# NATURAL LANDSCAPE DRIVERS OF TOTAL PHOSPHORUS CONCENTRATIONS IN MICHIGAN LAKES

By

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#### ABSTRACT

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Paleolimnological reconstructions have demonstrated an inherent variability in phosphorus concentrations among lakes prior to anthropogenic landscape disturbances. However, the natural drivers of this variability are less frequently studied than agricultural or urban phosphorus loading to freshwaters, and are not as well understood. Therefore, this research examined pre-settlement phosphorus concentrations of 48 inland lakes in Michigan in conjunction with pre-settlement land cover data, hydrogeomorphic features, and geographic variables, to better understand these natural landscape drivers. This multi-step process involved an analysis of diatom assemblages from lacustrine sediment cores, relative dating of core samples using *Ambrosia* pollen, a native land cover analysis in GIS, and finally, regression modeling of the data. The results of OLS regression modelling suggest that the variability in pre-settlement phosphorus concentrations of my dataset is best explained by native wetland cover. Moreover, native wetlands at a local (100 m buffer) scale had a stronger relationship with pre-settlement phosphorus concentrations than native wetlands within the entire lake catchment. The strength of this relationship improved further when wetlands were modeled with maximum lake depth. My analyses show that as the amount of pre-settlement phosphorus concentrations in a lake increased as wetlands near the lake shore increased. Furthermore, as lake depth decreases, the effect of wetlands on lake phosphorus concentrations increases. However, because my study was only able to explain 34% of the variability in pre-settlement phosphorus concentrations, further research is warranted to better understand this natural variability.

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#### **1. INTRODUCTION**

#### **1.1 Cultural Eutrophication and Phosphorus Regulations**

Phosphorus is a naturally occurring nutrient required by all primary producers for cell maintenance (Harper 1992; Smith and Smith 2001; Wetzel 2001). However, excess phosphorus loading from anthropogenic sources, termed cultural eutrophication, is consistently ranked by the United States Environmental Protection Agency (EPA) as one of the leading causes of toxic algal blooms, oxygen depletion, and an attendant loss of aquatic biodiversity (USEPA 2000; Herlihy et al. 2013). In addition to these direct ecosystem impairments, degraded water resources increase the costs of water treatment, reduce aesthetic tourism and property values, and place unneeded strain on commercial and/or recreational fisheries (Seppälä et al. 2004).

The role of phosphorus as a limiting nutrient in freshwater lakes was first recognized by limnologists in the mid-20<sup>th</sup> century (Hutchinson 1973; Schindler 1977; Juggins et al. 2013). In an effort to correct these impairments and degradations of lakes and streams, effluents are now limited from discrete conveyances under the jurisdiction of the National Pollutant Discharge Elimination System (NPDES) program, a cornerstone of the 1972 Clean Water Act (CWA) (Clean Water Act 1972). The NPDES regulation has reduced the impact of point source pollutant discharges, but has only partially lessened the effects of cultural eutrophication (Svendsen et al. 1995). In fact, an analysis of a subset of lakes from the National Eutrophication Survey (NES) indicated the number of eutrophic lakes actually increased from 16% in 1972 to 26.9% of lakes in 2007 (USEPA 2009). Indisputably, point source pollution control alone is not enough to achieve nutrient reduction goals for water bodies (Svendsen et al. 1995).

It is now widely recognized that the majority of cultural eutrophication in freshwater lakes has emanated from agricultural and urban runoff contributions (Carpenter et al. 1998;

Reynolds and Davies 2001). Unfortunately, these nonpoint source inputs of phosphorus from overland flow are inherently more difficult to monitor and regulate in comparison to point source pollutants (Carpenter et al. 1998). Furthermore, because of the decentralized nature of nonpoint phosphorus sources, it is challenging to distinguish between natural and anthropogenic inputs. Moreover, due to natural variations in ambient levels of total phosphorus, nutrient management strategies now acknowledge that standardized nutrient regulations may not be appropriate to achieve water restoration goals (USEPA 2000).

Rather, current regulations and management efforts focus on establishing numeric nutrient criteria (enrichment limits, typically for phosphorus and nitrogen) for geographic regions based on water parameters representative of 'pristine' waters (USEPA 2000). Several methods have been developed to identify lake phosphorus concentrations as they were in undisturbed conditions. These methods include the use of reference lakes, expert judgments, and paleolimnological reconstructions of lake conditions prior to Euro-American settlement in the early to mid-19<sup>th</sup> century (USEPA 2000; Herlihy et al. 2013). Reference lakes are "minimally impacted" sites representative of undisturbed water quality conditions in a given region. While this method has its merits, it has been argued that anthropogenic impacts are so pervasive and ubiquitous in the environment that truly undisturbed aquatic systems no longer exist (Herlihy et al. 2013).

Therefore, paleolimnological reconstructions of pre-settlement lake conditions are a valuable alternative approach to establishing numeric phosphorus criteria in the absence of undisturbed sites (Dixit et al. 1999; Herlihy et al. 2013). This method employs the use of fossil diatom assemblages from lake sediment cores as proxies to infer past water conditions, and is particularly well-suited to moist-temperate environments where phosphorus is the limiting

nutrient and lake sediments are well preserved (Cohen 2003). Furthermore, the coupling of presettlement, diatom-reconstructed phosphorus concentrations with pre-settlement land cover and hydrogeomorphic (HGM) data can provide a better understanding to the natural sources of phosphorus loading.

#### **1.2 Natural Variations in Lake Phosphorus Concentrations**

It is well known that lake water quality is reflective of the watershed characteristics that control the transport of materials transferred by surface runoff (Soranno et al. 1996). The relationship between lake nutrients and the gradient of urban and/or agricultural intensities in a watershed has been clearly demonstrated over the last few decades (Benoit and Fizaine 1999; Berka et al. 2001; Cuffney et al. 2000). There has been comparatively little research examining the natural variations in phosphorus concentrations because very few long-term datasets exist and it is rare to examine an undisturbed watershed today (Fritz et al. 1993). In many cases, the paleolimnological record is the only means of documenting pre-settlement trends in water quality data within the ecological context needed to separate natural lake variability from anthropogenic stress (Smol 2002). Exploring the spatial complexity of natural phosphorus variation, reflective of differences in vegetation and hydrogeomorphology within a region, will contribute to a better understanding of the combined effects of the surrounding landscape on nutrient and water inflow, a precursor to the establishment of more accurate and attainable nutrient criteria as management goals (Zhang et al. 2012; Herlihy et al. 2013).

