseira ambigua* and *A. granulata* are dominant in this phase. According to the classification by van Dam et al. (1994), the planktonic taxa present are associated with mesotrophic to meso-eutrophic conditions, and likely indicative of Muskegon Lake's flow-through hydrology which inputs ample nutrients from the vast Muskegon River Watershed. *Aulacoseira ambigua* is present in this zone and has a relatively high Si requirement for growth (Kilham; Kilham and Hecky, 1986). Further, *A. ambigua* was found in nearby Lower Herring Lake (MI) and reported to be prevalent in turbulent conditions, as they require mixing conditions to remain suspended in water column (Wolin and Stoermer 2005). This requirement may explain its pre-disturbance dominance and peaks of *A. ambigua* at the 148 and 132 cm depths in that it may reflect periods of flood events, or otherwise high flow from the Muskegon River Watershed.

Phase 2

Phase two of the Muskegon Lake core is characterized by the initial period of human disturbance, which is also reflected in the fossil diatom assemblages. The initial stage of this phase (corresponding to sub-zone DIIA) commenced in the 1840s and corresponds to large-scale logging in the Muskegon River watershed (Alexander 2006). This activity led to pronounced effects in the composition of the algae population, perhaps initially due to increased erosion that increased turbidity in water column, and perhaps due to increased nutrients released with erosion (Fritz et al. 1993). Because *Aulacoseira granulata* and *A. ambigua* maintain overall dominance in this sub-zone, algae indicate no broad change in dominant productivity habitat. *Aulacoseira ambigua* was reported by Bradbury et al. (2002) to be present after deforestation. In Muskegon Lake, this species is at least 8% abundant in nearly all levels and does not appear to specifically indicate a deforestation signal. However, the peak, then quick decline of *Cavinula scutelloides*, reported to be a cooler water taxa, may signify a change in the thermal regime in the watershed. The *C. scutelloides* decline may be the result of water temperature increases due to changes in climate following the end of the Little Ice Age. Or, the data could indicate that the removal (from logging) of shade trees along the shores of the Muskegon River and its tributaries warmed the inflow waters enough to have influenced the decline of *C. scutelloides*. The peaks of *Cocconeis neodiminuta* and *C. neothumensis* at 112 cm depth correlate to the *Cavinula scutelloides* peak; then, they also decrease to abundances lower than in predisturbance phase (zone DI). The coupling of these three taxa suggests a change in environmental conditions, and because they are all benthic taxa, it may suggest an increase in turbidity from erosion or logging material (saw dust) which blocked light from the benthic habitat, thus decreasing their abundances. *Cocconeis pediculus, C. placentula,* and *C. placentula var euglypta* all slightly increase in zone DIIA. The increase with these taxa may indicate the first human induced contributions of increased nutrients, either from terrestrial erosion, or increased population, as these taxa respond to such conditions (Steinberg and Schiefele 1988). As the logging era ended due to depleted lumber resources (~1905 AD) the next phase of Muskegon Lake began as deforested land was converted for agriculture, industry and urbanization (Tang et al. 2005; Alexander 2006; Ray and Pijonowski 2010).

Sub-phase IIB in the Muskegon Lake core (corresponding to sub-zone DIIB) marks the next wave of anthropogenic influence, namely due to industrialization. Muskegon Lake has a history of industrial activity on its shoreline and surrounding city (see Steinman et al. 2008 and Ray and Pijanowski 2010). By the mid-1900s, industrial activity along the lake's shoreline included foundries, metal finishing plants, a paper mill, and petrochemical storage facilities that influence ecological conditions. Through the advent of industrial pollution, Muskegon Lake's cosmopolitan species, *Aulacoseira granulata* and *A. ambigua*, remain dominant. The benthic and planktonic taxa ratios are relatively stable, with the exception of the 84 cm depth where benthic taxa become dominant for one level.

Despite this stability, the abundances of specific taxa seemingly respond to the environmental change. *Aulacoseria* spp. show high relative abundances in this sub-phase as *Aulacoseira granulata* fo *curvata* and *Aulacoseira granulata* var *angustissima* increase. Also, *Cocconeis placentula* and *Cocconeis placentula* var *euglypta* both peak in this sub-zone to nearly 20 and 15%, respectively, together comprising nearly 35% of the total at 84 cm depth. These taxa are indicative of eutrophic conditions, and as noted above, have maximum growth in nutrient rich and well oxygenated waters (Steinberg and Schiefele 1988; van Dam et al. 1994). *Stephanodiscus minutulus*, a planktonic eutrophic species, also gains dominance in this zone, with a 24% abundance at the 64 cm depth, marking the boundary with sub-zone DIIC. Other high nutrient indicator taxa that increase toward the sub-zone boundary (e.g. *Fragilaria capucina*, *F. capucina var gracilis*, and *F. crotonensis*) further illustrate the influence of increasing nutrient inputs that are the focus of sub-zone DIIC.

The youngest portion of the disturbance phase corresponds to sub-zone DIIC, which seemingly reflects the floristic response to excessive nutrient inputs (see Freedman et al. 1979; Steinman et al. 2008); presumably the result of population growth, advent of widespread fertilizer use, and direct inputs of municipal and industrial waste water. From 61- 24 cm depth, planktonic diatom taxa indicative of highly eutrophic conditions gain prevalence in the fossil diatom assemblage (though attempted diatom inferred total phosphorus reconstructions were inconclusive), similar to qualitative interpretations by

Fritz et al. 1993. Taxa in this sub-phase include *Actinocyclus normanii* fo *subsalsus*, *Asterionella formosa*, *Stephanodiscus hantzschii*, and *S. minutulus*; which all reach their maximum abundances in this sub-zone and who all have been documented as being associated with human disturbance and changing nutrient conditions (Bennion and Appleby 1999; Leitao and Leglize 2000; Witkowski et al. 2004; Wolin and Stoermer 2005; Taylor et al. 2006; Chen et al. 2008). Moreover, two common benthic species abundant in this sub-zone, *Diatoma tenuis* and *Fragilaria crotonensis*, both reach sustained high relative abundances at these depths. These taxa, because of their distinct presence in the core and well-defined ecological preferences specifically noted in other regional investigations (e.g. Ramstack et al. 2003; Wolin and Stoermer 2005), they are indicative of further changing nutrient conditions and inputs, as compared to zone DIIB (Bradbury 1975; Brugam 1979; Wolin and Stoermer 2005). In particular, *Diatoma tenuis* is common in culturally eutrophied areas of the Great Lakes region (Wolin and Stoermer 2005), and a study of water quality trends in Minnesota published by Ramstack et al. (2003) inferred a chloride optimum of 81.8 meq l^{-1} for the taxon *D. tenuis*. Moreover, the Wolin and Stoermer (2005) investigation supported these findings, as this taxon was found to correlate to a "high conservative ion loading". Wolin and Stoermer (2005) also report that *Fragilaria crotonensis* is associated with high nitrate concentrations, which is consistent with the reported increases of nitrate in Muskegon Lake during this time period (see Chapter 1) (Freedman et al. 1979; Steinman et al. 2008). Conversely, benthic taxa *Amphora pediculus*, *Cocconeis pediculus* and *C. placentula* decline in this zone to their lowest abundances, perhaps the result of abundant planktonic taxa blocking light and nutrients from benthic habitats. (Another caveat to the data worth noting is that seasonal productivity cannot be determined when using abundance data at this resolution.) Also, noticeable increases in the small *Stephanodiscus* spp., which reportedly have high phosphorus optima, occur in this sub-phase of the Muskegon Lake core (Bradbury 1975; Fritz et al. 1993; Bennion et al. 1996). These taxa become largely absent in the next phase, likely due to the competitive advantage of benthic taxa as increased water clarity results from environmental regulations (Steinmann et al. 2008).

Phase 3: Mitigation

The mitigation phase lends insight to a unique event in recent environmental management of the Muskegon River Watershed, and particularly for efforts to remediate Muskegon Lake. In 1973, the County of Muskegon installed a tertiary waste water treatment plant, the Muskegon County Wastewater Management System (MCWMS) to divert municipal and industrial waste water (Alexander 2006; Steinman et al. 2008). The MCWMS replaced smaller individual community systems which were overburdened, and to capture industrial effluent being directly discharged to Muskegon Lake with inadequate treatment (http://www.muskegoncountywastewater.com/aboutus.shtml). Consequently, this zone reflects the ecological response of management actions, designed to mitigate much of the anthropogenic influence of phase II. Using correlating chronology of ^{210}Pb dated short core the MCWMS installation event is dated to ~24 cm depth, and is associated with change in the biostratigraphy of diatoms.

At the 24 cm depth, the Muskegon Lake core shows a shift from planktonic to benthic dominated diatom habitat. The first subzone (DIIIA) of the Mitigation Phase appears to be a transition phase where taxa attempt to‘re-equilibrate’ to decreasing nutrients, metals, and conservative ions (e.g. Cl^-) that were diverted by the 1973 MCWMS installation. A distinct change in this zone is the sharp decline in *Diatoma tenuis*. This is a notable change, as this taxon has been found to be common in eutrophicated lakes of the Great Lakes and indicative of high conservative ion loading (van Dam et al. 1994; Chen et al 2008). Its decline is likely related to the wastewater diversion that reduces chloride from the lake, allowing other taxa to be more competitive, and thus, higher relative abundance. The decline of *Stephanodiscus hantzschii* and *S. minutulus*, and the disappearance of *Asterionella formosa*, all of which are associated with hypertrophic conditions, also reflect the ecological response to the diversion of wastewater (and nutrients) (Bennion and Appleby 1999; Witkowski et al. 2004; Taylor et al. 2006). *Stephanodiscus hantzschii* is notable for its presence in the anthropogenic zone of the core, since it is a species tolerant of human disturbance (Bennion and Appleby 1999; Taylor et al. 2006). The decline of this taxon after the installation of the waste water treatment plant suggests the system responded by allowing less tolerant taxa to become more competitive under improved water quality conditions. Finally, the decline of *Aulacoseira ambigua* and *A. granulata*, play a significant role in the overall decline of planktonic taxa in sub-zone DIIIA; a trend that was most distinct in sub-zone DIIIB.

The modern diatom assemblage in the top 8 cm (sub-zone DIIIB) reflects further floristic change. This sub-zone indicates the continued evolution of the diatom paleoproductivity as it responds to human influences. Benthic taxa become dominant in this subzone, and maintain that dominance to the most modern sample at 2 cm depth, a sustained transition that suggests a shift in productivity habitat. Though this shift should be interpreted with caution as the system could still be in a transitional state, increases in the benthic *Achnanthidium minutissimum, Amphora pediculus*, *Cocconeis pediculus*, *C. placentula*, *C. placentula* var. *euglypta*, *Fragilaria capucina, F. capucina* var. *gracilis*, *Pseudostaurosira brevistriata* and *Staurosirella pinnata* do indicate that the benthic habitat became more competitive compared to the planktonic habitat in this phase. Though, benthic diatoms probably do not compete for resources with planktonic diatoms; however, planktonic productivity may influence the benthic through shading, but no similar reciprocal effect of benthic on planktonic taxa.

Evidence of productivity regime shifts reflected habitat of fossil diatoms

Shifts in benthic and planktonic primary productivity indicate shifts in ecological function, because primary productivity is an important factor in trophic transfer, and thus the flow of energy through the lake system (Vabdeboncoeur et al. 2003). Metal contamination, as observed in Muskegon Lake, can also play a role in habitat shifts, as Cattaneo et al. (2008) reported that under increased contaminant conditions, there was a shift from planktonic to benthic taxa dominance. Cattaneo et al. also suggested that littoral zones were a refuge under high contamination conditions. However, this result is not supported by diatom data from Muskegon Lake.

In assessing the Muskegon Lake diatom community post-management policies, one would expect a community structure similar to pre-perturbation; or at least a community trending in that direction. However, what is observed in the biostratigraphy of Muskegon Lake suggests a *restructuring* of the diatom community. The total shift in habitat dominance from planktonic to benthic in the Mitigation Phase is by a substantial margin, from 88% total planktonic at 24 cm depth, decreasing to 27% at 2 cm depth, indicating a clear shift in primary productivity regimes. Though, because we are interpreting relative abundance data, care must be taken with the interpretations, as the increase in benthic may not be attributed to an actual increase in benthic diatom productivity, but a decrease in relative planktonic diatom abundance.

Referring to Figure 3.9, one can relate the ‘perturbation’ of the MCWMS installation as the event that kicked the system to a new regime (a benthic dominated regime). This is interpreted to indicate that as water clarity increased, the benthic habitat reflected an increase in relative productivity, perhaps aided with the legacy of ample nutrients left over in the sediment (evidenced by sediment P profile). Further supporting the idea of a new productivity regime for Muskegon Lake is that the benthic community has remained dominant for close to two decades.

This leads to questions of how and why has the benthic community dominance been sustained, since it is unlikely that ‘extra’ sediment nutrients would last this long. Furthermore, the historic loss of littoral habitat (see Figure 1.8, chapter 1) should decrease the relative abundance of benthic diatoms; however, their abundance is increasing to surface despite this? An attempt to answer these questions suggests an additional change in the function of the ecosystem that alters feedback mechanisms. It has been reported that invasive zebra mussels entered Muskegon Lake in ~1989 (Jeff Alexander, personal communication, 2008). Zebra mussels further increase water clarity, and their waste adds to nutrients in the benthic habitat, thus creating a positive feedback loop. It is important to note that caution needs to be applied to this idea, since there could be unknown factors contributing to these trends not related to this interpretation. For example, it is possible that increased urbanization and/or non-point sources are contributing to the trajectory of change by adding nutrients to the system/sediment. This possibility will be explored in chapter 4, with a comparison of sediment geochemical productivity proxies.

Influence of management strategies on environmental recovery

The Muskegon Lake record provides the opportunity to explore the response of highly perturbed lake to pollution reduction due to management strategies. While the implied 'recovery' for Muskegon Lake is somewhat unclear, the diatom community structure has contributed to understanding of ecological process. The compositional structure in the pre-disturbance phase reflects a state of system stability. The abrupt changes in diatom assemblage stratigraphy in the Anthropogenic phase (DII) indicate dramatic (possibly irreversible) environmental change. The productivity habitat regime shift in the Mitigation phase indicates a more dramatic change in the ecological function of the lake, suggesting there were unforeseen consequences, or factors driving ecological response to this management strategy (though, this is not necessarily negative, just arguably unexpected). If environmental response is evaluated using the conceptual model in chapter 1 (Figure 1.2), it would indicate that Muskegon Lake is currently at point C2. Moreover, there are additional impacts to the Muskegon Lake ecosystem from human influence that need addressed. These results indicate that ecological assessments and policy recommendations need to consider environmental change over longer spans of time. Looking at systems in this paleoecological method is a beneficial approach to understanding system response and recovery.

Recovery challenges

This study suggests continued challenges to the ecological health of Muskegon Lake. Though the fossil diatom record from this core suggests that efforts to reduce nutrient loading have had some success (supported by change in taxa and record of monitoring data); new sources (e.g. riverine inputs from increasing urbanization and storm water effluent), or continued recycling of phosphorus from sediments may degrade water quality, and be reflected by the future diatom community structure. Additionally, the overprint of warmer climate in the Muskegon River Watershed could have implications for ecological recovery in this system, as has been observed already from the bottom of core to the top with taxa indicating thermal sensitivities (e.g. *Cavinula scutelloides*). Finally, the influence of invasive species (e.g. zebra mussel) could further alter ecological dynamics in the larger Great Lakes Region.

Comparison to regional diatom studies

Diatom results from Muskegon Lake are in many ways distinct from other regional studies. In particular, the presence of *Cavinula scutelloides* is notable because it has not been identified in other regional studies (Fritz et al. 1993; Garrison and Fitzgerald 2005; Wolin and Stoermer 2005; Watchorn et al. 2008). Also, because of the high relative abundance of this taxon in the the lower (reference) part of the core, it is difficult to reconstruct a quantitative ecological history (e.g. total phosphorus). Other taxa not identified in other regional studies include the *Cocconeis* spp.

However, many taxa which were identified and described in regional studies can be used for comparisons to the Muskegon Lake community. For example, the small benthic *Fragilaria capucina* has been found to be associated with floating agal mats (mostly green algae) (Garrison and Fitzgerald 2005). *Staurosira construens* is a planktonic taxon identified by Bradbury and Winter (1976) to be found growing attached to macrophytes in eutrophic conditions; this taxa increases in the top 8 cm of the core.

Importantly, the shift in habitat of primary productivity is a key indicator of ecological dynamics in Muskegon Lake. In a study of Canadian Arctic lakes, the taxonomic shift was towards greater relative abundance in the planktonic community, not from planktonic to benthic as reflected in the Muskegon lake core (Rühland et al. 2003). Whereas in a study of shoreline development and commercial cranberry farming in Wisconsin, USA, Garrison and Fitzgerald (2005) found that the lake experienced an increase in the density of the epiphytic diatom community, driven by increased nutrient inputs.

Limitations

There are limitations to this study that need addressed. First, it is important to note that this study reports relative abundance of fossil diatoms, which is not the same as diatom biomass. Therefore, what is represented here is not the volume of algae present, but the relative abundance of each taxon. It is also important to note that inferences from diatom stratigraphy can be muted, or otherwise convoluted due to varying accumulation rates, chemical dissolution, or bioturbation/re-suspension during the sedimentation process (Engstrom and Swain 1986; Wolin and Stoermer 2005). Errors in chronology are also possible, though have no influence on biological interpretations, only the estimated timing of ecosystem state changes. The relative importance and magnitude of productivity changes are still reflected accurately.

Also, due to the lack of documented ecological preferences with some of the taxa, it is difficult to assign specific optimal conditions for each core level and did not allow for credible diatom inferred reconstructions of total phosphorus or pH, etc. Additionally, many taxa have a range of tolerance which complicates interpretations (Panizzo et al. 2008). Anderson et al. (1993) and Bradshaw and Anderson (2001) caution the ability of diatoms to infer nutrients, as they observed little change in diatom species composition above 100 µg/L (epilimnetic TP). That said, there is adequate variability within the community structure of the Muskegon Lake diatom biostratigraphy to confidently assert a shift in primary productivity habitats.

Interpretations from Wolin and Stoermer (2005) were particularly important to this study, due to the close geographical proximity that adds credibility to the ecological preferences of the diatoms, as so many other studies are outside of North America (e.g. Chen et al. 2008; Taylor et al. 2006; Bennion and Appleby 1999). Moreover, the similarity of species composition in the Wolin and Stoermer (2005) study facilitates a better evaluation of algal response to changing environmental conditions.

3.5 Conclusions

This chapter examined the history of human influence in Muskegon Lake using diatom stratigraphy as an indicator of the biological response of the system. Human impacts have not only changed the relative abundance of individual taxa within

Muskegon Lake, but have restructured the dominant habitats of primary productivity. Floristic changes in the Muskegon Lake core reflect the relative timeline of anthropogenic impacts in the Great Lakes region and support the hypothesis that human influence has changed the diatom community structure, and has shifted the dominant habitat of productivity.

This study demonstrated that Muskegon Lake experienced seemingly irreversible change in the structure of its primary productivity dynamics in the past two centuries due to natural and anthropogenic environmental change. The overall diatom community structure has changed, with distinct shifts in habitat dominance between planktonic and benthic habitats. Importantly, these changes do not yet appear to be stabilized, and may indicate the system is in a state of hysteresis.

The biostratigraphic change in Muskegon Lake diatoms was mainly attributed to various human stressors, based on the timeline of human activity. The diatoms have clearly undergone dramatic change in response to various stressors, including thermal pollution and water chemistry changes, resulting in accelerated eutrophication due to deforestation, agriculture and urbanization of the catchment. Though there was recent recovery in the water quality of the lake (Steinmann et al. 2008), data indicate that the biological community is still changing in response to human impacts and will continue to be influenced by contemporary human activity and legacy effects. This suggests that further management strategies need to target the non-point source nutrient inputs, and emphasize the influence of emerging disturbances (e.g. invasive species and climate) to productivity dynamics.

This chapter further concludes the following:

- The pre-settlement diatom community is markedly different than composition of the modern assemblage, though diatoms display natural variability in the bottom of the core, which could reflect conditions unique to the Muskegon Lake system (large drowned river mouth ecosystem). In phases II (Anthropogenic) and III (Mitigation), biostratigraphic change suggests a relationship to chronology of human activity in the watershed.

- The shifts in benthic versus planktonic dominated primary productivity is dramatic in the top of the core; presumably in response to the installation of the WWTP and invasion of zebra mussels. This suggests a re-structuring of the diatom community to a condition not resembling the pre-perturbation regime.

- These interpretations are predicated on the response of diatom flora alone. As a result, they only indicate how the biological community is behaving under anthropogenic stressors. Integrating diatom data with additional environmental indicators (e.g. geochemistry) can further untangle certain system complexities that will better identify signals of system response.

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CHAPTER 4

INTEGRATING SEDIMENT GEOCHEMICAL CHRONOLOGIES AND FOSSIL DIATOM STRATIGRAPHY TO VALIDATE SHIFTS IN PALEOECOLOGICAL PHASES

Abstract

Paleoecological relationships can be difficult to interpret when using a single indicator from a sediment core. This chapter integrated geochemical and fossil diatom data from a sediment core from Muskegon Lake, MI, USA, to investigate the response of the lake system to a variety of natural and human changes. The hypothesis for this investigation was that continued human disturbances (e.g., industrialization) have been at an intensity that will not allow ecosystems to return to a pre-disturbance state; and because of other continuing and emerging stressors (e.g. climate change), ecosystems will not evolve to a new state of balance (characterized as steady state and/or an observed equilibrium with biogeochemical processes). Geochemical and diatom inferred trends established in earlier chapters were compared to examine this hypothesis, which, if true, should show that the phases for each proxy determined separately are compatible when merged, and further that the overall contemporary system state is not yet equilibrated (as identified by stabilized trends among proxies). Results of the integration did reveal a similar temporal response to initial disturbance in the G1 and D1 phases, which was correlated with the chronology of human activity. However, there were minor offsets with the disturbance phases of the core identified by geochemical and biological proxies. The transition to the 'recovery' segment showed good overall agreement; though there was dissimilarity between the geochemistry and biology within recovery phase. Understanding the relationships among proxies is important within the context of ecological dynamics, and should be considered in studies where only one proxy may be available.

4.1 Introduction

Aquatic ecosystems are complex and often respond to stressors in unpredictable ways (Scheffer et al. 2001). Earlier chapters in this study have evaluated the chemical and biological indicators archived in lake sediment, both resulting in delineated phases of environmental response to human perturbations. In chapters 2 and 3, it was suggested that interpretations from individual indicators could be validated and enhanced using additional proxies. Therefore, this chapter seeks to integrate earlier biological and chemical phase delineations to investigate the degree to which a single indicator is useful; and determine if overall paleoecological interpretations can be improved using more than one line of evidence.

Integrating indicators can help answer questions about the significance of feedbacks among system constituents (e.g. physical, chemical and biological) which are unclear. Specifically, in the Muskegon Lake core, chemical and biological proxies infer different things; the geochemistry largely reflects *extrinsic* inputs, while the diatoms capture *intrinsic* processes (Carpenter and Brock 2006; Patoine and Leavitt 2006; Das et al. 2009). Theoretically, merging the proxies will tell more about whole system dynamics and interrelationships among the system constituents (Scheffer et al. 2001; Engstrom et al. 2006).

Integration approach

Complexity is inherent in ecological systems as many factors influence the overall ecological state. Concepts of ecological, planetary and human forcings (all components of the environmental system) were discussed in Chapter 1 to introduce the complex nature of systems (Dearing et al. 2006a). Chapters 2 and 3 assessed how individual proxies reflected changing environmental conditions. In chapter 2, sediment geochemistry was examined to evaluate historic reference conditions, and chapter 3 investigated the record of fossil diatoms to demonstrate a primary productivity regime shift; both reconstructed impacts of human influence in a highly perturbed watershed in the Great Lakes region. The intent of this chapter is to gain insight into the interrelationships among system components resulting from abrupt environmental change.

The hypothesis of this chapter was that continued human disturbances (e.g., industrialization) have been at an intensity that will not allow ecosystems to return to a pre-disturbance state; and because of other continuing and emerging stressors (e.g. climate change), ecosystems will not evolve to a new state of balance (characterized as steady state and/or an observed equilibrium with biogeochemical processes).
This hypothesis was tested using the results of previous chapters, with particular attention given to G and D phases delineated based on geochemical and productivity indicators, respectively. For example, to accept this hypothesis, then the interpretation of phases identified by the sediment geochemistry and the fossil diatom species assemblages will demonstrate similar responses to environmental changes, but will not yet be stabalized. The results of integrating proxies in this chapter can be used to infer temporal environmental change in the lake over time, with interest and consideration of system thresholds and resiliency. Identifying thresholds in lake systems is critical for avoiding a system collapse, though this has been reported more frequently in small, shallow lakes than large lakes like Muskegon Lake (Sondergaard et al. 2001; Scheffer and Jeppesen 2007; Scheffer and van Nes 2007). Though, environmental changes are also of concern

for larger lakes under extreme conditions, and to that end, exploring system resiliency in large systems is an important. Muskegon Lake can provide an example of this due to its large size and short residence time that should, at least conceptually, support a high degree of stability compared to shallow lake systems that have been studied (Kenney et al. 2002; Bayley et al. 2007; Scheffer and Jeppesen 2007).

Additionally, this chapter provides suggestions for a single indicator that best "fingerprints" specific environmental disturbances. For example, it is possible to examine data and determine the indicator with the strongest correlation to various human influences. It was expected that the indicator would vary with the activity; for instance, deforestation is best identified by increased erosion of soils, which is reflected in the sediment record by increasing terrestrial elements. This was done for biological and geochemical proxies in the Muskegon Lake core to provide a quick assessment tool. The intent of approach is to evaluate if reducing extraneous data collection in paleoenvironmental investigations is possible.

4.2 Methods

An integrated investigation of ecological dynamics in Muskegon Lake used the history of the study area, sampling methods, and description of sediment core characteristics from Chapter 1. Methods for core chronology and sediment geochemical chronologies are detailed in Chapter 2. Methods for diatom sampling and processing, as well as fossil diatom taxa data are included in Chapter 3. Chapters 2 and 3 detail methods for delineating zones for geochemical and diatom phases, respectively.

Phase integration

Ecological phases (G and D) determined for the Muskegon Lake core using trends of geochemistry and diatom taxa were compared by the changes at each sediment level. This assessed how each proxy identified phases, and contrasted the response of the indicators.

4.3 Results

Results for sediment geochemistry and fossil diatoms are given in Appendix III - IV. Figures detailing geochemical and diatom phases are found in Chapters 2 and 3.

A biogeochemical history of Muskegon Lake's response to human perturbation was reconstructed using the phases delineated by geochemical and biological proxies. A compilation of phases, with major watershed events is shown in Figure 4.1. Beginning with the pre-disturbance phases (e.g. GI and DI), there is an offset between the sediment level which the chronology (and corresponding wood chips) indicates as the onset of logging in the watershed and the actual proxy response. However, despite the offset in established chronology, there is good agreement with the indicators for the phase I to phase II transition. There was a 1 cm difference in the phase transition here, which can likely be attributed to the resolution of analysis. Both the GIIA and DIIA sub-phases are completely within the logging period, which was the first major human perturbation in the watershed. Geochemistry was first to transition to out of IIA, when GIIB starts at 97 cm depth, followed by DIIB transition (92 cm depth). Though this is a 5 cm difference, when accounting for analysis resolution, there is near agreement with this response as well. The change to sub-phase IIC for both indicators occurs during the time of the Great Depression/Dust bowl, at 57 and 61 cm depth for geochemistry and diatom assemblages, respectively. The diatoms are the first to transition here, but again the analysis resolution makes this shift nearly concomitant. The economically depressed Great Depression was

Geochemistry Phase	Diatom Phase (average planktonic:benthic)	Watershed Events
GIII	DIIIB (1.97)	Flood (2004)
		Dreissena polymorpha appear (~1990)
	DIIIA (13.55)	Flood (1986)
		WWTP Installed (1973)
		Clean Air/Water Acts (1970)
		US131 opens (1964)
GIIC	DIIC (9.41)	Heavy industry and manufacturing (1950s - 1960s)
		WWII (early 1940s)
		Great Depression/ Dust Bowl (1930s)
GIIB	DIIB (3.60)	
		WWI (1914-1918)
	DIIA (2.41)	Logging
GIIA		1874 Fire
		Logging
GI	DI (3.21)	Pre-logging (1835)

Figure 4.1: A comparison of geochemical and biological phases as related to watershed events.

followed by a resurgence of manufacturing in the city of Muskegon during and post WWII. The industrialization in the watershed temporally corresponds to significant geochemical disturbance (sub-phase GIIC), and to the DIIC sub-phase which included taxa that were tolerant of high nutrient and pollution conditions (see Chapter 3). Next, the IIC disturbance sub-phases shift to the phase III of recovery. In the recovery phase, geochemistry shows a singular zone of steadily decreasing concentrations of anthropogenic elements (see Chapter 2); however, the diatom taxa specify two sub-phases that are distinguished by a shift dominant productivity habitat at 8 cm depth.

Land use, population and climate data were incorporated to this integration. Figures 4.2a - 4.2d shows trends from AD 1800 for land use, population, precipitation and temperature, respectively.

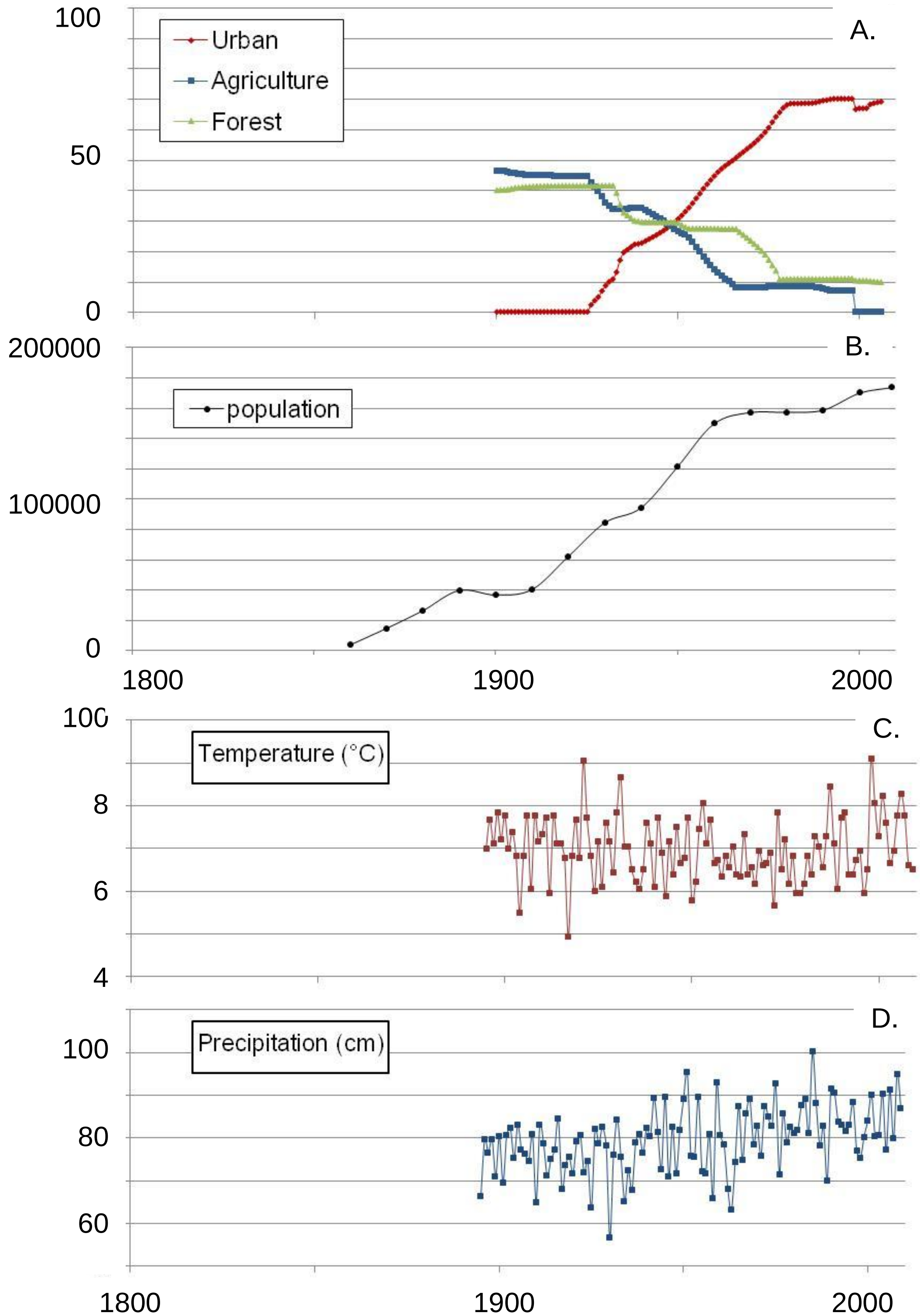

Figure 4.2a – d: Trends from 1800 for a) land use change in the Muskegon River Watershed; b) population in Muskegon County; c) precipitation; and d) temperature in the state of Michigan.