These complex relationships are best approached from a landscape limnology perspective which is defined as, "the spatially explicit study of lakes, streams, and wetlands as they interact with freshwater, terrestrial, and human landscapes to determine the effects of pattern on ecosystem processes across temporal and spatial scales" (Soranno et al. 2010, p. 442). In other

words, the effects of a landscape on surface water chemistry should not be presumed consistent among regions, but rather, these relationships may be influenced by cross-scale interactions (Fergus et al. 2011). Additionally, water chemistry, in a landscape limnology context, requires the incorporation of hydrogeomorphic (HGM) features such as basin morphometry and hydrologic connectivity (Bremigan and Soranno 2008; Fergus et al. 2011).

Therefore, many management strategies rely on landscape-based ecological classifications to improve the tractability of thousands of lakes within the United States (Soranno et al. 2010). For example, the EPA's Level III Ecoregions divides Michigan into five different ecological units (Figure 1.1), each characterized by local interactions of vegetation, climate, soils





and topography (USEPA 2013a). Each unit in Michigan denotes a group of terrestrial and aquatic ecosystems that are assumed to respond similarly to management actions because of comparable landscape characteristics such as forest species composition, remnant glacial landforms, and land use capabilities.

#### **1.3 Problem Statement**

Although Michigan has approximately 11,000 inland lakes, there is little understanding of how these lakes have changed since Euro-American settlement in the mid-1800s. The majority of lake phosphorus research is associated with modeling the effects of human land use/land use change activities to predict and explain nutrient patterns in lakes *after* Euro-American settlement. Mostly, these studies focus on individual lakes and specific nonpoint sources such as fertilizer runoff or septic systems leakages. Therefore, much of our understanding of lake ecology is derived from systems already highly modified by anthropogenic activities. There are comparatively few studies that examine the variability in pre-settlement lake phosphorus concentrations in the context of natural landscape variables in the United States.

However, as demands on aquatic ecosystems continue to escalate, the knowledge of these baseline conditions and natural variability in total phosphorus concentrations is increasingly recognized of prime importance for effective lake ecosystem management (Smol 1992). Although long-term records of previous lake conditions are often unavailable, proxy indicators can be used to reconstruct these pre-settlement patterns and provide insight into natural drivers of ecosystem functioning (Smol 1992; Reavie et al. 1995). This reference information can be used in informed ecosystem management to define restoration goals, determine restoration potential, and evaluate the success of restoration efforts (White and Walker 1997; Rhemtulla and Mlandenoff 2007; Stein et al. 2010).

#### **1.4 Research Objectives**

The objective of this paper is to measure the pre-Euro-American landscape-lake phosphorus relationships Michigan. I hope that this knowledge can be applied to better-informed management plans and policy decisions that regulate anthropogenic phosphorus loading to lakes. I used fossil diatom assemblages, recovered from the bottoms of short sediment cores taken from 48 lakes distributed throughout Michigan, as paleoindicators of pre-settlement lake phosphorus concentrations. Native vegetation data (obtained from the Michigan Natural Features Inventory's compilation of 19<sup>th</sup> century Public Land Office surveys (Comer et al. 1995)) and hydrogeomorphic lake features were included as predictor variables in a regression model that describes the variability of diatom-inferred phosphorus concentrations among the 48 lakes.

Similar research has been done before by others, however prior work tends to be sitespecific, reconstructing pre-settlement changes of a single lake based on diatom assemblages from several stratigraphic levels of a sediment core, dated using radioisotopes <sup>210</sup>Pb and <sup>137</sup>Cs. Unlike these intensive studies of individual lakes, the aim of this thesis is to assess broad spatial patterns in landscape composition and correlated trends in lake phosphorus concentrations in Michigan.

My research will therefore address the following questions:

- Which land cover features are the most effective predictors (higher statistical explanatory power) of phosphorus concentrations in lakes, prior to Euro-American settlement?
- 2. Are land cover metrics combined with hydrogeomorphic features and/or geographic variables better predictors of phosphorus variability than land cover alone?

3. Can land cover within 100 m of the lake shore better account for the variability in pre-settlement phosphorus concentrations than land cover for the entire lake catchment?

#### 1.5 Outline of Thesis

This thesis is presented in six chapters. Chapter One introduces the topic and presents the research questions in more depth. Chapter Two is a literature review that primarily includes a discussion of existing studies of relevance to this research. The main topics presented in the chapter are eutrophication, the phosphorus cycle, management and regulations, and landscape limnology. Chapter Three includes a description of the study lakes and relevant physical parameters within the state of Michigan. This chapter also includes an outline of the site selection process. The fourth chapter presents a synopsis of the materials and methods used in this thesis. This work was conducted in several stages both in the field and in the laboratory. First, an overview of the sediment core extraction and the diatom analysis from 2007 is presented, followed by a description of the pollen analysis, GIS methods, and lastly the statistical modeling used to explore the relationship between phosphorus concentrations and landscape variables. Chapter Five presents the results of the methods used, while Chapter Six discusses the major findings and also discusses further research needed if we are to understand the complexity in landscape-water interactions. Finally, a conclusion for the entire study is presented in Chapter Seven.

#### **2. LITERATURE REVIEW**

This chapter will review the major scientific findings related to phosphorus and the physiographic factors that influence nutrient loading in freshwater lakes. In the first sections, I will provide essential background information regarding the phosphorus cycle and lake productivity. I will then discuss how paleolimnology can provide data useful to inform and supplement existing water regulations and management efforts. The latter half of this chapter is devoted to an assessment of the complex array of landscape factors that have been used as predictors of water chemistry.