4.4 Discussion

An integration of geochemical and fossil diatom data in a sediment core from Muskegon Lake compared phases of human influence to better understand system response. Results show that the causal agents for many of the chemical and biological changes in the core were directly related to watershed events. Human activity is the dominant agent of change in this system, as land use alterations and population increases have impacted the dynamics and trophic status of Muskegon Lake.

Beginning with phase I, the delay in response of the system at the chronological start of logging to the G and D phase transition could be due to the time it took for the system to reflect a response, or attributed to an error in chronology. Both possibilities reflect the close agreement with the transition, importantly indicating that both proxies responded at approximately the same time, and/or had the same offset in response to the logging perturbation. The shift to the IIB sub-phase was associated with the start of industrialization in the watershed, marked by increased metals and change in select diatom taxa. The change to the GIIB sub-phase occurred before the change to DIIB, suggesting that the response to an extrinsic change was inferred more quickly by the geochemical proxies than the diatom taxa. This may expected, as the erosion associated with logging would quickly halt export and deposition of geochemical terrestrial proxies compared to the response of intrinsic processes driving diatom productivity. In contrast, the transition to sub-phase IIC occurs first for the biological indicator. Though the two indicators are closely associated, the difference in response at this sediment level may suggest that the biological system was the first to be impacted by the climate conditions associated with the Dust Bowl. Alternatively, the lag-time may suggest that the reduction of anthropogenic elements occurred slowly in tandem with a manufacturing slow down. GIII trends toward a geochemical recovery from industrialization, though the DIII phases is subdivided to DIIIA and DIIIB; this suggests something is going on biologically in the lake, which is not significant from a geochemical perspective – it is possible this is related to the zebra mussel invasive species – which is plausible as it is chronically supported – or it could be due to lake level changes that is increasing benthic habitat – or the decrease in water column P, thus decreasing planktonic taxa and allowing the residual P in sediment to support greater benthic productivity).

It is now possible to return to the questions asked in the beginning of this study to consider how using the multi-proxy approach furthered an understanding of system behavior, and how this insight can be a useful tool for other aquatic ecosystems.

How have systems responded to multiple anthropogenic stressors?

Many of the stressor-responses observed in the Muskegon Lake sediment core appear to be sensitive to a combination of human forcings. The complexity of having various stressors through time (deforestation, then industry, then invasive species) as well as those that concurrently influence a system (climate, land use modifications, invasive species) require a detailed interpretation of high resolution samples. Untangling what happened when and the subsequent system response to "x" stress by "y" proxy can be difficult. For example, in this study, direct sensitivity was seen in the geochemical proxies to logging and industrial events while the biostratigraphy of fossil diatoms reflected sensitivity to nutrient inputs; indicating that different indicators are proxies for different stressors. If an indicator is disregarded, it may not be possible to evaluate all agents of change in the system. Merging the stressor-responses using a multi-proxy approach in the Muskegon Lake system resulted in the three major phases of influence reflected in the core.

The physical constituent of Muskegon Lake was not directly analyzed in this investigation. However, a generalization of observed regional conditions can be used to infer both the state of the system and overall dynamics that may have influenced the system. For example, the effect of climate is difficult to interpret for Muskegon Lake, as it is moderated by the adjacent, larger Lake Michigan. To that end, it has been reported that climate has been changing in the Great Lakes region for at least the past 100 years (Magnuson et al.1997; 2000). This affects not only lake temperature, which in and of itself would greatly influence productivity, but also ice phenology, which has the potential to disrupt seasonal chemical and biological coupling. An example is this interrelationship between the seasonal influx of nutrients and the Spring diatom blooms (Winder and Schindler 2004). A shift in one or the other could disjoint the synchrony of processes that the system relies on for its overall flow of energy.

How does the current ecosystem state compare to the pre-perturbation state?

In chapter 1, the conceptual model for ecological forcings discussed the 2 and 3 component system. If the results from this investigation are put in the context of Figure 1.1 from the first chapter, it illustrates how a change in one constituent can impact the whole system. For example, in the Muskegon Lake core, we learned that socio-economic developments (e.g. industrial jobs = population growth and urbanization) can have a significant impact on the physical, chemical and biological constituents, thus shifting the system from a two component to a three component human-dominated system (Dearing et al. 2008; Messerli et al. 2000). Using insight from this study, the conceptual figure was modified to fit the Muskegon Lake ecosystem. Figure 4.3 reconfigures the component relationships from the original figure, to show the relative influence each has on the other (depicted by boldness of the arrow). Using the Muskegon Lake ecosystem to understand the concept, it was indicated that the human forcing has become more dominant, which is predicted to continue.

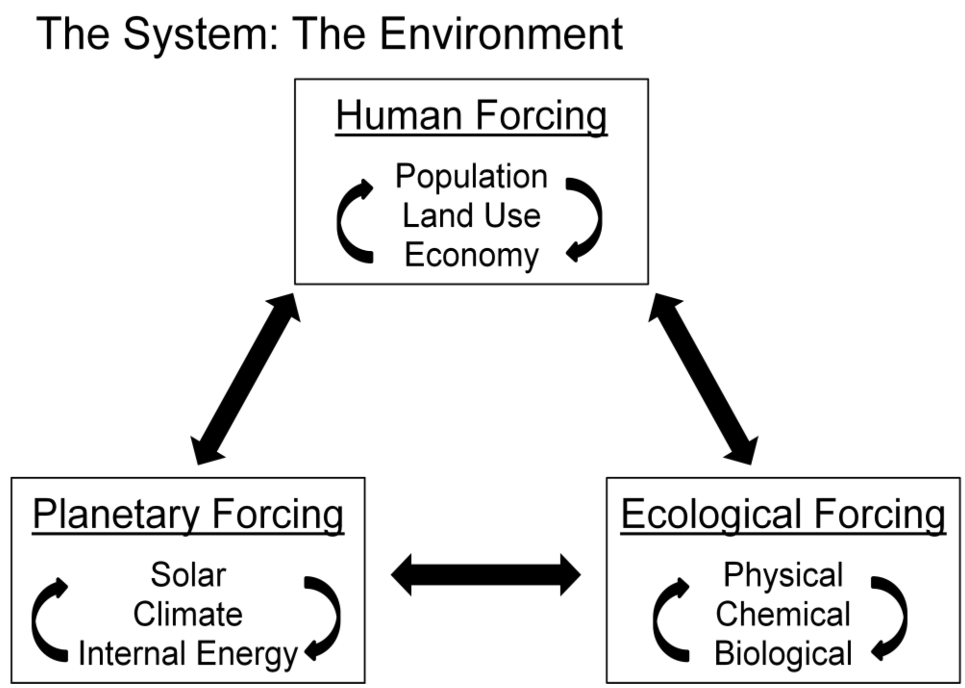

Figure 4.3: Conceptual map of Muskegon Lake ecosystem interactions (modified from Dearing 2006)

Is it possible to infer significant system regime shifts using indirect (e.g. fossil diatom) proxy methods?

In terms of regime shifts, this study suggests it is possible to identify regime shifts using indirect proxies, particularly that of fossil diatoms, as primary productivity is an important indicator of overall system function. Because of the dramatic floristic change observed, especially in the relative abundances of planktonic v. benthic diatoms, the Muskegon Lake system varies from other systems in that the benthic community, not the planktonic, becomes most dominant in modern systems (Garrison and Fitzgerald 2005).

Does using multiple proxies (e.g. chemical and biological) improve interpretations of environmental change?

Using multiple proxies to interpret the Muskegon Lake core did improve the overall understanding of system dynamics. Without utilizing more than one line of evidence, there would be greater uncertainty in the phase delineation and less insight about the geochemical/biological interrelationships observed in the system. Additionally, identifying the role of social v. natural drivers was more robust using the multi-proxy approach. This study is unique as it explores a system that has been significantly impacted by humans in a very short temporal scale. While other studies evaluate long term dynamics in systems where the environmental response is over thousands of years and encompasses broader scale influences (see Ekdahl et al. 2007; Dearing et al. 2008), this study has combined high resolution data with historic observational records to evaluate the specific role of human influence and specific feedbacks in the chemical and biological response such as nutrient input and primary productivity. A comparison of climate attributes (see figures 4.2c-d) and human changes (see figures 4.2a-b) in the Muskegon Lake region further emphasizes that environmental change in this study has been dominated by human activity.

Can we predict future ecosystem recovery and/or a stable state scenario due to the advent of environmental regulations (e.g. Clean Air/Clean Water Acts)?

The dynamics of Muskegon Lake can also be conceptually linked to Figure 1.2, also from Chapter 1. This figure is evaluated within the context of future uncertainty

(Higgins et al. 2002; Bennett et al. 2003), shown in Figure 4.4, and is important to consider as the cumulative effects of overall global change on the Muskegon Lake system continues. Importantly, environmental legislation in the past 40 years has put the system on a trajectory towards scenario C2 (as discussed in Chapter 3), or perhaps an ideal C3. However, depending on how emerging stressors develop, and how society decide to respond, the resulting system (depicted by the range within the circle) could potentially be a severely impacted one, as shown by scenario C1. As this figure represents, it is important to consider the environmental decision making of different nations, as developing nations have the greatest potential to shape their impact trajectory. For that reason, it is important to evaluate ongoing system response to capture the impact of present-day (and future) human induced stressors (Mayer et al. 2004; Walker and Meyers 2004). Among future stressors, the role of invasive species is critical to the overall status and dynamics of Muskegon Lake.

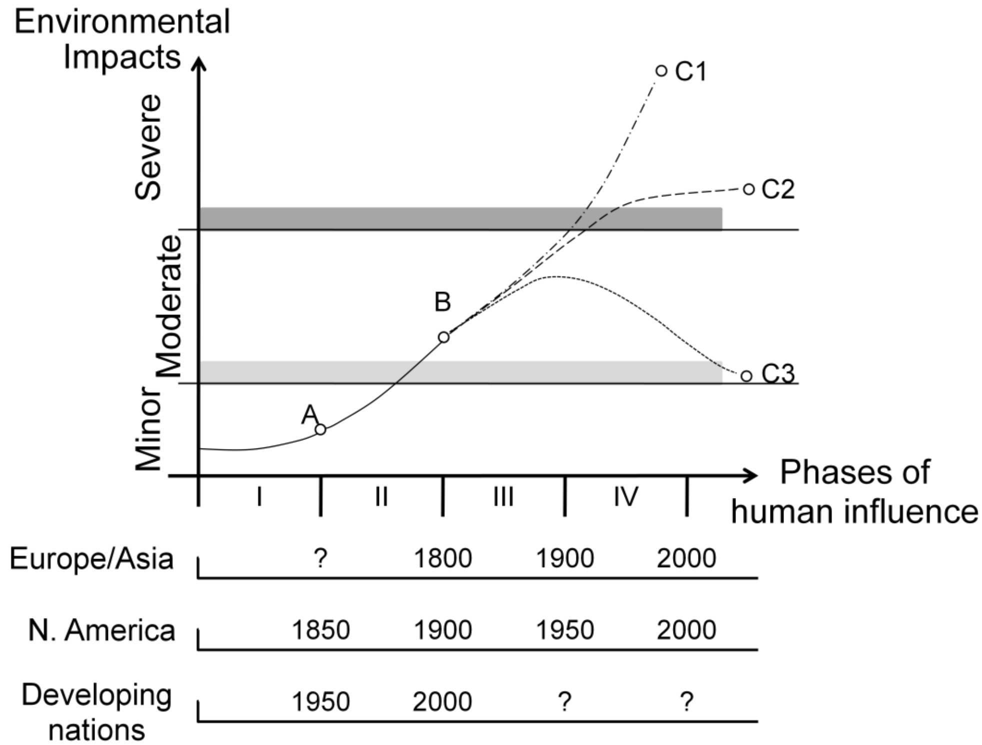

Figure 4.4: Conceptual model of environmental response to anthropogenic influence in varying regions delineated by development status; and under contrasting management strategies through time. (Modified from Garcier 2007)

4.5 Conclusion

The Muskegon Lake core revealed a chronological accumulation of sediment which recorded a sequence of historical events and stressor-responses in the watershed. As stated in the first chapter, the goal of this research was to examine environmental change through the lens of an "environmental systems approach" that would interpret ecological processes. The overarching hypothesis was that ecosystems highly disturbed by human activity cannot be expected to return to a pre-disturbance state; and further, with the influence of climate change and other emerging stressors, ecosystems will not obtain a new regime (steady state and/or equilibrium). This hypothesis was accepted, as results from chapters 2-4 demonstrate that the current ecological state of the system is not same as the pre-perturbation state. There have been alterations to geochemical profiles, as well as significant shifts in the diatom community structure that suggest an altered ecological state. Further, because of Muskegon Lake's fluctuating geochemical concentrations and dynamic changes in biostratigraphy, it is concluded that this new state is not stable. Due to the uncertainty of emerging stressors (e.g. climate change, invasive species, and land use alterations in the watershed), it is not possible to predict when the system will become stable, and the observed changes are likely irreversible. Revisiting specific objectives of this study should make clear what the research was able to accomplish and contribute to the field of study.

The first objective was to determine how aquatic ecosystems responded to specific perturbations (e.g. logging, industry, urbanization) by using individual paleoecological proxies (e.g. geochemistry and diatoms). This objective was met by a detailed analysis of high resolution data that clearly delineated phases of human influence as recorded by lake sediment. With the geochemical profiles, the anthropogenic proxy group best reflected many of the stressor-responses in the watershed, particularly the intense period of industry in the Muskegon area. The terrestrial proxy group captured the onset of logging, while interpretations of the productivity proxy group led to questions further addressed by the biological component. The fossil diatoms responded both on an individual level, and as classified by habitat. The most significant influence to the diatom community seemed to be triggered recently. The data suggest this is partly in response to the installation of a waste water treatment facility, and also by recent ecological influences of cultural nutrient inputs and invasive species (e.g. zebra mussels). Overall, using individual proxies was useful to fingerprint specific watershed scale events. Though, it was more difficult to clearly define the impact of broader scale stressors (e.g. climate), suggesting a need for continued monitoring of the ecological status of the system.

A second objective was to determine the environmental legacy of anthropogenic stressors by calculating deviations from geochemical reference concentrations and by identifying ecological regime shifts. This objective was discussed in detail in chapters 2 and 3, that evaluated reference conditions and ecological regime shifts, respectively. Specifically, it was observed that many modern geochemical concentrations (e.g. anthropogenic proxy group elements) have not decreased to pre-perturbation concentrations. This suggests that: 1) the system is still being influenced by the legacy of contamination, and 2) that the system is under the influence of modern inputs of anthropogenic elements, likely from existing industrial practices, atmospheric inputs, and watershed runoff (e.g. agriculture and urban land uses). Given the current scenario of a continued human-dominated system, the calculated deviations from references are not likely to return fully to the historical reference, but continue a trajectory of an adapting reference state. A regime shift from planktonic to benthic dominated primary production was identified using fossil diatoms. The most dramatic shift is near the top of the core, and likely reflects a combination of the legacy of nutrients stored in the sediment with the emergence of zebra mussels which allow greater light penetration. This scenario has led to a considerable increase in the relative abundance of benthic diatoms. Given this stressor-response signal, it can be predicted that the biological communities will continue to restructure under the influence of regional climate changes (Magnuson et al. 2000) and nutrient inputs (Steinman et al. 2008).

Third, developing correlations/relationships among geochemical and diatom sediment chronologies to infer process (e.g. lag-time and feedback mechanisms). This chapter integrated geochemical and biological phases determined from earlier chapters to evaluate how they are related, and what it indicates for overall system behavior. Coupling the indicators allowed processes, suspected from interpretations of individual proxies, to be better interpreted. In the anthropogenic phases (II and III for both G and D indicators), the integrated data support the significant role of human disturbance in driving system change, as both indicators respond to this influence. Also, using select responses with the core proxies, it was possible to identify causal agents for the observed environmental changes (e.g. deforestation and industry). Moreover, this chapter demonstrated it is possible to link geochemical and biological response (e.g. nutrient inputs (P) and select diatom taxa). Understanding these relationships are valuable for environmental management strategies, particularly as threats to aquatic ecosystems continually evolve.

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Literature Cited

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Appendix

APPENDIX I

DESCRIPTION OF MUSKEGON LAKE CORE SEDIMENT CHARACTERISTICS

Table A.1: Description of Muskegon Lake Sediment Core Characteristics

Sample #	**Thickness (cm)**	**Depth (cm)**	**Description**
1	1.0	1.0	dark, extremely runny
2	1.0	2.0	dark and very runny
3	1.0	3.0	dark and runny
4	1.0	4.0	dark and runny
5	1.0	5.0	dark and runny
6	1.0	6.0	dark and runny
7	1.0	7.0	dark and runny
8	1.0	8.0	dark and somewhat runny
9	1.0	9.0	dark
10	1.0	10.0	
11	1.0	11.0	dark brown
12	1.0	12.0	dark brown
13	1.0	13.0	dark brown
14	1.0	14.0	dark brown
15	1.0	15.0	dark brown
16	1.0	16.0	dark brown
17	1.0	17.0	dark brown
18	1.0	18.0	dark brown
19	1.0	19.0	dark brown
20	1.0	20.0	
21	1.0	21.0	dark brown, small flowoiejr?
22	1.0	22.0	dark brown
23	1.0	23.0	dark, moderately thick
24	1.0	24.0	dark, moderately thick
25	1.0	25.0	looks brown, turns darker when mixed
26	1.0	26.0	dark, moderately thick
27	1.0	27.0	dark, moderately thick
28	1.0	28.0	dark, moderately thick
29	1.0	29.0	dark, moderately thick
30	1.0	30.0	
31	1.0	31.0	less oil smell, more brown, thinner consistancy
32	1.0	32.0	dark, moderately thick
33	1.0	33.0	dark (brownish)
34	1.0	34.0	dark

Table A.1: Description of Muskegon Lake Sediment Core Characteristics, continued

35	1.0	35.0	dark – oil odor
36	1.0	36.0	dark, moderately thick – shell fragments
37	1.0	37.0	dark
38	1.0	38.0	dark – oil odor
39	1.0	39.0	dark, moderately thick
40	1.0	40.0	
41	1.0	41.0	dark, moderately thick – shell fragments
42	1.0	42.0	dark, moderately thick
43	1.0	43.0	dark, moderately thick
44	1.0	44.0	dark, moderately thick
45	1.0	45.0	dark, moderately thick – shell fragments
46	1.0	46.0	dark, moderately thick
47	1.0	47.0	dark, moderately thick
48	1.0	48.0	dark, moderately thick – shell fragments
49	1.0	49.0	dark, moderately thick – shell fragments
50	1.0	50.0	dark, moderately thick
51	1.0	51.0	dark, moderately thick
52	1.0	52.0	dark, moderately thick – shell fragments
53	1.0	53.0	dark, moderately thick – shell fragments
54	1.0	54.0	dark, moderately thick
55	1.0	55.0	dark, moderately thick
56	1.0	56.0	dark, moderately thick – shell fragments
57	1.0	57.0	dark, moderately thick – shell fragments and organic
58	1.0	58.0	dark, moderately thick – shell fragments
59	1.0	59.0	dark, moderately thick
60	1.0	60.0	
61	1.0	61.0	dark, thick – organic fragments
62	1.0	62.0	dark, thick
63	1.0	63.0	dark, thick – shells and organic fragments present
64	1.0	64.0	dark, thick – organic fragments
65	1.0	65.0	dark, thick
66	1.0	66.0	dark, thick – shells present
67	1.0	67.0	dark, thick – shells present
68	1.0	68.0	dark, thick
69	1.0	69.0	dark, thick – shells present

Table A.1: Description of Muskegon Lake Sediment Core Characteristics, continued

70	1.0	70.0	dark, moderately thick
71	1.0	71.0	
72	1.0	72.0	dark, thick
73	1.0	73.0	dark, thick
74	1.0	74.0	dark, thick
75	1.0	75.0	dark, thick
76	1.0	76.0	dark, thick
77	1.0	77.0	dark, thick
78	1.0	78.0	dark, thick
79	1.0	79.0	dark, thick
80	1.0	80.0	
81	1.0	81.0	dark, thick
82	1.0	82.0	dark, thick with some tan streaks
83	1.0	83.0	dark, thick
84	1.0	84.0	dark, thick
85	1.0	85.0	dark, thick
86	1.0	86.0	dark, thick
87	1.0	87.0	dark, thick
88	1.0	88.0	dark, thick
89	1.0	89.0	dark, thick
90	1.0	90.0	
91	1.0	91.0	dark, thick
92	1.0	92.0	dark, thick
93	1.0	93.0	dark, thick
94	1.0	94.0	dark, thick – more variation in color
95	1.0	95.0	dark, thick
96	1.0	96.0	dark, thick
97	1.0	97.0	dark, thick
98	1.0	98.0	dark, thick
99	1.0	99.0	dark brown, thick
100	1.0	100.0	
101	1.0	101.0	dark brown, thick
102	1.0	102.0	dark brown, thick
103	1.0	103.0	dark brown, thick
104	1.0	104.0	dark brown, thick

Table A.1: Description of Muskegon Lake Sediment Core Characteristics, continued

105	1.0	105.0	dark brown, thick
106	1.0	106.0	dark brown, thick
107	1.0	107.0	dark brown, thick
108	1.0	108.0	dark brown, thick
109	1.0	109.0	dark brown, thick – small rock/fossil present
110	1.0	110.0	
111	1.0	111.0	dark brown, thick – chunk of some sort of matter
112	1.0	112.0	dark brown, thick
113	1.0	113.0	grainy, some stringy matter
114	1.0	114.0	dark brown, thick (very homogenous color)
115	1.0	115.0	dark brown, thick – shell fragments
116	1.0	116.0	dark brown, thick
117	1.0	117.0	dark brown, thick
118	1.0	118.0	dark brown, thick
119	1.0	119.0	dark, thick
120	1.0	120.0	
121	1.0	121.0	dark brown, thick
122	1.0	122.0	dark brown, thick
123	1.0	123.0	dark brown, thick
124	1.0	124.0	dark brown, thick
125	1.0	125.0	dark brown, thick
126	1.0	126.0	dark brown, thick
127	1.0	127.0	dark brown, thick
128	1.0	128.0	dark brown, thick
129	1.0	129.0	dark brown, thick
130	1.0	130.0	dark brown, thick – more goopy texture
131	1.0	131.0	dark brown, thick
132	1.0	132.0	dark brown, thick
133	1.0	133.0	dark brown, thick
134	1.0	134.0	dark brown, thick – organic fragments
135	1.0	135.0	dark brown, thick
136	1.0	136.0	dark brown, thick
137	1.0	137.0	dark brown, thick
138	1.0	138.0	dark brown, thick – slightly greenish tinge
139	1.0	139.0	dark brown, thick

Table A.1: Description of Muskegon Lake Sediment Core Characteristics, continued

140	1.0	140.0	
141	1.0	141.0	dark brown, thick
142	1.0	142.0	dark brown, thick
143	1.0	143.0	dark brown, thick
144	1.0	144.0	dark brown, thick
145	1.0	145.0	dark brown, thick
146	1.0	146.0	dark brown, thick
147	1.0	147.0	dark brown, thick
148	1.0	148.0	dark brown, thick
149	1.0	149.0	dark brown, thick

APPENDIX II

MUSKEGON LAKE SEDIMENT CORE POROSITY

Table A.2: Muskegon Lake sediment core porosity

Depth (cm)	Wet Weight (g)	Dry Weight (g)	% Ø
1			
2			
3			
4	6.26	3.69	41.04
5	10.40	4.17	59.88
6	9.66	4.20	56.51
7	8.20	4.00	51.23
8	8.39	4.08	51.38
9	9.31	4.20	54.92
10			
11	9.26	4.30	53.58
12	9.33	4.25	54.42
13	9.43	4.20	55.43
14	9.48	4.19	55.83
15	8.12	3.95	51.35
16	8.23	3.99	51.58
17	10.40	4.30	58.66
18	11.15	4.55	59.20
19	9.70	4.28	55.84
20			
21	10.64	4.37	58.89
22	10.80	4.42	59.06
23	11.55	4.54	60.75
24	12.10	4.52	62.65
25	11.30	4.47	60.43
26	11.34	4.58	59.62
27	11.89	4.60	61.29
28	9.54	4.39	54.01
29	10.95	4.61	57.88
30			
31	10.83	4.51	58.35
32	10.10	4.35	56.88
33	11.37	4.59	59.62
34	11.44	4.58	59.96
35	10.47	4.53	56.71
36	11.31	4.71	58.37
37	12.40	4.76	61.62
38	9.32	4.40	52.75
39	12.24	4.77	61.08
40			
41	12.17	4.79	60.60
42	11.77	4.82	59.02
43	11.55	4.86	57.94
44	10.02	4.41	55.99
45	11.47	4.63	59.67
46	13.24	5.26	60.24
47	7.71	4.38	43.21
48	6.93	2.80	59.69

Table A.2: Muskegon Lake sediment core porosity, continued

Depth (cm)	Wet Weight (g)	Dry Weight (g)	% Ø
49	6.80	2.63	61.39
50			
51	8.03	2.95	63.33
52	6.57	2.62	60.19
53	7.25	2.69	62.90
54	7.82	2.82	63.92
55	7.79	2.63	66.27
56	8.63	3.04	64.79
57			
58	7.88	2.67	66.18
59	8.42	2.73	67.56
60			
61	6.49	2.61	59.77
62	7.83	2.78	64.46
63	10.30	3.22	68.71
64	6.25	2.49	60.19
65	6.82	2.79	59.09
66	8.90	2.95	66.91
67	6.91	2.72	60.67
68	5.85	2.59	55.77
69	6.27	2.71	56.86
70			
71	7.46	2.86	61.66
72	9.17	3.22	64.85
73	6.94	2.77	60.14
74	8.64	3.07	64.48
75	7.06	2.75	61.03
76	9.29	3.34	64.05
77	8.86	3.26	63.17
78	8.19	3.20	60.95
79	5.78	2.78	51.94
80			
81	7.76	3.18	58.98
82	8.35	3.23	61.27
83	8.63	3.32	61.57
84	5.70	2.66	53.28
85	9.05	3.28	63.80
86	7.04	2.96	57.97
87	7.31	2.98	59.26
88	8.89	3.38	62.01
89	7.28	3.06	57.97
90			
91	7.54	3.15	58.22
92	8.14	3.33	59.14
93	8.62	3.50	59.45
94	7.70	3.29	57.27
95	8.69	3.62	58.37
96	9.21	4.80	47.89

Table A.2: Muskegon Lake sediment core porosity, continued

Depth (cm)	Wet Weight (g)	Dry Weight (g)	% Ø
97	9.30	4.24	54.45
98	6.23	3.25	47.79
99	7.14	3.55	50.29
100			
101	7.20	3.35	53.46
102	9.68	3.88	59.95
103	7.11	3.19	55.15
104	6.73	2.94	56.27
105	6.61	2.94	55.49
106	9.09	3.60	60.42
107	6.11	2.84	53.50
108	6.65	2.98	55.27
109	6.28	2.89	54.02
110			
111	7.65	3.65	52.35
112	9.22	4.33	53.04
113	5.99	3.44	42.55
114	7.58	3.93	48.14
115	6.80	3.62	46.69
116	7.62	4.18	45.10
117	7.45	3.92	47.39
118		3.12	
119	6.56	3.14	52.21
120			
121	7.06	3.09	56.21
122	5.23	2.60	50.30
123	6.22	2.87	53.90
124	6.04	2.82	53.30
125	4.76	2.50	47.56
126	7.89	3.23	59.02
127	4.68	2.47	47.28
128	6.61	2.99	54.78
129	6.70	3.03	54.79
130	5.79	2.80	51.69
131	5.92	2.77	53.17
132	5.72	2.74	52.04
133	4.96	2.60	47.68
134			
135	6.50	2.95	54.64
136	6.28	2.85	54.55
137	6.04	2.82	53.37
138	6.63	2.95	55.47
139	6.05	2.40	60.29
140			
141	5.99	2.83	52.72
142	8.23	3.34	59.46
143	7.38	3.16	57.15
144	8.47	3.42	59.57

Table A.2: Muskegon Lake sediment core porosity, continued

Depth (cm)	Wet Weight (g)	Dry Weight (g)	% Ø
145	8.07	3.32	58.88
146	7.84	3.29	58.03
147	10.28	3.89	62.14
148	7.93	3.42	56.85
149	6.91	3.13	54.77