#### 2.1 Background

Phosphorus is a naturally occurring element derived from eroded sedimentary rocks, and required by all organisms for cell maintenance (Harper 1992; Smith and Smith 2001; Wetzel 2002). Due to the inherently low availability of phosphorus in freshwater ecosystems, as compared to terrestrial environments, it typically functions as the growth limiting nutrient for primary productivity in lakes, ponds, and streams (Harper 1992; Kalf 2002; Richardson and Vaithiyanathan 2009). However, under high levels of phosphorus loading, algae blooms and enhanced rooted macrophyte (submerged aquatic plant) growth can compromise the utility of surface waters for drinking, recreation, fisheries, and industry (Hasler 1947; Sharpley et al. 1994; Wetzel 2001). Therefore, control of nutrient pollution from point and nonpoint sources has been at the forefront of regulatory agencies since the 1970s (Sharpley et al. 1994). Unfortunately, setting goals for nutrient reduction is complicated by conflicting stakeholder interests and inherent regional watershed differences (Sharpley et al. 1994). To establish more realistic and impartial lake restoration targets based on natural, background phosphorus concentrations, paleolimnological techniques can be used (USEPA 2000).

#### 2.2 Phosphorus Cycle

Many quantitative studies have been conducted within the last 60 years to determine the fate of various phosphorus fractions in freshwater ecosystems, often with the overarching goal of identifying major threats to water quality (Johnston 1991; Boers et al. 1993; Reddy and Flaig 1995; Dong et al. 2011). However, measuring the mobility and bioavailability of phosphorus in aquatic ecosystems has proved challenging because cycling is strongly regulated by a myriad of complex biogeochemical mechanisms (Richardson 1985; Richardson and Vaithiyanathan 2009). Indeed, phosphorus is an extremely active element that can change from particulate to dissolved forms as it undergoes biotic and abiotic transformations in the environment (Richardson and Vaithiyanathan 2009). Therefore, most water quality studies refer to measures of total phosphorus, an aggregate term for both particulate and dissolved phosphorus content in a water sample (Pathak et al. 2012).

Particulate phosphorus compounds bound to soil sediments and/or organic matter often dominate the total phosphorus budget (Svendsen et al. 1995; Reynolds and Davies 2001). This fraction of phosphorus enters aquatic systems primarily through stream bank erosion or nonpoint source runoff with a high content of suspended sediments (Williams 1971; Harper 1992; Wetzel 2001). Although it typically constitutes a smaller fraction of total phosphorus, dissolved inorganic phosphate (sometimes referred to as orthophosphate or soluble reactive phosphate) represents the portion of total phosphorus immediately available for biological uptake (Sharpley et al. 1994; Wetzel 2001; Mitsch and Gosselink 2007). Once considered an inconsequential transport process, subsurface and surface runoff containing dissolved phosphorus leached from agricultural soils is now recognized as an environmentally significant source of anthropogenic phosphorus enrichment of surface waters (Sims et al. 1998; Djodjic et al. 2004). Even though

soil leaching rates can be high following rainfall or snowmelt events, dissolved phosphorus is more commonly sourced from industrial inflows or sewage effluents (Holtan et al. 1988; Harper 1992; Dorioz et al. 1997; Wetzel 2001; Kalf 2002; Djodjic et al. 2004). The amount and composition of dissolved versus particulate phosphorus fractions are strongly regulated by retention and release processes in aquatic ecosystems (Svendson et al. 1995).

#### 2.2.1 Phosphorus Retention

The retention capacity of an aquatic ecosystem is the ability of that system to remove and retain phosphorus from the water column through biogeochemical processes (Reddy et al. 1999). Research on aquatic ecosystems has repeatedly attributed phosphorus retention to mechanisms such as adsorption, sedimentation, and vegetative assimilation (Harper 1992; Reddy et al. 1999; Wetzel 2001; Devesa-Rey et al. 2009). Thus, the extent of ecosystem productivity is in part regulated by these biotic and abiotic mechanisms.

Abiotic adsorption to organic and inorganic soil constituents (e.g., clay particles, organic matter, and ferric/aluminum oxides) renders phosphorus insoluble and thus unavailable to plants and microconsumers (Wetzel 2001; Mitsch and Gosselink 2007). Therefore, elevated concentrations of dissolved phosphorus can be indicative of low adsorption capacity (Kleeberg et al. 2007). This adsorption capacity is largely determined by pedogenic soil properties; for example, highly weathered, acidic soils dominated by large quantities of sesquioxides readily adsorb phosphorus (Sollins et al. 1988). Although this mechanism substantially limits its bioavailability, adsorbed phosphate ( $PO_4^{3-}$ ) can become soluble and remobilized under anoxic conditions (Khalid et al. 1977; Scheffer 1998).

Sedimentation is a fairly permanent retention mechanism that occurs as organic matter and inorganic particles settle out of the water column, potentially reducing turbidity and algae growth by immobilizing total phosphorus (Wetzel 2001). Aquatic sediments, however, have a finite ability to retain phosphorus. The extent of phosphorus immobilization in benthic substrate is greatly influenced by the composition, temperature, bioturbation, and the oxidation-reduction potential of the sediment-water interface (Khalid et al. 1977; Johnston 1991; Reynolds and Davies 2001; Devesa-Rey et al. 2009; Richardson and Vaithiyanathan 2009).

Phosphorus is also retained through biotic assimilation by autotrophic organisms, such as phytoplankton, macrophytes and periphyton into microbial (Reddy et al. 1999). In general, phosphorus storage through assimilation is more short-term than the other retention mechanisms, and is regulated by the autotrophic organism's lifespan (Reddy et al. 1999; Lund 2000; Cheng 2011). Furthermore, the efficiency of biotic assimilation varies with site-specific circumstances such as the species of plants and the plant development stage (Reddy et al. 1999).

#### 2.2.2 Phosphorus Release

Under certain physiochemical and biological conditions, phosphorus bound to sediments can be released back into the overlying water column and made available, as dissolved phosphorus, to primary producers (Johnston 1991; Dong et al. 2011). The resuspension and entrainment rates of benthic phosphorus depend on the chemical bonding nature of the sediments, oxidation-reduction (redox) conditions, and mixing from biota or wind/wave action (Engstrom and Wright 1984; Kalf 2002; Dong et al. 2011). For example, lakes with anoxic hypolimnetic waters have demonstrated elevated rates of phosphorus release from benthic sediments. Phosphorus can also be released, or desorbed, from suspended sediments in the

photic zone of the epilimnion, where it is readily available for phytoplankton uptake (Kalf 2002). Additionally, the partial senescence of submerged aquatic macrophytes can also cause nutrient leaching as foliage matures (Wetzel 2001). However, it is more common that dissolved phosphorus is liberated from decomposing detritus (Reynolds and Davies 2001). Combined, these processes ultimately determine the productivity and trophic status of lakes.

#### 2.3 Lake Productivity

#### 2.3.1 Trophic Classification of Lakes

European limnologists Einar Naumann and August Thienemann laid the foundation for the ecological classification of lakes shortly after the turn of the 20<sup>th</sup> century (Hasler 1947; International Symposium on Eutrophication 1969; Hutchinson 1973; Harper 1992). Naumann (1919) was the first limnologist to introduce the concept of oligotrophic and eutrophic waters in terms of visually observed phytoplankton productivity in lakes during the summer months (International Symposium on Eutrophication 1969; Hutchinson 1973; Wetzel 2001). Conceding to Naumann's terminology, Thienemann (1921) characterized lake trophic conditions as a function of the hypolimnetic oxygen regime during summer stratification based on observed benthic organism assemblages (Wetzel 2001). Thienemann found oligotrophic lakes with oxygenated waters to display great diversity in benthic fauna, whereas eutrophic lakes were prone to oxygen depletion and low biological diversity (Harper 1992; Wetzel 2001). It was because of these complimentary ideas that Naumann and Thienemann eventually collaborated to develop a general lake classification system reflective of trophic conditions that Naumann referred to as "asymmetric", or disharmonious (Table 2.1; Hutchinson 1973; National Research

Council 1996). However, as the number of recognized lake types increased, so did the

complexity and ambiguity of the terminology (Carlson and Simpson 1996).

Trophic Classification	Characteristics
Oligotrophy	Lakes with low production associated with low
Oligouophy	phosphorus and nitrogen
Futrophy	Lakes type with high production, associated with
Europhy	high phosphorus and nitrogen
Asidotrophy	Lake type with low production, with pH values
Acidotrophy	less than 5.5
Alkalitranhy	Lake type with low production, associated with
Акапиорпу	high calcium concentrations
Argillotrophy	Lake type with low production, associated with
Arginouophy	high clay turbidity
Siderotrophy	Lake type with low production, associated with
Siderotrophy	high iron content
Dustrophy	Lake type with low production, associated with
Dystrophy	high humic color

**Table 2.1:** Trophic lake types, as suggested by Naumann (1919) (modified from Carlson and Simpson 1996).

Regardless, lake trophic classifications are useful for communicating technical information to stakeholders, given the escalating demands on aquatic resources. Therefore, adaptations of the original, qualitative lake classification system still persist today (Hutchinson 1973). Numerous indices of trophic state have been proposed that reflect the general level of primary productivity, quantitatively defined by total phosphorus concentrations, chlorophyll-*a*, and/or Secchi disk depth (Carlson and Simpson 1996; Wetzel 2001; USEPA 2007a). Although there is still some disagreement about the precise trophic boundaries in temperate freshwater lakes, total phosphorus generally ranges from < 4  $\mu$ g/L in unproductive, oligotrophic waters to >100  $\mu$ g/L in highly productive, hypereutrophic waters (Carlson 1977; Rast and Lee 1978; Ney 1996; Wetzel 2001). These trophic conditions are required to be monitored and reported to the EPA under Section 314 of the Clean Water Act (Clean Water Act 1972).

#### 2.3.2 Eutrophication

Eutrophication is a natural part of the successional development of aquatic ecosystems that often signifies the intermediate and terminal stages of lake ontogeny (Wetzel 2001; Brenner and Escobar 2009). Under eutrophic conditions, rates of photosynthetic productivity are relatively high and the accumulation of organic biomass and sediments can eventually result in a reduction in water depth or even a complete infill of the lake basin (Wetzel 2001). The timeframe required for autochthonous and allochthonous sediments to accumulate and cause natural lake senescence can be thousands of years, long surpassing the duration of most ecological investigations (Wetzel 2001; Brenner and Escobar 2009; Mitraki 2012).

Although eutrophication is a natural part of lake succession, anthropogenic nutrient enrichment has accelerated changes that would have otherwise taken centuries or millennia to occur (Sawyer 1947; Harper 1992). This acceleration of primary production, aptly referred to as cultural eutrophication, is consequential of nonpoint source pollution from agricultural drainage and urban effluents (Richardson and Vaithiyanathan 2009). There have been ample experiments to date that have demonstrated these associations, particularly as they pertain to phosphorusbased fertilizers, which increased in use by commercial agricultural operations from 0.5 million metric tons in 1945 to 10.3 million metric tons in 1993 (e.g., Puckett 1995; Liu et al. 2010; Liu et al. 2012).

Yet up until the mid-20<sup>th</sup> century, lake fertilization studies were few in number and mostly offered only circumstantial evidence that was difficult to interpret because experimental controls were lacking (Hasler 1947). Carbon dioxide production from bacterial, aerobic respiration was even hypothesized to be the primary regulator of algal growth for a period in the late-1960s (Kuenzel 1969; Harper 1992). However, several projects involving deliberate lake fertilization as a surrogate for nonpoint runoff during the early 1970s conclusively identified total phosphorus as the limiting nutrient in temperate, freshwater aquatic systems (Dillon and Rigler 1974; Schindler 1974; Smol 2002).

The most recognized of these studies to unequivocally accredit eutrophication to excess phosphorus came from a series of controlled, whole-lake experiments in Northwestern Ontario's Experimental Lakes Area (Schindler 1974). This experiment manipulated the chemical input of several adjacent lakes and examined the outcome for biological reactions (Schindler 1974; National Research Council 1996). Noxious algal blooms were observed in the lake with high phosphorus influents, while the lake with only carbon and nitrogen inputs remained relatively unchanged (Schindler 1974). This significant limnological breakthrough disproved speculations that hypothesized carbon or nitrogen to be the limiting nutrient for primary production in most freshwater lakes, and instead identified phosphorus as the most important nutrient (Schindler 1977). As a result, phosphorus has arguably become the most intensively studied element in fresh waters (Vollenweider 1968; Chapra and Reckhow 1979; Wetzel 2001; Håkanson and Boulion 2002), contributing data useful for the management and regulation of this limiting nutrient.