APPENDIX III

MUSKEGON LAKE CORE SEDIMENT GEOCHEMISTRY

Table A.3: Muskegon Lake core sediment geochemistry, continued

Depth (cm)	Al	K	Ti	Mg	Fe	Mn	As	Ca	P	Sr	Rb	Ba
2	9978.85	2049.04	33.28	10219.67	21253.80	1980.67	13.28	64387.01	1279.41	103.91	14.84	180.78
3	13161.52	1742.96	72.41	15177.78	31780.04	2560.08	11.79	84021.60	1858.27	109.16	16.36	208.00
4	11742.92	1369.38	15.88	15343.99	32625.32	2280.28	9.64	78186.48	2097.85	87.15	12.55	191.76
5	14495.44	1477.47	58.36	17233.82	37730.29	2852.70	11.12	92746.27	1520.56	102.37	14.71	187.93
6	13283.47	1390.72	53.67	16046.40	35409.32	2498.73	11.44	103061.02	1305.57	114.17	14.19	172.48
7	12900.00	1364.79	51.54	15532.28	34835.86	2549.16	11.18	106254.85	1301.05	118.16	14.24	170.89
8	12222.83	1548.15	19.74	13644.09	32367.72	2263.19	12.26	74592.91	1235.43	89.47	14.15	157.46
9	13802.96	1413.02	16.03	14683.60	33333.40	2157.25	12.00	77730.24	1502.96	81.44	13.64	158.40
11	17498.94	1613.21	67.81	19164.54	45196.60	2483.44	12.85	69449.04	1054.65	77.62	16.62	219.41
12	16975.98	1562.05	98.18	18426.76	44202.73	1830.86	13.28	62524.80	897.21	65.43	15.14	139.57
13	19623.06	1748.86	23.12	19415.71	48792.04	1093.90	14.92	62889.39	2168.78	58.20	14.84	140.84
14	13728.39	1210.89	42.03	16136.65	38451.91	1321.61	14.75	54119.28	825.95	63.54	13.60	139.49
15	14933.97	1428.35	67.54	15838.89	39167.74	1260.47	18.21	49692.95	893.59	61.60	15.19	145.90
16	17216.63	1729.91	38.21	15921.96	38980.17	910.13	17.40	46577.40	2205.33	50.30	14.71	144.46
17	16159.96	1652.29	34.09	15078.81	35253.18	967.61	17.52	44633.69	2443.43	49.62	14.45	146.61
18	14984.09	1396.39	20.82	15474.84	34532.26	1034.97	15.91	45520.65	2877.63	48.47	13.53	144.13
19	14151.12	1209.53	17.07	15166.04	32827.43	991.79	16.01	43988.99	2896.46	45.56	11.88	141.53
21	10951.39	1178.70	61.94	11664.82	27305.33	1087.42	19.13	37135.18	2085.22	55.10	12.73	173.39
22	13788.03	1056.41	63.23	14804.91	32353.85	1524.36	18.23	54201.28	2556.97	60.45	11.50	191.09
23	17257.00	1382.86	90.93	15993.83	35775.31	1642.18	17.22	61089.51	2697.51	63.19	14.07	197.18
24	17160.39	1286.53	81.18	16092.93	36987.15	1730.84	14.26	60166.38	2604.11	60.64	13.49	187.86
25	14413.81	1201.67	64.73	14944.98	30874.69	1089.96	12.97	52910.25	1922.24	64.73	13.01	226.17
26	13051.68	982.61	46.03	14398.95	30940.97	1102.10	11.51	61490.76	1929.18	73.24	11.07	203.21
27	15008.04	1280.67	72.93	13511.52	31413.70	1413.04	13.50	49267.17	2613.93	65.63	13.65	249.13
28	7220.00	1111.81	62.31	6551.96	14803.51	688.87	11.84	24257.32	2384.97	63.05	12.08	222.02
29	17746.38	1289.36	76.23	16252.34	33943.19	1478.72	13.51	76744.26	2175.62	83.77	13.64	231.45

Table A.3: Muskegon Lake core sediment geochemistry, continued

Depth (cm)	Al	K	Ti	Mg	Fe	Mn	As	Ca	P	Sr	Rb	Ba
31	14072.08	1293.96	69.46	13207.36	29508.01	1417.53	12.49	53707.14	2622.68	72.73	13.94	226.41
32	16261.76	1173.79	75.30	15118.36	33637.67	1726.96	13.73	92102.68	2730.44	94.91	12.66	239.45
33	12304.77	1124.82	75.27	11654.88	28547.51	1532.32	12.75	59683.30	3057.03	82.58	12.19	237.14
34	14526.80	1343.12	84.97	12835.95	32647.93	1575.60	12.51	53459.26	2866.67	75.16	13.73	247.12
35	18891.98	1103.62	59.74	20006.72	42612.13	1977.24	11.34	91088.25	2798.13	82.85	12.50	252.72
36	14120.81	1056.92	57.62	15168.15	32157.96	1612.10	11.32	73987.69	3005.84	89.51	12.19	238.66
37	24464.19	1345.50	92.77	20693.89	45994.10	2313.32	12.03	103809.17	3075.48	89.48	14.06	226.90
38	15375.47	1096.65	65.38	15909.36	33532.02	1568.81	11.10	99313.72	2700.21	109.42	12.06	228.77
39	15449.79	946.95	61.22	16742.53	41090.25	1343.98	10.52	52808.51	2975.81	93.44	10.60	203.76
41	21072.39	1183.43	85.00	16910.65	47799.35	1946.96	12.72	79235.00	2752.37	77.93	12.78	334.61
42	20509.83	1296.88	87.95	16846.79	44226.28	2067.31	12.09	73754.91	2706.75	72.44	13.40	222.44
43	13969.94	1117.55	70.06	14643.56	35904.70	1724.34	10.92	54789.16	3670.06	71.10	12.51	224.28
44	16023.66	1478.50	91.53	13197.23	31937.15	1417.56	12.61	51339.93	2665.36	68.52	14.68	228.89
45	12527.65	980.95	77.71	11566.95	40424.84	2334.99	10.11	48507.99	4885.64	58.25	10.50	192.59
46	15133.26	1176.01	51.74	14787.34	34381.55	1323.82	10.11	72023.61	2139.46	82.02	12.60	225.15
47	13598.71	1238.05	75.11	10933.26	30202.79	1021.89	11.12	52813.95	1726.39	70.99	12.38	198.52
48	17771.88	1252.55	66.30	13917.62	39991.49	972.67	11.35	63432.48	1255.37	69.52	12.67	207.49
49	13531.32	1157.34	57.17	11562.42	31991.36	1023.54	11.30	49516.63	1726.07	66.26	12.18	194.75
51	13706.86	1405.54	75.26	10914.10	35490.67	1322.67	15.77	33819.81	2520.13	54.76	14.59	192.86
52	14955.25	1182.78	51.91	13886.94	36632.76	1671.52	14.15	55504.28	2698.20	68.72	12.89	212.46
53	19072.18	1386.82	69.48	16170.08	46636.19	1283.05	12.58	50693.51	1884.36	53.96	14.28	184.74
54	24222.46	1159.48	33.71	24736.72	59718.79	1809.72	9.01	73087.26	1681.19	58.29	13.43	213.52
55	16013.58	1210.82	56.34	13896.09	40082.30	1064.40	13.62	46747.53	1351.54	57.35	12.82	190.02
56	14763.40	1598.52	25.66	11871.60	34141.80	604.18	14.48	40655.60	2065.80	53.40	14.32	194.18
57	14183.04	1326.27	17.44	13771.20	32102.76	706.92	12.09	46882.84	2299.41	55.90	13.00	193.25
58	10295.63	1023.17	17.77	9832.10	25927.51	634.89	11.79	50965.94	2012.45	63.28	9.91	162.55

Table A.3: Muskegon Lake core sediment geochemistry, continued

Depth (cm)	Al	K	Ti	Mg	Fe	Mn	As	Ca	P	Sr	Rb	Ba
59	11923.82	1052.08	17.05	11870.89	35989.04	643.40	13.52	52347.26	2333.84	59.89	10.19	178.70
61	13114.15	844.62	21.09	14512.26	39978.87	1123.40	7.00	50430.19	1668.94	61.92	10.08	178.70
62	15142.34	1342.70	54.60	14335.74	37799.15	1094.04	13.00	52997.87	1546.45	75.60	14.64	208.02
63	21962.34	1382.28	66.27	19073.45	56997.36	1588.51	15.33	59761.02	1745.95	61.32	14.39	199.89
64	18232.04	1167.51	55.88	17565.31	55927.14	1577.76	15.53	64926.12	1807.29	62.35	12.63	182.12
65	12428.39	1018.81	51.82	12619.07	37058.69	1194.28	12.86	70744.92	1747.44	77.18	11.27	176.02
66	20247.45	1417.35	76.70	18911.41	48888.39	1781.47	13.40	77783.91	2265.52	69.33	14.97	183.46
67	16148.16	1572.05	58.24	16149.59	35723.77	1330.33	11.35	38249.59	2240.68	52.36	17.56	201.86
68	17505.14	1798.66	71.34	16694.24	34369.14	1107.41	11.28	42742.39	1652.65	61.01	19.61	215.41
69	15122.80	1422.26	40.38	16229.71	33786.61	1096.03	8.95	37840.59	1706.30	51.23	16.55	198.70
71	15137.75	1840.13	79.08	13067.22	28884.16	910.31	10.64	38108.29	1295.80	60.57	19.34	199.17
72	15664.24	1490.25	49.25	16550.29	34711.85	1064.44	6.88	36441.49	1488.51	52.41	16.39	199.58
73	17247.57	1889.36	65.13	15610.67	33050.56	885.21	9.53	39007.12	1207.79	56.48	20.77	210.60
74	18396.31	1832.09	67.83	15878.69	34876.84	929.10	10.51	41110.86	1039.20	64.41	19.80	443.83
75	15969.55	1385.93	30.27	16952.88	36915.43	1180.25	7.49	37526.34	1168.56	50.00	16.95	201.83
76	19079.87	1827.69	64.88	16506.42	36197.64	999.57	10.11	43477.30	1038.99	76.49	19.64	1194.71
77	18142.68	965.71	12.72	22097.45	48607.86	1276.65	5.01	54291.08	1278.17	51.23	12.51	200.08
78	18699.17	1564.05	46.51	19408.88	40917.15	1085.74	10.89	52205.37	1350.52	59.92	18.14	195.27
79	18191.22	1754.76	73.28	17074.95	40259.31	1361.03	15.69	32972.81	1555.58	63.75	18.23	186.47
81	18191.22	1718.31	47.90	17074.95	40259.31	1361.03	11.58	32972.81	1788.35	42.46	19.64	169.64
82	21176.20	2216.16	94.30	17351.77	42280.17	1255.32	13.53	35610.44	1289.60	45.76	22.30	171.94
83	23119.83	1564.77	42.74	20565.61	49805.27	1771.10	9.54	37535.44	1793.76	43.29	17.55	175.11
84	19480.26	1776.81	52.02	20454.23	45896.96	1739.26	11.32	39648.16	1539.44	45.68	18.68	179.78
85	19305.79	1930.39	59.48	19425.11	44911.37	1925.97	11.82	35064.59	1563.15	43.07	20.26	169.08
86	22331.81	2197.69	68.63	21200.00	49343.36	2030.94	11.29	36849.24	1479.76	43.25	22.85	173.18
87	21703.39	2194.47	65.06	20361.86	43440.47	1551.06	8.64	33025.21	1255.51	40.00	23.20	169.43

Table A.3: Muskegon Lake core sediment geochemistry, continued

Depth (cm)	Al	K	Ti	Mg	Fe	Mn	As	Ca	P	Sr	Rb	Ba
88	19764.73	1938.41	61.67	20412.79	46889.53	1512.79	9.52	33246.32	1134.59	44.50	20.83	361.98
89	19411.43	2184.09	64.22	19463.52	47556.26	1560.66	8.53	31330.99	1469.47	42.59	22.97	168.09
91	20812.55	2182.18	66.37	20243.73	43366.42	1335.06	8.25	30753.51	1180.90	42.60	25.85	182.07
92	16419.61	1860.11	43.00	19197.63	42695.26	1376.29	6.49	31231.90	1039.70	42.72	20.84	259.46
93	21146.90	2226.99	78.64	22510.12	53429.55	1691.32	8.61	34736.36	1323.67	42.43	25.89	176.25
94	12249.82	1859.87	37.36	14261.72	34161.90	1084.98	4.58	22152.75	1178.59	38.10	20.20	161.01
95	19714.76	1919.19	49.25	21345.53	49345.11	1655.51	4.05	34543.87	899.77	34.37	20.08	147.23
96	10069.06	872.03	14.10	12685.62	28730.07	964.49	3.09	19722.00	502.35	18.13	10.50	90.02
97	14135.39	1388.07	34.99	16857.65	36963.02	1338.57	2.13	26174.55	851.99	25.92	15.09	111.91
98	14020.83	1458.50	36.36	16958.63	35885.15	1235.65	2.23	26365.86	820.57	26.38	15.51	113.68
99	13906.26	1528.93	37.74	17059.60	34807.27	1132.73	2.32	26557.17	789.15	26.83	15.94	115.45
101	17124.90	1866.24	43.47	19988.82	42338.43	1566.47	3.73	31314.31	1127.04	35.69	19.45	175.04
102	9541.30	1677.04	34.39	11740.22	24966.52	866.09	2.22	18028.26	1396.37	34.83	18.70	152.89
103	13440.78	1740.89	38.00	15520.32	35722.88	1259.27	3.41	25370.02	1324.30	36.20	19.17	154.17
104	17340.26	1804.73	41.61	19300.43	46479.23	1652.46	4.60	32711.78	1252.23	37.58	19.64	155.44
105	21704.27	2323.10	72.03	21035.04	52554.27	1588.89	6.50	33398.08	1376.67	40.71	23.31	164.47
106	15560.92	2175.88	55.97	16071.85	38277.31	1126.26	6.05	25220.17	1246.68	41.70	22.37	167.14
107	20578.79	2285.97	59.03	20559.96	48957.14	1388.31	5.02	34301.52	1116.56	41.43	23.25	170.11
108	15908.95	1646.20	39.50	17355.46	44594.76	1169.87	3.12	28096.94	947.38	36.24	18.58	146.38
109	17486.87	1799.80	53.41	17360.39	49704.38	1175.71	4.11	28461.05	931.12	40.00	19.37	152.28
111	12584.99	1185.50	51.52	12751.43	44771.08	960.04	2.34	23085.65	1371.72	32.47	13.69	115.17
112	8281.96	608.67	24.98	11472.35	45715.29	959.61	0.24	20968.24	1012.45	26.16	8.35	97.55
113	6625.26	633.62	33.20	7418.32	29993.26	527.37	0.72	13765.68	559.03	16.23	7.66	62.40
114	7155.11	693.43	35.84	8455.68	31520.04	574.95	1.52	15856.45	517.07	18.79	8.69	69.52
115	9291.51	813.95	40.00	10214.01	41718.68	733.33	3.52	18360.72	579.70	21.80	10.00	80.17
116	10242.73	937.03	39.49	11274.23	41903.52	782.60	5.62	19979.52	479.16	26.06	11.17	125.37

Table A.3: Muskegon Lake core sediment geochemistry, continued

Depth (cm)	Al	K	Ti	Mg	Fe	Mn	As	Ca	P	Sr	Rb	Ba
117	15053.64	1505.70	59.91	15007.95	50228.70	1020.31	11.85	25286.75	570.60	29.71	16.31	111.90
118	18388.04	1797.49	68.71	17563.73	56289.80	1182.35	12.29	28533.33	618.76	34.47	18.84	130.75
119	14913.52	1326.95	29.25	16676.82	51493.78	1169.10	4.74	28356.01	616.91	31.46	15.94	123.33
121	15213.32	1668.26	46.69	15493.13	47777.22	1152.27	7.18	25998.80	699.90	34.04	18.77	133.87
122	15513.11	2009.57	64.14	14309.43	44060.66	1135.45	9.61	23641.60	782.89	36.62	21.60	144.41
123	21903.25	2003.55	59.81	19786.80	63677.92	1640.04	9.55	31279.00	761.36	37.71	21.52	148.01
124	18951.08	1693.30	38.36	18643.97	59521.98	1712.07	4.07	30016.38	808.43	36.08	19.59	147.24
125	17123.58	1740.96	41.42	16170.09	54256.99	1564.85	3.21	26174.24	808.65	37.90	20.00	145.90
126	14647.40	2096.73	64.52	12799.13	44018.61	1208.44	8.59	20259.74	847.90	36.30	22.23	150.65
127	18924.00	1862.73	47.45	17670.73	60925.45	1815.09	5.71	27993.64	962.67	38.58	20.75	150.27
128	15188.45	1423.11	41.20	15020.52	52515.74	1429.08	3.92	24038.65	761.00	30.76	16.57	201.22
129	18832.75	2158.58	65.11	17964.63	60805.24	1474.89	5.87	28197.60	931.62	42.38	23.84	169.78
130	15337.65	1847.35	48.29	17245.27	41208.44	1573.46	3.89	28154.73	1363.21	38.37	19.59	157.37
131	6985.83	1766.60	47.63	7761.54	22022.27	584.62	5.97	12659.72	721.54	34.47	19.05	134.43
132	17441.03	1782.16	55.98	19954.85	57185.57	1597.94	10.06	33598.97	1041.24	39.22	19.55	156.95
133	18864.10	1915.73	63.78	20382.91	57002.14	1499.15	7.18	32651.28	699.29	35.83	20.26	140.28
134	24123.83	1863.06	49.02	25564.68	73471.28	1953.40	5.98	41042.77	771.32	36.13	20.45	145.57
135	8646.15	975.04	9.64	12953.85	34867.52	938.03	4.69	21527.78	739.10	33.59	12.44	135.41
136	22142.74	1456.14	32.82	28260.79	74115.35	2013.69	3.40	47918.26	767.68	34.92	17.32	138.65
137	16261.87	1521.76	45.88	20387.36	53003.27	1437.25	3.70	34945.75	710.87	34.84	16.78	133.62
138	16544.69	1726.41	56.27	20390.36	51827.25	1531.63	6.49	38999.63	796.33	44.72	18.72	145.48
139	16827.51	1826.14	56.69	20393.36	50651.23	1626.00	9.10	43053.51	829.86	50.92	19.20	154.48
141	17381.07	1852.96	59.40	19296.30	50004.32	1795.06	9.92	35584.57	806.30	39.77	19.51	148.56
142	13824.06	1370.53	31.53	16658.58	45706.69	1777.20	3.98	28514.64	866.84	36.86	16.44	150.59
143	17903.18	1855.49	67.54	18511.86	56701.27	2101.48	4.36	30361.23	1280.13	37.29	20.38	150.59
144	17384.43	1657.12	53.66	17718.86	57769.60	1718.13	3.41	28582.23	1059.65	37.78	18.74	305.02

Table A.3: Muskegon Lake core sediment geochemistry, continued

Depth (cm)	Al	K	Ti	Mg	Fe	Mn	As	Ca	P	Sr	Rb	Ba
145	15153.55	1230.93	27.91	17272.41	53107.51	1386.00	1.46	28179.31	744.46	31.85	15.96	142.84
146	15813.10	1554.54	43.86	16503.06	50120.52	1311.79	5.11	26972.05	763.78	34.37	18.01	147.84
147	18261.20	1732.44	54.68	18173.80	56159.00	1553.80	6.54	33932.40	837.28	36.96	19.36	151.06
148	16409.98	1395.27	41.75	17007.94	52894.50	1543.38	4.40	29628.72	709.59	31.00	16.01	128.90

Table A.3: Muskegon Lake core sediment geochemistry, continued

Depth (cm)	Sn	Cr	Co	Ni	Cu	Mo	Cd	Pb	Zn	U	Hg
2	0.65	46.16	6.22	19.89	48.09	1.47	2.17	50.61	136.90	0.78	0.26
3	0.49	53.35	8.52	22.45	54.53	0.86	2.16	59.49	215.56	0.78	0.12
4	0.45	45.99	5.32	19.57	51.50	0.90	2.21	60.11	159.21	0.71	0.15
5	0.44	49.56	7.95	21.89	53.03	0.93	1.93	57.05	152.26	0.77	0.14
6	0.47	47.90	7.82	19.98	51.65	1.02	1.84	55.78	158.31	0.78	0.14
7	0.38	49.62	8.04	20.11	53.19	1.03	1.81	57.85	156.12	0.78	0.14
8	0.51	49.63	6.16	20.08	54.67	1.50	2.20	61.38	151.14	0.93	0.12
9	0.42	54.23	5.91	24.58	53.91	1.46	2.31	65.40	182.47	0.95	0.14
11	0.57	72.78	8.49	23.61	59.96	1.89	2.57	83.63	440.64	1.10	0.16
12	0.80	86.23	8.05	23.54	61.72	2.83	3.01	99.59	245.10	1.09	0.19
13	0.82	98.24	5.12	23.08	61.92	3.59	4.12	119.98	259.10	1.20	0.22
14	0.81	135.08	7.63	26.29	71.27	3.69	4.62	135.87	345.76	1.00	0.40
15	0.92	146.47	7.93	28.44	75.68	4.38	4.94	144.57	287.67	1.07	0.25
16	1.45	138.44	4.37	25.69	67.44	5.63	5.35	142.90	266.06	1.09	0.26
17	1.38	138.11	4.51	25.32	68.26	5.34	5.44	145.38	265.00	1.06	0.27
18	1.20	140.02	4.47	25.55	66.97	4.39	5.42	150.02	279.46	0.90	0.25
19	1.42	148.69	4.79	25.97	66.53	4.25	5.60	146.34	363.94	0.86	0.24
21	1.83	236.10	8.10	38.87	78.89	6.52	7.72	158.42	457.51	0.92	0.27
22	2.44	339.94	8.87	44.17	82.46	8.78	10.53	174.36	517.39	0.79	0.35
23	2.82	412.00	8.33	39.34	90.47	9.40	10.64	184.36	481.73	0.91	0.35
24	2.61	392.38	7.71	34.37	87.17	9.29	9.57	176.15	421.01	0.88	0.34
25	2.34	398.68	7.66	35.65	81.30	10.84	10.29	176.17	447.18	0.90	0.36
26	2.44	413.80	7.21	35.71	79.43	9.89	10.97	172.21	431.30	0.78	0.40
27	3.20	452.04	6.89	38.63	88.17	10.17	11.63	183.39	565.07	0.83	0.43
28	2.72	403.05	6.33	33.79	80.95	9.57	10.41	170.19	703.07	0.72	0.42
29	2.57	434.85	7.70	37.23	83.62	9.91	10.57	173.64	412.30	0.79	0.39

Table A.3: Muskegon Lake core sediment geochemistry, continued

Depth (cm)	Sn	Cr	Co	Ni	Cu	Mo	Cd	Pb	Zn	U	Hg
31	2.68	429.07	7.32	37.90	82.73	6.62	9.91	169.42	355.78	0.80	0.38
32	3.37	506.42	7.88	42.70	94.55	3.08	10.96	179.10	407.78	0.75	0.47
33	3.90	506.31	7.70	43.60	94.14	2.75	10.98	183.43	373.67	0.80	0.48
34	3.86	489.80	7.86	44.58	90.52	3.01	10.81	188.17	435.45	0.89	0.46
35	3.84	456.40	8.08	38.94	86.19	2.44	10.82	188.40	402.33	0.76	0.49
36	3.38	489.04	7.88	37.58	96.50	1.34	12.34	191.63	397.64	0.72	0.54
37	2.86	466.66	7.23	34.10	91.29	1.27	11.42	175.79	357.71	0.76	0.57
38	2.33	405.51	6.88	31.64	83.72	1.19	10.48	165.68	343.04	0.71	0.56
39	2.53	404.96	6.00	29.09	83.01	0.98	11.04	154.32	422.61	0.64	0.54
41	2.89	417.96	6.78	29.09	86.24	1.33	18.02	188.15	597.87	0.87	0.53
42	2.54	406.90	7.52	26.84	95.56	1.30	18.42	178.42	395.24	0.85	0.45
43	3.21	353.17	7.13	25.16	93.54	1.10	18.22	171.39	528.23	0.82	0.47
44	2.50	355.18	7.21	26.34	98.45	1.33	16.82	193.14	410.65	0.94	0.47
45	3.76	299.11	5.21	22.05	86.72	1.19	12.76	176.26	335.46	0.78	0.47
46	1.61	320.67	6.12	28.00	95.36	1.20	14.18	201.37	351.70	0.94	0.52
47	1.59	329.18	5.06	23.18	95.73	1.42	14.36	177.15	308.48	1.16	0.42
48	1.50	295.50	5.76	22.83	100.71	1.62	14.99	192.24	330.10	1.07	0.42
49	1.73	260.09	5.64	21.77	92.96	1.77	16.37	192.89	311.81	0.97	0.43
51	2.53	165.85	7.43	23.94	78.90	1.87	14.38	199.60	335.26	1.05	0.45
52	2.76	192.14	6.85	23.19	93.88	1.39	18.07	271.69	383.25	0.90	0.44
53	3.10	171.14	6.76	22.40	77.26	1.88	17.22	195.86	306.34	0.98	0.44
54	2.66	185.36	6.78	24.75	80.24	1.47	17.82	197.21	389.09	0.89	0.43
55	3.62	227.63	5.91	21.77	85.04	1.71	18.17	167.92	429.40	0.91	0.45
56	3.64	200.70	4.20	22.74	70.52	2.12	17.84	148.04	383.60	1.06	0.44
57	2.78	160.28	4.02	22.25	66.61	1.26	17.42	137.71	269.53	0.83	0.44
58	2.58	157.86	2.14	15.61	54.91	1.14	15.96	119.32	198.58	0.68	0.37

Table A.3: Muskegon Lake core sediment geochemistry, continued

Depth (cm)	Sn	Cr	Co	Ni	Cu	Mo	Cd	Pb	Zn	U	Hg
59	4.12	183.27	3.42	18.71	72.95	1.64	20.15	144.52	238.64	0.83	0.50
61	5.36	220.38	6.51	22.55	87.72	1.28	18.81	184.00	283.09	0.87	0.54
62	3.17	312.32	6.72	21.94	89.36	1.32	17.85	213.36	288.64	0.89	0.59
63	3.50	362.52	7.08	22.50	84.01	1.53	15.40	298.25	615.35	0.94	0.74
64	4.16	335.51	6.00	20.63	82.80	1.45	14.16	324.12	251.78	0.84	0.61
65	3.62	344.39	5.53	18.14	71.89	1.06	10.91	221.93	227.14	0.70	0.51
66	3.42	290.94	6.74	20.84	74.22	0.98	11.65	224.28	258.11	0.75	0.54
67	2.56	201.19	7.36	24.41	78.32	0.59	14.90	268.07	296.39	0.78	0.58
68	1.87	186.13	8.11	27.41	78.85	0.51	15.97	292.33	335.37	0.78	0.62
69	1.57	189.83	7.18	25.88	83.24	0.44	19.31	308.33	296.19	0.71	0.64
71	1.23	115.65	8.49	27.02	79.56	0.48	16.54	262.12	299.08	0.79	0.69
72	1.26	96.08	8.07	27.32	90.99	0.40	17.27	262.47	462.96	0.76	0.74
73	0.92	99.21	8.50	29.48	92.85	0.41	17.23	268.63	271.80	0.90	0.73
74	0.94	97.21	8.44	32.62	97.60	0.51	17.58	268.24	1170.25	0.94	0.67
75	0.91	85.33	7.76	29.05	100.84	0.39	18.17	272.14	267.88	0.78	0.81
76	1.01	77.43	7.90	29.66	111.13	0.45	20.39	234.75	1275.87	0.88	0.81
77	1.46	49.83	7.79	30.93	161.23	0.64	24.12	134.82	434.65	1.04	0.77
78	1.05	44.13	8.45	30.83	107.46	0.70	5.12	108.33	305.45	0.97	0.71
79	1.31	38.38	7.95	32.65	95.80	0.80	3.06	107.39	258.36	0.93	0.63
81	0.75	30.28	7.82	25.16	49.89	0.45	1.61	77.73	183.92	0.79	0.56
82	0.61	32.67	8.83	25.80	50.92	0.75	1.86	78.68	178.25	1.02	0.53
83	0.78	31.54	8.99	25.08	51.29	0.55	1.67	101.37	187.47	0.86	0.71
84	0.54	36.12	8.92	23.32	57.31	0.48	1.63	118.05	213.58	0.93	0.75
85	0.43	34.55	9.25	22.02	50.13	0.43	1.42	99.94	151.82	0.86	0.61
86	0.35	33.64	9.43	23.07	43.38	0.41	1.24	72.11	156.56	0.96	0.49
87	0.25	33.07	9.24	22.99	33.71	0.36	1.17	57.80	146.36	0.87	0.33

Table A.3: Muskegon Lake core sediment geochemistry, continued

Depth (cm)	Sn	Cr	Co	Ni	Cu	Mo	Cd	Pb	Zn	U	Hg
88	0.27	32.73	9.57	23.29	33.49	0.52	1.09	61.07	486.59	0.87	0.34
89	0.29	32.57	10.07	23.74	29.67	0.35	0.95	57.10	151.08	0.88	0.30
91	0.18	34.93	11.16	27.51	27.38	0.37	0.74	45.39	115.52	1.00	0.17
92	0.15	31.40	9.96	25.28	25.97	0.39	0.75	47.65	488.04	0.88	0.19
93	0.21	36.91	11.62	27.06	24.50	0.36	0.66	45.03	113.31	1.00	0.15
94	0.15	30.26	10.26	24.38	21.25	0.26	0.55	38.75	143.46	0.88	0.14
95	0.10	27.88	8.73	21.85	17.63	0.29	0.44	31.04	99.56	0.77	0.13
96	0.09	14.92	3.81	11.13	9.46	0.17	0.26	17.36	78.65	0.44	0.08
97	0.10	20.60	6.38	16.60	12.47	0.22	0.32	21.27	65.84	0.64	0.06
98	0.10	21.20	6.42	17.09	12.86	0.21	0.30	20.98	58.93	0.65	0.07
99	0.10	21.80	6.46	17.58	13.25	0.20	0.28	20.69	52.02	0.67	0.09
101	0.12	27.98	9.71	24.63	17.53	0.31	0.33	25.04	100.57	0.80	0.08
102	0.09	27.17	9.26	23.59	17.37	0.30	0.33	23.93	68.30	0.74	0.08
103	0.09	27.79	9.36	23.46	17.20	0.32	0.36	23.63	71.12	0.74	0.08
104	0.09	28.42	9.46	23.34	17.02	0.34	0.39	23.32	73.94	0.75	0.08
105	0.13	31.86	9.87	23.95	17.05	0.34	0.38	21.77	73.12	0.81	0.07
106	0.11	31.68	9.96	24.75	17.86	0.32	0.40	22.18	82.12	0.84	0.12
107	0.11	32.27	9.83	24.61	17.38	0.28	0.39	20.97	100.67	0.80	0.09
108	0.09	27.42	8.80	20.63	14.89	0.24	0.35	18.58	64.48	0.68	0.10
109	0.09	28.53	8.18	19.98	14.40	0.26	0.37	17.05	111.14	0.72	0.07
111	0.09	20.93	5.74	12.23	8.76	0.24	0.29	9.69	54.50	0.51	0.08
112	0.08	17.67	5.90	10.73	8.35	0.24	0.29	8.98	62.02	0.41	0.08
113	0.04	13.12	3.05	5.54	4.11	0.17	0.17	4.93	25.33	0.29	0.05
114	0.04	14.97	3.83	7.53	5.59	0.23	0.21	5.99	31.83	0.35	0.03
115	0.11	17.01	5.03	8.87	6.18	0.23	0.25	6.35	63.38	0.40	0.05
116	0.04	18.74	6.10	11.08	8.00	0.29	0.33	6.37	320.81	0.48	0.03