#### **2.4 Phosphorus Management and Regulations**

In systems degraded by cultural eutrophication, the reduction of phosphorus can produce beneficial results such as the elimination of nuisance algal blooms that otherwise clog the filters at waste water treatment facilities and cause odors in treated waters (Juggins et al. 2013). Phosphorus reduction in eutrophic lakes has also rehabilitated fish communities that were

previously limited in biodiversity (Makarewicz and Bertram 1991; Smith et al. 1999; Ludsin et al. 2001; Wetzel 2001; Juggins et al. 2013). For example, nutrient abatement programs implemented as part of the Great Lakes Water Quality Agreement of 1972 restored the severely degraded Lake Erie basin system to its naturally oligotrophic condition (Ludsin et al. 2001). As a result, the higher oxygen levels in the hypolimnion aided the return of sensitive benthic macroinvertebrate species and increased fish diversity (Nürnberg 1995; Ludsin et al. 2001). Furthermore, enhanced light penetration improved prey detection capabilities, causing an increase in piscivorous fish abundance (Persson et al. 1991; Miner and Stein 1993; Ludsin et al. 2001).

Nevertheless, some scholars argue that the over-reduction of nutrients can cause an imbalance in an aquatic ecosystem's food web structure, ultimately leading to unproductive and declining fisheries (Ney 1996; Ashley et al. 1997; Larkin and Slaney 1997; Stockner et al. 2000). Stockner et al. (2000, p.7), for example, even suggest the reintroduction of phosphorus in a "carefully controlled and ecologically sensitive way to restore sufficient fisheries production levels." Furthermore, nutrient removal can stem the natural ontogeny of lakes to become shallower and more productive as they senesce (Stockner et al. 2000). This dichotomy of stakeholder interests in ecosystem services can make it challenging for watershed organizations and/or local governments to develop water quality strategies that uphold the objectives of the Clean Water Act (CWA 1972).

To address the need for impartial benchmarks by which to gage water pollution prevention and control measures under CWA jurisdiction, the EPA developed a national nutrient strategy to encourage states to adopt numeric nutrient criteria that define concentrations limits in lakes and reservoirs (USEPA 2000). However, setting nutrient criteria for phosphorus has been

difficult, in part, because there is a great deal of variability in inherent phosphorus levels and in the manifestation of eutrophication in freshwater systems throughout the country (USEPA 1998). For this reason, a single comprehensive concentration number is not appropriate for managing nutrient pollution (USEPA 1998). It is thus necessary to determine natural limnological conditions, prior to human influences, to effectively manage lake systems; however, long-term data that can provide this anecdotal water quality information prior to 1950 are generally lacking (Kalff 2002).

Therefore, considerable attention has been given to reconstructing past nutrient levels from lake sediments profiles (Smol 2002). However, because of diagenetic physiochemical changes that occur after deposition, it is inaccurate to simply analyze lacustrine sediments for an archived chemical record (Engstrom and Wright 1984; Smol 2002). Rather, paleolimnological proxies can be used to reconstruct pristine baseline limnological conditions that function as the benchmarks for management and restoration goals in disturbed watersheds.

#### **2.5 Paleolimnological Reconstructions**

Sedimentary chemical profiles from lacustrine cores are limited in their ability to reconstruct pre-settlement phosphorus loading because retention in sediments is strongly regulated by geochemical processes that alter the sorption capacity of soluble reactive phosphorus (Carignan and Flett 1981; Anderson et al. 1993; Wetzel 2001; Dong et al. 2011). Additionally, the possibility of post-depositional mobility of phosphorus in lacustrine sediments had long been proposed by many (Williams 1971; Bloesch 1976), but was first conclusively proven by Carignan and Flett (1981). Their study provided evidence of an upward migration in dissolved phosphorus from the reduced zone in lake sediments toward the oxidized sediment-

water interface. Consequentially, interpretation of dated sedimentary phosphorus may yield erroneous records of past lake conditions.

Although misinterpretations of sedimentary phosphorus obviate its usefulness as a record of past ambient lake chemistry, other methods have been developed to reconstruct nutrient concentrations based on biological proxy data from the sediment record (Anderson et al.1993; Garrison and Fitzgerald 2005). These proxies are not mobile in sediments to the same extent that sedimentary phosphorus is in the lacustrine sediment profile. The most common paleolimnological indicators for pre-settlement water quality reconstructions are fossil diatomaceous algal assemblages (Schindler 1987; Anderson et al. 1993; Reavie et al. 1995).

#### 2.5.1 Diatoms as Proxy Indicators of Water Chemistry

Diatoms are small, rapidly-reproducing algae (Class Bacillariophyceae) found abundantly in fresh and saline waters. Species are relatively easily distinguished by distinct morphological features in the frustule (cell wall) structure (Smol and Stoermer 2010). These intricate, siliceous frustules of diatoms are well preserved in most lake sediments and can be used to decipher longterm ecological changes (Hall and Smol 1992; Battarbee 2000; Smol and Stoermer 2010). Fossil diatom assemblages are exceptionally useful for quantifying pre-settlement lake conditions to a high degree of certainty because each species has optimal ecological preferences and specific abiotic constraints (Fritz et al. 1991; Dixit et al. 1992). These paleolimnological proxies are well-suited for studies of lake eutrophication, in particular, because certain diatom taxa have demonstrated a rapid response to changes in nutrient concentrations (Hall and Smol 1999).

Prior to the late 1980s, diatom records in lacustrine sediments were subjectively interpreted using a relatively small number of indicator species to qualitatively describe trophic

status (Brumgan 1978; Ennis et al. 1983; Hall and Smol 1992; Anderson et al. 1993). For example, a diatom assemblage dominated by *Stephanodiscus* revealed eutrophic conditions, whereas an abundance of *Cyclotella* was indicative of oligotrophic conditions (Anderson et al. 1993). Fortunately, the ability to make quantitative, statistically-based inferences of historic phosphorus concentrations has strengthened within the last few decades due to the development of multivariate ordination techniques and transfer functions for diatom data (ter Braak 1986; Birks 1998; Hall and Smol 1992; Anderson et al. 1993; Reavie et al. 1995; Smol 2002).