Table A.3: Muskegon Lake core sediment geochemistry, continued

Depth (cm)	Sn	Cr	Co	Ni	Cu	Mo	Cd	Pb	Zn	U	Hg
117	0.07	25.21	8.01	15.61	10.62	0.44	0.38	8.39	57.15	0.71	0.04
118	0.08	28.39	8.59	18.47	12.31	0.41	0.37	8.96	207.02	0.76	0.05
119	0.04	26.05	8.28	17.79	12.34	0.32	0.36	9.03	55.75	0.71	0.05
121	0.05	28.99	9.04	20.27	13.76	0.38	0.39	9.73	61.76	0.82	0.05
122	0.06	31.93	9.80	22.75	15.18	0.43	0.41	10.43	67.77	0.92	0.05
123	0.09	31.93	9.94	22.68	15.37	0.41	0.41	10.50	87.86	0.87	0.05
124	0.06	30.30	10.06	22.33	14.96	0.34	0.39	10.32	67.22	0.78	0.05
125	0.09	30.17	9.85	21.97	14.48	0.31	0.37	10.09	78.54	0.76	0.05
126	0.09	33.05	10.06	24.11	14.87	0.32	0.39	10.04	148.81	0.80	0.05
127	0.05	31.62	10.45	23.84	15.65	0.29	0.45	10.40	83.27	0.82	0.05
128	0.06	25.28	9.14	19.38	12.95	0.26	0.36	8.84	282.95	0.66	0.04
129	0.07	36.20	11.53	26.40	18.08	0.37	0.46	11.86	77.01	0.96	0.04
130	0.10	28.70	9.59	23.54	17.35	0.33	0.37	10.77	85.10	0.72	0.04
131	0.04	28.97	9.27	20.81	14.31	0.43	0.34	9.68	59.01	0.79	0.05
132	0.08	30.37	10.12	22.37	15.20	0.43	0.39	10.43	188.41	0.82	0.05
133	0.09	29.62	9.23	20.96	14.83	0.32	0.34	9.76	80.90	0.77	0.05
134	0.02	30.91	9.81	22.17	15.15	0.30	0.38	10.23	68.15	0.81	0.04
135	0.04	25.45	9.19	19.74	14.10	0.21	0.34	9.85	66.47	0.68	0.04
136	0.06	28.46	9.23	21.14	15.35	0.33	0.39	10.04	60.60	0.77	0.04
137	0.07	27.60	8.71	19.30	14.07	0.44	0.39	9.26	60.72	0.72	0.05
138	0.04	30.10	9.57	21.53	15.24	0.52	0.39	9.90	57.73	0.85	0.04
139	0.04	31.08	10.22	22.65	15.96	0.56	0.40	10.26	63.49	0.88	0.06
141	0.06	30.70	9.90	21.79	14.92	0.45	0.39	10.02	67.61	0.80	0.04
142	0.04	28.05	9.62	21.25	17.37	0.38	0.40	9.92	193.39	0.70	0.04
143	0.08	30.83	9.70	21.67	14.39	0.34	0.40	10.13	67.42	0.74	0.04
144	0.07	29.19	9.65	21.52	14.29	0.29	0.38	9.73	545.71	0.73	0.04

Table A.3: Muskegon Lake core sediment geochemistry, continued

epth (cm)	Sn	Cr	Co	Ni	Cu	Mo	Cd	Pb	Zn	U	Hg
145	0.02	26.73	8.92	19.57	13.89	0.34	0.39	9.66	76.39	0.73	0.04
146	0.09	29.30	9.28	22.10	19.04	0.39	0.39	11.72	158.89	0.85	0.04
147	0.04	30.48	9.76	21.84	15.26	0.44	0.40	10.06	73.48	0.86	0.04
148	0.04	25.50	8.23	17.68	12.77	0.33	0.35	8.59	60.88	0.67	0.04

APPENDIX IV

RELATIVE ABUNDANCES OF COMMON MUSKEOGN LAKE DIATOMS

Table A.4: Relative abundances of common Muskegon Lake diatoms, continued

Depth (cm)	*Achnanthidium minutissimum* (Kützing) Czarnecki	*Actinocyclus normanii fo subsalsus* (Juhlin-Dannfelt) Hustedt	*Amphora pediculus* (Kützing) Grun	*Asterionella formosa* Hassal
2	5.24	0.00	4.56	0.00
4	1.44	0.16	1.60	0.00
8	2.25	0.00	5.78	0.00
12	0.17	0.83	3.51	0.00
16	0.00	0.66	1.64	0.00
21	0.16	0.48	0.97	0.00
24	0.00	0.83	0.50	0.33
28	0.00	1.70	1.54	0.62
32	0.30	3.14	0.30	0.00
36	0.47	1.57	1.10	0.63
40	0.00	0.92	1.69	0.00
44	0.97	3.72	0.16	0.65
48	0.42	1.94	0.28	3.74
52	0.49	0.16	0.98	5.69
56	0.63	0.00	0.00	1.09
61	0.16	0.00	1.55	0.47
64	0.36	0.18	0.90	0.90
68	0.32	0.00	0.97	0.00
72	0.16	0.00	0.82	0.00
76	0.30	0.00	1.95	0.00
81	1.12	0.00	0.96	0.00
84	0.80	0.00	3.19	0.00
88	0.00	0.00	2.37	0.00
92	0.31	0.00	10.42	0.00
96	0.17	0.00	1.50	0.00
101	0.00	0.00	2.93	0.00
104	0.46	0.00	2.28	0.00
108	0.16	0.00	2.09	0.00
112	0.17	0.17	3.80	0.00
116	0.00	0.00	5.37	0.00
121	0.16	0.00	1.48	0.00
124	0.17	0.00	2.65	0.00
128	0.00	0.00	2.28	0.00
132	0.00	0.00	2.79	0.00
136	0.00	0.00	1.18	0.00
139	0.00	0.00	8.70	0.00
144	0.00	0.00	3.81	0.00
148	0.00	0.00	0.96	0.00

Table A.4: Relative abundances of common Muskegon Lake diatoms, continued

Depth (cm)	*Aulacoseira ambigua* (Grunow) Simonsen	*Aulacoseira granulata* (Ehrenberg) Simonsen	*Aulacoseira granulata fo curvata*	*Aulacoseira granulata var angustissima* (O Müller) Simonsen
2	12.67	7.60	0.00	0.00
4	5.75	8.63	0.32	0.00
8	19.42	16.05	0.00	0.00
12	33.89	23.21	1.34	0.00
16	37.17	39.47	0.00	0.00
21	37.52	27.05	0.32	0.81
24	48.43	32.40	0.00	0.00
28	36.42	6.33	0.00	0.93
32	60.03	9.28	0.00	0.00
36	31.76	18.71	1.42	1.10
40	30.67	45.40	3.07	0.00
44	29.08	5.17	2.10	0.65
48	35.04	6.23	1.39	0.00
52	19.02	12.36	6.50	0.81
56	20.31	16.25	12.34	1.25
61	13.64	36.12	18.60	0.00
64	16.34	31.24	0.00	1.80
68	35.71	33.44	2.76	0.00
72	34.09	39.31	0.00	0.00
76	30.08	31.88	10.08	1.50
81	9.79	52.33	0.00	6.10
84	19.65	14.38	0.00	0.00
88	10.17	44.92	0.00	3.22
92	16.02	41.84	0.47	0.78
96	30.78	33.28	0.00	0.00
101	21.34	39.41	0.00	0.65
104	40.77	25.97	0.00	0.00
108	30.50	32.42	0.00	0.00
112	9.42	24.13	0.00	0.00
116	17.89	35.77	0.33	0.00
121	12.52	57.33	0.00	0.00
124	7.63	56.38	0.00	0.00
128	12.23	56.93	0.00	0.00
132	40.00	32.13	0.00	0.00
136	14.62	55.46	0.00	0.00
139	11.82	43.84	0.00	0.00
144	19.37	34.93	0.00	0.00
148	44.41	30.67	0.00	0.00

Table A.4: Relative abundances of common Muskegon Lake diatoms, continued

Depth (cm)	*Aulacoseira subartica* (O Müller) Haworth	*Cavinula scutelloides* (W Smith) Lange-Bertalot et Metzeltin	*Cocconeis neodiminuta* Krammer	*Cocconeis neothumensis* Krammer
2	0.00	0.00	0.00	0.51
4	0.00	0.16	0.32	0.32
8	0.00	0.00	0.48	0.00
12	0.33	0.50	0.17	0.33
16	1.32	0.16	0.00	0.00
21	0.48	0.00	0.00	0.00
24	0.33	0.00	0.00	0.00
28	1.54	0.00	0.00	0.00
32	0.45	0.15	0.30	0.00
36	0.47	0.16	0.31	0.16
40	1.53	0.15	0.31	0.31
44	2.75	0.00	0.16	0.00
48	0.97	0.00	0.28	0.00
52	0.00	0.00	0.33	0.00
56	1.56	0.00	0.00	0.00
61	0.00	0.00	0.16	0.00
64	1.44	0.18	0.18	0.00
68	0.00	0.16	0.16	0.00
72	2.77	0.33	0.82	0.16
76	0.75	0.15	0.90	0.45
81	0.00	0.16	0.00	0.00
84	0.64	0.16	0.80	0.48
88	0.34	0.51	0.85	0.34
92	0.47	1.09	1.71	0.62
96	0.00	4.99	1.83	1.00
101	0.16	3.42	1.63	0.81
104	0.00	2.51	1.37	0.46
108	0.00	2.57	0.80	1.12
112	1.16	20.99	5.95	6.78
116	0.33	4.88	4.07	3.74
121	0.00	5.27	3.29	1.32
124	0.00	4.64	2.32	2.82
128	0.33	6.69	1.63	2.94
132	0.00	4.59	2.62	0.66
136	0.00	5.71	2.69	2.18
139	0.00	4.93	3.78	2.96
144	0.00	11.26	3.64	3.15
148	0.00	3.19	2.40	2.40

Table A.4: Relative abundances of common Muskegon Lake diatoms, continued

Depth (cm)	*Cocconeis pediculus* Ehrenberg	*Cocconeis placentula* Ehrenberg	*Cocconeis placentula var euglypta* (Ehrenberg)	*Diatoma tenuis* Ag
2	0.68	2.53	4.56	0.17
4	1.12	2.56	2.72	0.16
8	2.41	4.17	4.33	0.00
12	1.17	0.83	0.33	0.33
16	0.33	0.99	0.33	0.16
21	0.32	0.64	0.64	2.58
24	0.17	0.50	0.00	3.31
28	0.31	1.08	0.31	6.17
32	0.45	1.80	0.45	2.84
36	0.31	2.20	1.10	1.73
40	0.31	0.61	0.46	0.15
44	0.48	4.36	0.81	3.39
48	0.28	1.80	0.55	2.35
52	0.81	4.55	2.60	1.79
56	0.31	2.50	2.81	1.56
61	0.00	3.41	1.09	0.16
64	0.18	1.97	1.62	0.36
68	1.14	2.92	2.92	0.00
72	0.98	4.73	3.59	0.00
76	0.60	4.96	4.06	0.00
81	0.64	8.35	6.90	0.00
84	0.96	19.65	14.38	0.48
88	2.03	9.66	8.47	0.00
92	0.47	3.27	1.71	0.31
96	0.17	2.00	1.83	0.00
101	0.81	4.40	1.14	0.00
104	0.91	3.64	1.59	0.00
108	0.80	2.89	0.96	0.00
112	0.00	0.99	0.33	0.00
116	0.00	0.81	0.33	0.00
121	0.00	0.49	0.00	0.00
124	0.00	1.16	0.17	0.00
128	0.00	0.33	0.16	0.00
132	0.00	0.66	0.00	0.00
136	0.17	1.34	0.50	0.00
139	0.00	0.49	0.00	0.00
144	0.00	0.33	0.00	0.00
148	0.16	0.48	0.00	0.00

Table A.4: Relative abundances of common Muskegon Lake diatoms, continued

Depth (cm)	*Fragilaria capucina* Desmazieres	*Fragilaria capucina var_ gracilis* (Østr) Hust	*Fragilaria crotonensis* Kitton	*Opephora martyi* Hérib
2	15.71	7.43	0.00	0.00
4	30.51	0.00	3.51	0.80
8	23.27	0.64	1.93	0.00
12	4.67	0.00	5.34	1.34
16	1.97	0.16	2.63	0.33
21	5.96	0.16	2.90	0.00
24	1.65	0.00	0.83	0.00
28	1.54	0.00	8.02	0.00
32	0.45	0.00	2.54	0.15
36	8.49	0.00	1.73	0.00
40	0.00	0.00	2.91	0.61
44	3.39	0.00	7.59	0.00
48	2.22	0.00	4.02	0.42
52	0.65	0.33	10.41	0.00
56	4.84	0.00	9.38	0.00
61	5.74	2.17	7.60	0.00
64	3.77	0.18	5.03	0.00
68	0.00	0.00	2.60	1.62
72	0.65	0.00	0.00	3.59
76	0.00	0.30	0.15	0.60
81	0.80	0.16	0.64	0.32
84	0.64	0.48	2.24	1.44
88	0.00	0.00	0.34	1.53
92	0.93	0.00	0.00	2.49
96	0.00	0.00	0.00	5.49
101	0.16	0.00	0.00	5.37
104	0.00	0.00	0.00	6.83
108	1.28	0.00	0.16	3.53
112	0.00	0.00	0.00	4.96
116	0.00	0.49	0.00	6.02
121	0.00	0.00	0.00	4.45
124	0.00	0.17	0.00	6.97
128	0.00	0.00	0.00	3.26
132	0.00	0.00	0.00	5.57
136	0.00	0.00	0.00	6.22
139	0.00	0.00	0.00	5.09
144	0.00	0.00	0.00	9.77
148	0.00	0.00	0.16	4.31

Table A.4: Relative abundances of common Muskegon Lake diatoms, continued

Depth (cm)	*Planothidium frequentissimum* (Lange-Bertalot) Lange-Bertalot	*Pseudostaurosira brevistriata* (Grun in VH) Williams & Round	*Staurosira construens* (Ehr) Williams & Round
2	0.17	14.53	5.07
4	1.12	14.06	8.63
8	0.32	7.22	0.64
12	0.83	7.18	1.84
16	0.49	0.00	3.29
21	0.00	1.45	0.16
24	0.33	0.17	0.00
28	1.54	0.93	0.00
32	0.45	0.15	0.15
36	0.63	3.14	0.00
40	0.46	4.60	0.31
44	0.16	0.00	0.32
48	0.14	0.00	0.00
52	0.98	0.00	2.76
56	0.47	0.00	3.75
61	0.47	0.62	0.00
64	0.36	0.00	1.80
68	0.97	2.11	0.00
72	0.16	0.33	0.16
76	0.90	0.15	0.30
81	0.48	0.00	0.00
84	1.76	0.00	1.76
88	0.68	0.17	2.88
92	0.93	0.16	4.04
96	0.83	2.50	1.66
101	0.81	0.81	3.42
104	0.46	3.19	0.23
108	2.57	1.61	1.77
112	2.98	1.32	1.98
116	1.30	0.81	2.28
121	0.16	2.31	0.33
124	0.50	0.50	1.16
128	0.82	0.65	0.49
132	0.82	0.98	0.00
136	0.00	1.68	0.84
139	0.66	1.64	0.82
144	0.00	2.65	0.66
148	0.32	1.44	0.00

Table A.4: Relative abundances of common Muskegon Lake diatoms, continued

Depth (cm)	*Staurosirella pinnata* (Her) Williams & Round	*Stephanodiscus hantzschii* Grun	*Stephanodiscus minutulus* (Kütz) Cleve & Moller	*Stephanodiscus niagarae* Her
2	1.18	0.51	1.01	0.17
4	1.12	0.64	1.12	0.48
8	1.12	0.48	0.64	0.48
12	0.17	0.67	1.17	2.17
16	0.00	0.66	2.47	0.33
21	0.97	7.09	2.90	0.16
24	0.00	2.48	3.31	0.33
28	0.31	6.64	12.50	0.93
32	0.00	3.74	4.64	1.05
36	0.16	6.92	5.82	0.94
40	0.00	0.31	0.31	1.23
44	0.00	13.73	13.09	0.00
48	0.00	12.88	18.56	0.55
52	0.00	3.74	15.61	0.65
56	0.78	3.44	7.81	2.50
61	0.62	0.47	2.64	0.62
64	0.90	0.54	24.78	0.18
68	0.65	0.49	1.46	1.14
72	0.00	0.65	1.14	1.79
76	0.45	0.30	1.65	1.50
81	0.64	0.64	1.61	0.80
84	1.28	0.16	1.12	0.80
88	0.68	0.51	0.51	1.36
92	4.04	0.00	0.00	1.40
96	1.66	0.00	0.00	1.16
101	2.12	0.00	0.00	1.79
104	1.82	0.00	0.00	2.05
108	1.93	0.00	0.00	5.46
112	3.47	0.00	0.00	0.99
116	2.28	0.00	0.00	4.55
121	0.82	0.00	0.00	2.64
124	0.66	0.00	0.00	2.65
128	0.82	0.00	0.00	2.45
132	1.15	0.00	0.00	0.33
136	0.84	0.00	0.17	0.84
139	1.64	0.16	0.00	3.12
144	0.99	0.00	0.17	0.83
148	0.32	0.00	0.00	2.08