Before a transfer function can be developed to infer pre-settlement limnological conditions from fossil diatom assemblages, the environmental optima of taxa must be estimated (Anderson et al. 1993; Smol 2002). This information can be gathered from pre-existing lake surveys, but more frequently involves using surface calibration sets (also known as 'training sets') from a range of lakes, under the assumption that ecological preferences of taxa have not changed over time (Fritz et al. 1999; Stevenson and Pan 1999; Smol 2002). To produce a calibration set, diatom assemblages are collected from sediments at the surface-water interface (i.e., the uppermost ~1-2 cm) of a calibration lake core (Smol 2002). Under normal sedimentation rates, this interval represents the last two or three years of accumulation within a lake (Smol 2002). Water samples are also collected at the time of sediment coring and analyzed for environmental variables such as total phosphorus, total nitrogen, and pH and used in conjunction with the diatom data (Dixit et al. 1992). Finally, ordination techniques are applied to summarize the calibration dataset.

Direct and indirect gradient analyses are the most commonly used ordination techniques to relate latent (theoretical) environmental variables to diatom growth and assemblage distributions in a calibration dataset (ter Braak 1986; Smol 2002; Smol and Stoermer 2010). An

indirect gradient analysis is a linear, two-step approach involving ordination and environmental gradient identification to reveal overall patterns in species variation within a dataset (ter Braak 1986; Smol 2002). Essentially, variations of diatom species and water chemistry variables are explored independently using techniques such as detrended correspondence analysis (DCA) and principal components analysis (PCA), respectively (Figure 2.1; Battarbee et al. 1999).



**Figure 2.1:** An example of a common ordination approach to transfer function development, redundancy analysis (RDA), to assess the significance, strength, and direction of environmental variables as predictors of diatom species composition from a calibration dataset (modified from Juggins et al. 2013).

Conversely, direct gradient analysis allows both datasets to be analyzed together (Battarbee et al. 1999). This form of multivariate regression identifies the best fitting environmental variable to the diatom species data using a response model in which the abundance of any species corresponds to the value of each latent environmental variable (ter Braak 1986; Smol 2002; Smol and Stoermer 2010). For example, a canonical correspondence analysis (CCA) is a commonly used direct gradient analysis technique developed by ter Braak in 1986. Diagrams generated by CCA use arrows of varying length and orientation to signify the relative importance of the relationship between diatom community composition and the measured environmental variables of the calibration lakes (ter Braak 1986; Battarbee et al. 1999; Smol 2002).

The results of these techniques can provide valuable information for lake management, however if the sediment samples corresponding to the diatom assemblages are not dated (either with absolute or relative dating techniques), the inferred phosphorus concentrations could be misinterpreted. For example, in a recent publication, Bachmann et al. (2013) estimated the extent of trophic changes that have occurred from anthropogenic activities by examining diatom-inferred phosphorus concentrations for 240 lakes from the EPA's National Lake Assessment (NLA). They concluded that total phosphorus in these lakes decreased by 14% since Euro-American settlement. However, this determination was made under the assumption that the bottom-core sediments represent a time prior to Euro-American settlement, when in fact; the EPA only conducted a rudimentary, and somewhat subjective, evaluation of bottom-core sample ages (USEPA 2010). Therefore, these results, which could have serious implications to nutrient criteria establishment, are indefensible because of the unconfirmed time period represented by the reconstructed phosphorus concentrations.

#### 2.5.2 Pollen as a Proxy Indicator of Landscape Disturbances

In addition to diatom fossil interpretations, a considerable amount of information can be gained from allochthonous materials in the sediment profile (Smol 2002). Pollen grains deposited from the vegetational community surrounding a lake and preserved in lake sediments offer a high-resolution archive of past vegetation (Smol 2002). The identification and enumeration of grains to the genera or species level allows palynologists to reconstruct trends in vegetation succession and/or document landscape changes associated with anthropogenic activities. Moreover, an increase in *Ambrosia artemisiifolia* (common ragweed) pollen percentages has been used as a chronostratigraphic marker of agriculture, development, and deforestation in lake catchments (Reeder and Eisner 1994; Smol 2002; Gerber et al. 2011).

*Ambrosia* is a herbaceous plant native to temperate North America that quickly flourishes in disturbed habitats such as roadsides, urban areas, and agricultural clearings (Smol 2002; Gerber et al. 2011). In the Great Lakes region, several sediment core studies have found a distinct peak in *Ambrosia* pollen corresponding to <sup>14</sup>C dates at  $150 \pm 50$  years B.P., the approximate time frame of European land clearance in the Midwestern United States and adjacent Canada (McCarthy and McAndrews 1988; Weninger and McAndrews 1989; Reeder and Eisner 1994; Finkelstein et al. 2005). For this reason, *Ambrosia* pollen can serve as a paleoenvironmental proxy in place of written analogs of landscape changes during Euro-American settlement.

#### 2.6 Lake-Landscape Interactions

Numerous studies have shown that the allochthonous materials in a lake are reflective of the land use/land cover composition in the local catchment. Proportions of impervious surface,

agriculture, and lake shore development are some of the most widely used predictors of phosphorus concentrations because these are the anthropogenic land uses known to strongly influence water chemistry (Wang et al. 1997; Paul and Meyer 2001). Although land cover is often used as a direct linkage to water quality, not all of the variation in lake phosphorus concentration can be explained by these effects alone (Bremigan and Soranno 2008). The chemistry and biology of inland lakes can also be naturally influenced by hydrogeomorphic (HGM) features such as lake morphometry, geology, and hydrologic flowpaths (Bremigan and Soranno 2008; Martin et al. 2011).

#### 2.6.1 Hydrogeomorphic Features

HGM features are natural aspects of the landscape, minimally impacted by anthropogenic activities, and can therefore be used as a foundation upon which to measure changes in the environment (Bremigan and Soranno 2008; Martin et al. 2011). Spatial variations of these physio-chemical landscape factors are hypothesized to influence the rate at which phosphorus enters a lacustrine environment, at both local and regional scales (Soranno et al. 1996; Schnurrenberger 2002; Veith et al. 2003; Wagner et al. 2011). For example, phosphorus mobilization processes at a regional scale are largely affected by climatic and geologic factors such as precipitation, lake alkalinity, and soil chemistry (Wagner et al. 2011). However, in local-scale models, primary productivity has been predicted by basin morphometry (i.e., lake depth and surface area) that influence water residence time and internal loading in lentic (still) surface waters (Wetzel 2001; Kalff 2002; Wagner et al. 2011).

Water residence time (WRT), a function of surface hydrology and lake depth, is defined as the average time required to drain a basin from the outflow if the inflow was removed (Wetzel

2001). This metric is often used as a rough measure of how quickly water quality and planktonic communities will respond to changes in phosphorus loading (Wetzel 2001; Kalff 2002). Generally, if surface runoff from the local catchment is rich in nutrients, organic matter, and/or pollutants, lakes with a shorter WRT will have greater water clarity (Reavie et al. 1995).

Lake depth is another important HGM feature that can influence biological and chemical parameters through the regulation of solar energy distribution, thermal stratification, and lake circulation patterns (Kalff 2002; Wetzel 2001). For example, summer thermal stratification of deep lakes (> 10 m) in temperate regions can isolate the epilimnion from substrate interaction and prevent the resuspension of sediments in the water column (Hutchinson 1973; Scheffer 1998). Deep lakes generally maintain thermal stratification following periods of fall and spring turnover because the metalimnetic thermocline acts as an effective barrier to convection currents and wind-induced circulation (Wetzel 2001; Kalf 2002). Without mixing, the cold temperatures (~4°C) of the hypolimnia promote oxygen solubility and phosphorus retention in benthic sediments (Kalf 2002). Anoxic hypolimnia most commonly occur when lakes are shallow ( < 3 m) and have a WRT of < 2 years (Kalf 2002).

Shallow lakes have a higher ratio of sediment surface to water column, resulting in increased susceptibility to internal nutrient loading via resuspension of sediments (Scheffer 1998; Dong et al. 2011). Håkanson (1996), for example, found a significant positive relationship between lake depth and Secchi depth because deeper lakes had less resuspension of materials in the water column. Frequent mixing by wind and wave action is more likely to occur in lakes that are shallow relative to their surface area (Osgood 1988; Detenbeck et al. 1993). This replenishment of phosphorus into the water column has, in many cases, delayed the reduction of in-lake phosphorus concentrations (Jeppesen 2005). Furthermore, Vollenweider (1968) and

others have demonstrated that consideration of total phosphorus loading and mean depth alone can be a fairly accurate metric to predict lake eutrophication (Volleinweider 1968; Dillon and Kirchner 1975; Chapra 1997). As shown in Figure 2.2, shallow lakes become eutrophic at lower rates of phosphorus loading, in comparison to deep lakes.



**Figure 2.2:** Vollenweider's (1968) phosphorus loading plot classifies lakes into trophic states based on mean depth (m) and annual phosphorus loads  $(gPm^{-2}yr^{-1})$ . The lines marking the thresholds between eutrophic and oligotrophic lake conditions are suggested to correspond to phosphorus concentrations of 20 µg/L and 10 µg/L, respectively (modified from Chapra 1997).
# 2.6.2 Phosphorus Dynamics in Wetland Ecosystems

Wetlands are transitional ecosystems commonly located at the interface between terrestrial and aquatic ecosystems. They are defined by prolonged periods of saturation that result in the establishment of organic or mineral hydric soils and hydrophytic vegetation (Tiner 1999; Mitsch and Gosselink 2007). Often gleying and redoximorphic features (mottling) of hydric soils reflect the poor drainage and reductive conditions common to wetlands (Etherington 1983; Tiner 1999; Mitsch and Gosselink 2007). Due to their strategic landscape location, wetlands have received considerable attention for their potential to sequester both sediments and nutrients (Tiner 1997). In fact, among the many valued ecosystem services provided by wetlands (i.e., waterfowl habitat, flood mitigation, and groundwater recharge), perhaps none are as highly regarded as phosphorus retention (Mitsch and Gosselink 2007; Stein et al. 2010).

Numerous studies have demonstrated the ability of wetlands to reduce water quality impairments related to excess nutrient loading (Johnston 1991; Reddy et al. 1999). In fact, much of the published research within the last two decades has largely focused on the use of constructed wetlands as a nutrient sink in the wastewater treatment process. For example, one study found total phosphorus removal in a constructed wetland to vary between 40 to 60%, depending on inflow loading (Vymazal 2007).

Although wetlands are hypothesized to be natural nutrient sinks that filter out contaminants and improve water quality, some studies show the amount of total phosphorus lost from wetlands to lakes or rivers is significantly greater than that from terrestrial systems (Richardson 1985; Devito et al. 2000; Richardson and Vaithiyanathan 2009). For example, Richardson (1985) suggested phosphorus retention efficacy may be overestimated because the

biogeochemical mechanisms controlling phosphorus transformation and translocation in wetlands remain poorly understood.

For example, emergent macrophytes in wetland systems can effectively assimilate and store phosphorus during the growing season, and even during the fall/winter months, albeit with decreased uptake rates (Reddy and Flaig 1995). However, a relatively large amount of dissolved phosphorus is returned back into the water column as a result of initial leaching or detrital decomposition (Reddy and Flaig 1995; Reddy et al. 1999). Additionally, accumulations of phosphorus in poorly and very poorly drained soils with anaerobic conditions have a high potential to become remobilized (Benhart 1994; Carpenter et al. 1998). Therefore, phosphorus storage in wetland biomass is usually short-term, particularly when compared to the long-term storage ability of upland terrestrial ecosystems (Reddy et al. 1999).

# 2.6.3 Phosphorus Yields from Forested Ecosystems

The distribution and abundance of tree species reflects broad spatial patterns in soil texture and climate (Harmon 2009). Generally, sandy-textured soils in cold climate regimes (i.e., Spodosols) are well-tolerated by coniferous trees, which produce a thick, acidic O horizon that fosters slow decomposition rates (Barnes and Wagner 1981; Harmon 2009). The eastern deciduous forest tree species that dominate southern Lower Michigan's forest composition, such as American beech, red maple, and tulip poplar, tend to require sandy loam soils and longer growing seasons (Barnes and Wagner 1981; Yahner 2000; Harmon 2009). Regardless of the forest classification, the relatively coarse-textured forest soils enhance precipitation and snowmelt infiltration, resulting in minimal surface runoff and nutrient contributions from upland ecosystems (Fisher and Binkley 2000; Miller et al. 2005). Therefore, forested buffers, narrow

bands of arboreal vegetation surrounding lakes and streams that reduce inputs of solar radiation (by shading), sediments, and nutrients, are one of the strategies to mitigate phosphorus runoff from agricultural and urban areas (Osborne and Wiley1998; Devito et al. 2000; Baker 2006).