APPENDIX V

MUSKEGON LAKE FOSSIL DIATOM TAXA

Table A.5: Muskegon Lake fossil diatom data

Achnanthes
A. conspicua Mayer

Achnanthidium
A. exiguum (Grunow) Czarnecki
A. minutissimum (Kützing) Czarnecki

Actinocyclus
A. normanii fo. subsalsus (Juhlin-Dannfelt) Hustedt

Amphora
A. copulata (Kützing) Schoeman et Archibald
A. ovalis (Kützing) Kützing
A. pediculus (Kützing) Grun.
A. veneta Kützing

Aneumastus
A. tusculus (Ehrenberg) Mann et Stickle

Asterionella
A. formosa Hassal

Aulacoseira
A. ambigua (Grunow) Simonsen
A. ambigua fo curvata
A. granulata fo curvata
A. granulata fo curvata
A. granulata (Ehrenberg) Simonsen
A. granulata fo curvata
A. granulata fo. curvata
A. granulata var. angustissima (O. Müller) Simonsen
A. italica (Ehrenberg) Simonsen
A. subartica (O. Müller) Haworth

Caloneis
C. bacillum (Grunow) Cleve

Campylodiscus
C. echineis Ehrenberg

Table A.5: Muskegon Lake fossil diatom data, continued

Cavinula
C. cocconeiformis (Gregory ex Greville) Mann et Stickle
C. lacustris (Gregory) Mann et Stickle
C. pseudoscutiformis (Grunow ex Schmidt) Mann et Stickle
C. scutelloides (W. Smith) Lange-Bertalot et Metzeltin

Cocconeis
C. disculus (Schumann) Cleve
C. neodiminuta Krammer
C. neothumensis Krammer
C. pediculus Ehrenberg
C. placentula Ehrenberg
C. placentula var. euglypta (Ehrenberg) Cleve

Craticula
C. halophila (Grunow in Van Heurck) Mann

Ctenophora
C. pulchella (Ralfs ex Kützing) Williams & Round

Cyclostephanos
C. dubius (Fricke in A. Schmidt) Round

Cyclotella
C. atomus Hustedt
C. bodanica Grunow
C. distinguenda Hustedt
C. meneghiniana Kützing
C. michiganiana Skvortzow
C. ocellata Pantosek
C. pseudostelligera Hustedt
C. stelligera (Cleve et Grunow) Van Heurck

Cymatopleura
C. solea var. apiculata (W. Sm.) Ralfs

Cymbella
C. affinis Kütz.
C. amphicephala Naeg. ex Kütz.

Table A.5: Muskegon Lake fossil diatom data, continued

C. caespitosa Brun
C. cistula (Ehr.) Kirchn.
C. delicatula Kütz.
C. ehrenbergii Kütz.
C. helvetica Kütz.
C. incerta (Grun.) Cl.
C. lata Grun.
C. leptoceros (Ehr.) Kütz.
C. mexicana (Ehr.) Cl.
C. subcuspidata Kram.

Cymbellonitzschia
C. diluviana Hust.

Diatoma
D. tenuis Ag.
D. vulgaris Bory

Diploneis
D. elliptica (Kütz.) Cl.

Encyonema
E. prostratum (Berk.) Kutz.
E. silesiacum (Bleisch in Rabenhorst) Mann
E. microcephala (Grun.) Kram.

Entomoneis
E. ornata (Bail.) Reim.

Epithemia
E. adnata (Kütz.) Bréb.
E. sorex Kütz.
E. turgida (Ehr.) Kütz.
E. turgida var. granulata (Ehr.) Hust.
Eucocconeis
E. alpestris (Brun) Lange-Bertalot
Eunotia
E. arcus Ehr.

Table A.5: Muskegon Lake fossil diatom data, continued

E. faba (Ehr.) Grun. in V. H.
E. formica Ehr.
E. minor (Kütz.) Grun.
E. spp.

Fragilaria
F. capucina Desmazieres
F. capucina var. gracilis (Østr.) Hust.
F. capucina var. mesolepta Rabh.
F. crotonensis Kitton
F. vaucheriae (Kütz.) Peters.

Frustulia
F. vulgaris (Thwaites) DeT.

Geissleria
G. decussis (Hustedt) Lange-Bertalot et Metzeltin

Gomphoneis
G. eriense (Grun.) Skv. & Meyer
G. acuminatum Ehr.
G. angustum Ag.
G. clavatum Ehr.
G. gracile Ehr. emend. V. H.
G. grovei M. Schmidt
G. grovei var. herrmanniana (Palik) Kociolek, Yang et Stoermer
G. minutum (Ag.) Ag.
G. olivaceum (Lyngb.) Kütz.
G. parvulum (Kütz.) Kütz.
G. truncatum Ehr.
Gyrosigma
G. acuminatum (Kütz.) Rabh.

Hantzschia
H. amphioxys (Ehr.) Grun.

Hippodonta
H. capitata (Ehrenberg) Lange-Bertalot, Metzeltin et Witkowski
H. hungarica (Grunow) Lange-Bertalot, Metzeltin et Witkowski

Table A.5: Muskegon Lake fossil diatom data, continued

Karayevia
K. clevei Grun. in Cl. et Grun.
Lemnicola
L. hungarica (Grun.) Round and Basson

Luticola
L. mutica (Kutz.) Mann

Mastogloia
M. smithii Thw.

Melosira
M. undulata var. normanii Arnott
M. varians Ag.

Meridion
M. circulare (Grev.) Ag.
M. circulare var. constrictum (Ralfs) V. H.

Navicula
N. antonii Lange Bertalot in Rumrich
N. capitatoradiata Germain
N. cari Ehr.
N. cincta (Ehr.) Ralfs
N. cryptocephala Kütz.
N. cryptotenella L.B. in Kramm. & L.-B.
N. gregaria Donk.
N. integra (W. Sm.) Ralfs
N. laterostrata Hust.
N. peregrina (Ehr.) Kutz.
N. pseudoanglica var. signata (Hustedt) Lange-Bertalot
N. radiosa Kütz.
N. recens Lange-Bert.
N. reichardtiana Lange-Bert.
N. reinhardtii (Grun.) Grun.
N. rostellata Kützing

Table A.5: Muskegon Lake fossil diatom data, continued

N. tripunctata (O. F. Müll.) Bory
N. trivialis Lange-Bert.
N. utermoehlii Hust.
N. veneta Kütz.
N. viridula (Kütz.) Kütz. emend. V. H.

Neidium
N. ampliatum (Ehr.) Kramm.

Nitzschia
N. amphibia Grun.
N. dissipata (Kütz.) Grun.
N. linearis (Ag. ex W. Sm.) W. Sm.
N. palea (Kütz.) W. Sm.

Opephora
O. martyi Hérib.

Pinnularia
P. acrosphaeria (Bréb.) W. Sm.

Placoneis
P. clementis (Grun) Cox
P. elginensis (Greg.) Cox
P. exigua (Greg.) Mereschk.
P. explanata (Hust.) Cox
P. gastrum (Ehr.) Meresch.
P. placentula (Ehr.) Hienzerling
P. pseudoanglica (Lange-Bertalot) Cox

Plagiotropis
P. lepidoptera (Greg.) Kuntze
P. lepidoptera var. proboscidea (Cl.) Reim.

Planothidium
P. delicatulum (Kützing) Round et Bukhtiyarova
P. frequentissimum (Lange-Bertalot) Lange-Bertalot
P. lanceolatum

Psammothidium

Table A.5: Muskegon Lake fossil diatom data, continued

P. subatomoides (Hüst.) Bukht. et Round

Pseudostaurosira
P. brevistriata (Grun. in V.H.) Williams & Round
Reimeria
R. sinuata (Greg.) Kociolek & Stoermer

Rhoicosphenia
R. abbreviata (Agardh) Lange-Bertalot

Rhopalodia
R. gibba (Ehr.) O. Müll.
R. gibberula (Ehr.) O. Müll.

Sellaphora
S. mutata (Krass.) Lange-Bertalot
S. pupula (Kütz.) Meresckowsky
S. seminulum (Grun.) Mann
Stauroneis
S. smithii Grun.

Staurosira
S. construens (Ehr.) Williams & Round
S. construens fo. binodis (Ehr.) Hust.
S. construens var. venter

Staurosirella
S. lapponica (Grun. In V.H.) Williams & Round
S. leptostauron (Grun. In V.H.) Williams & Round
S. leptostauron var. dubia (Grunow) Edlund
S. leptostauron. Ehr.
S. pinnata (Her.) Williams & Round

Stephanodiscus
S. hantzschii fo. tenuis (Hust.) Hakan. & Stoerm.
S. hantzschii Grun.
S. minutulus (Kütz.) Cleve & Moller
S. niagarae Ehr.
S. sp.

Table A.5: Muskegon Lake fossil diatom data, continued

Surirella
S. angusta Kütz.
S. linearis var. constricta Grun.
S. minuta Bréb.

Synedra
S. acus var. angustissima Grun.
S. capitata Ehr.
S. ulna (Nitz.) Ehr.

Tabellaria
T. fenestrata (Lyngb.) Kütz.
T. tabulata (C. A. Ag.) Snoeijs

Thalassiosira
T. weissflogii (Grun.) Fryxell & Hasle

APPENDIX VI

HABITAT AND ECOLOGICAL PREFERENCE OF COMMON MUSKEOGN LAKE DIATOMS

Table A.6: Habitat and ecological preference of commonMuskegon Lake fossil diatoms

Species	Habitat	Ecology preferences	Reference
Achnanthidium minutissimum	Benthic		Fritz et al. (1993)
		Epiphytic	Panizzo et al. (2008)
		Generalist species	Chen et al. (2008)
		Eurytopic, periphytic	Anderson (1997)
Actinocyclus normanii fo subsalsus	Planktonic	Eutrophic and alkaliphilous	van Dam et al. (1994)
		Typically associated with human disturbance in Europe's modern eutrophicated rivers, but found in much older riverine sediment in this study	Witkowski et al. (2004)
		Large, centric, sometimes nuisance species	Leitao and Leglize (2000)
Amphora pediculus	Benthic		Taylor et al. (2006)
		Oligio-mesotrophic and alkaliphilous	van Dam et al. (1994)
		pH > 8.0	Chen et al. (2008)
		Epiphyte	Bennion and Appleby (1999)
Asterionella formosa	Planktonic	Open water	Taylor et al. (2006); Chen et al. (2008)
		Responds to changing nutrient conditions	Wolin and Stoermer (2005)
		Mesotrohic total phosphorus optima	Hall and Smol (1992); Reavie et al. (1995a)
		Sensitive to changing light conditions	Leitao and Leglize (2000)
		Introduced by salmon in New Zealand lakes, morphospecies considered cosmopolitan, seasonally dominant in eutrophic lakes	Vanormelingen et al. (2008)

Table A.6: Habitat and ecological preference of commonMuskegon Lake fossil diatoms, continued

Species	Habitat	Ecology preferences	Reference
Aulacoseira ambigua	Planktonic	Open water	Taylor et al. (2006); Chen et al. (2008)
		Require turbulent conditions because they rely on mixing to remain suspended in water column	Wolin and Stoermer (2005)
		High Si requirements for growth	Fritz et al. (1993); Kilham et al. (1986)
		Present after deforestation	Bradbury et al. (2002)
		Oligotrophic to mesotrophic	Ramstack et al. (2003); Reavie et al. (1995a); Reavie and Smol (2001)
		Presence unrelated to trophic status	Brugram (1979)
Aulacoseira granulata	Planktonic	Oligohalobous, alkaliphilous, limnophilous, planktonic	Hustedt (1957); Ehrlich (1973)
		Eutrophic, low silicate	Brugam (1983); Kilham et al. (1986)
		Resitant to human impacts	Witkowski et al. (2004)
Aulacoseira granulata fo curvata	Planktonic	Eutrophic and alkaliphilous	van Dam et al. (1994)
Aulacoseira granulata var angustissima	Planktonic	Eutrophic and alkaliphilous	van Dam et al. (1994)
Aulacoseira subartica	Planktonic		Taylor et al. (2006)
		Associated with oligio-mesotrophic conditions	van Dam et al. (1994)
		Associated with increasing nutrient rich conditions	Dokulil and Teubner (2005)
		Responds to changing nutrient conditions	Wolin and Stoermer (2005)

Table A.6: Habitat and ecological preference of commonMuskegon Lake fossil diatoms, continued

Species	Habitat	Ecology preferences	Reference
Cavinula scutelloides (also Navicula scutelloides)	Benthic	Alkalibionic (exclusively at pH >7) and eutrophic	van Dam et al. (1994)
Cocconeis neodiminuta	Benthic		
Cocconeis neothumensis	Benthic	Eutrophic	van Dam et al. (1994)
Cocconeis pediculus	Benthic	Hypereutrophic, alkaliphic and benthic	van Dam et al. (1994); Chen et al. (2008)
Cocconeis placentula	Benthic	Epiphytic, freshwater, alkaliphilous with pH values 7.5-8.0	Ehrlich (1973)
		Oligosaprobic, oligohalobous "indifferent", alkaliphilous, cosmopolitan diatom	Foged (1980)
		Epiphytic taxa	Bennion and Appleby (1999)
		Eutrophic species, with maximum development in utrient rich and very well oxygenated waters	Steinberg and Schiefele (1988)
Cocconeis placentula var euglypta	Benthic		
Diatoma tenuis Ag		High chloride optima (81.8 mg/l)	Ramstack et al. (2003)
		Common in eutrophied areas of Great Lakes and indicative of high conservative ion loading	Wolin and Stoermer (2005)
		Hypereutrophic, alkaliphic and benthic	van Dam et al. (1994); Chen et al. (2008)

Table A.6: Habitat and ecological preference of commonMuskegon Lake fossil diatoms, continued

Species	Habitat	Ecology preferences	Reference
Fragilaria capucina	Benthic	Small, tychoplanktonic (adaptive to dynamic thermal characteristics), require high light penetration (shallow water)	Panizzo et al. (2008)
		High chloride optima (63.2 mg/l)	Ramstack et al. (2003)
Fragilaria capucina var gracilis	Benthic	Small, tychoplanktonic (adaptive to dynamic thermal characteristics)	Panizzo et al. (2008)
Fragilaria crotonensis	Benthic	Small, tychoplanktonic (adaptive to dynamic thermal characteristics)	Panizzo et al. (2008)
		Respond to changing nutrient conditions	Wolin and Stoermer (2005)
		Rising nitrate concentrations	Wolin and Stoermer (2005)
		associated with nutrient enrichment in in lakes of low to moderate alkalinity	Bradbury (1975); Brugam (1979)
	Planktonic		Fritz et al. 1993
Opephora martyi	Benthic		
Planothidium frequentissimum	Benthic		
Pseudostaurosira brevistriata	Benthic	Benthic	Kenney et al. (2002); Taylor et al. (2006)
Staurosira construens	Planktonic	Grows well in short growing season, meso-eutrophic, competitive, common in low nutrient arctic lakes	Cremer et al. (2001)
Staurosirella pinnata	Benthic	Benthic and epiphytic	Taylor et al. (2006)
		Alkaliphic	Chen et al. (2008)

Table A.6: Habitat and ecological preference of commonMuskegon Lake fossil diatoms, continued

Species	Habitat	Ecology preferences	Reference
Stephanodiscus spp.	Planktonic	Large centric, mesotrophic	Morabito et al. (2002)
		High phosphorus requirements	Bradbury (1975); Bennion et al. (1996)
		Increases in this diatom in a lake in northeastern Minnesota was coincident with increased inputs of domestic and municipal wastes	Bradbury (1975)
		High total phosphorus optima	Fritz et al. (1993); Ramstack et al. (2003)
Stephanodiscus hantzschii	Planktonic	Planktonic, hypertrophic	Taylor et al. (2006)
		Small centric taxa, in highly eutrophic conditions	Bennion and Appleby (1999)
		Resistant to human impact	Witkowski et at. (2004)
Stephanodiscus minutulus	Planktonic		
Stephanodiscus niagarae	Planktonic		Fritz et al. (1993)