Unsurprisingly, the rates of dissolved and particulate phosphorus retention from upland ecosystems to aquatic ecosystems have been shown to decrease following tree harvesting or other periods of rapid landscape denudation (Omernik et al. 1981; Filippelli and Souch 1999; Reynolds and Davies 2001). This is, in part, due to the presence of long-chained, hydrophobic, organic molecules released from decaying or incinerated detritus and micro-organisms that can reduce the infiltration capacity of soils and increase runoff into surface waters (Reynolds and Davies 2001). Although the hydrophobicity of soils is spatially variable, it is most prominent after prolonged dry periods or episodes of intense wildfires (Doerr and Shakesby 2011). When hydrophobicity is low, densely forested areas with non-hydrophobic soils appear to retain phosphorus through biomass accumulation and sorption to clay sediments (Reynolds and Davies 2001; Kramer et al. 2006).

### 2.6.4 Proximity to Surface Waters

Simple measures of watershed composition are often evaluated to explain variability in water quality parameters, however there are uncertainties regarding the appropriate scale for these analyses (Soranno et al. 1996; Cifaldi 2002; Gémesi et al. 2011). Although they are less considered and more difficult to quantify, empirical evidence suggests that knowledge of spatial variations in land cover could more accurately explain source contributions of phosphorus loadings (Alexander 2002; Gémesi et al. 2011). Therefore, spatially explicit measures of landscape configuration and proximity are becoming more common in models (Gergel and

Turner 2002; Fraterrigo and Downing 2008; Gémesi et al. 2011). An evaluation of these variables in Michigan will help to achieve a greater understanding of lake-landscape interactions as a whole.

### **3. STUDY AREA**

### **3.1 Study Area Description**

The 48 inland lakes in this study are geographically distributed throughout Michigan. Given the expansive extent of the study area across several ecoregions, it is necessary to first discuss the spatial diversity of landscape patterns that are largely a factor of climatic controls and glacial history. I will then address the process of study lakes selection and the broad characteristics of the dataset.

# 3.1.1 Climate and Vegetation

Michigan is geographically situated between 41.5° N to 48.5° N, the approximate halfway point between the equator and the North Pole. Given this large range in latitude, as well as Michigan's proximity to the Great Lakes, temperature and precipitation fluctuate within the state (Andresen and Winkler 2009). The annual mean temperature, based on 1971-2000 climate data, ranges from approximately 45° F in the Lower Peninsula to approximately 40° F in the Upper Peninsula (from 1971-2000) (NOAA National Climate Data Center 2002). Colder temperatures prevail in more northerly regions, resulting in a shorter growing season. Historically, annual mean temperatures in the Midwest were approximately 6.6° cooler in 1900 than modern recorded temperatures, according to data obtained from the CRUMET3 data set (Brohan et al. 2006; Andresen et al. 2012).

Total annual precipitation tends to decrease from west to east as proximity to prevailing westerly winds over the Great Lakes decreases (Andresen and Winkler 2009). This gradient of moisture across Michigan, known as the "lake effect," enhances winter snowfall and moderates temperatures near the eastern shore of Lake Michigan and the southeast shore of Lake Superior. The result is a mesic and mild microclimate within 80 km of these Great Lakes that can support temperate hardwood tree species, such as red oak and sugar maple during winters in the otherwise frigid Upper Peninsula (Kenoyer 1933; Harmon 2009).

The study region lies within the northernmost extent of the Eastern Deciduous Forest biome and near the southernmost extent of the Canadian Boreal Forest biome (Figure 3.1; Harman 2009). Climate is the broad-scale influence of predominant vegetation in Michigan, which shifts from eastern broadleaf forest to mixed coniferous-deciduous forest along a transitional area in the central-Lower Peninsula referred to as the forest tension zone (Figure 3.1; Medley and Harmon 1998; Anderson 2005; Harmon 2009; Hupy and Yansa 2009a, 2009b).



**Figure 3.1:** Map of pre-settlement forests in Michigan as interpreted from the General Land Office (GLO) public land surveys between 1816-1856 (Comer et al. 1995).

The tension zone is an ecotonal area that marks a general climatic boundary between cooler and warmer mean annual temperatures and the decreased number of growing degree days as latitude increases and the influence of the polar jet stream becoming more predominant (Eichenlaub et al. 1990; Andresen and Winkler 2009; Hupy and Yansa 2009a). Localized patterns in vegetation, however, are strongly related to variations in soil types resulting from repeated glaciations (Blewett et al. 2009; Andresen et al. 2012).

#### 3.1.2 Glacial History

Thick glacial deposits and complex sedimentology and landforms are reflective of advancing and retreating continental ice that covered Michigan during the Late Wisconsin glaciation (Wetzel 2001; Blewett et al. 2009). The Laurentide ice sheet began to recede from the southernmost part of the Lower Peninsula of Michigan about 17,000 cal (calendar) years ago and had vacated the northern Upper Peninsula by 10,000 cal yr BP (Blewett et al. 2009). Glacial sediments vary in thickness across the state, ranging from 365 m in the interlobate region of Osceola County (Lower Peninsula) to none at exposed bedrock in the driftless regions of the Upper Peninsula (Blewett et al. 2009; Schaetzl 2009). Similarly, soil textures fluctuate across the state in close reflection to the glacial parent material (e.g., till, outwash, or loess) in which they formed (Schaetzl 2009).

These variations in soil textures and drainage classes are known to support structurally and functionally different vegetation communities (Henne 2006). Furthermore, the interaction of spatial edaphic variations with climatic gradients (latitudinal, as well as lake-affected areas vs. inland locales) creates a mosaic of different forest community types in Michigan (Albert 1995, Henne 2006). For example, coarse-textured sandy soils south of the forest tension zone mostly

support broadleaved forests, but coarse-textured sandy soils north of the tension zone are predominately associated with coniferous needleleaf forests (Harmon 2009; Schaetzl 2009).

# 3.1.3 Hydrology

The last glacial recession not only shaped Michigan's soil and vegetation patterns, but also left a mosaic of interacting lakes, rivers, and wetlands (Figure 3.2). Kettle lakes commonly formed in interlobate areas and outwash plains when chunks of ice detached from a retreating



**Figure 3.2:** Map of pre-settlement lakes, rivers, and wetlands in Michigan as interpreted from the General Land Office (GLO) public land surveys between 1816-1856 (Comer et al. 1995).

glacial lobe became buried in outwash and eventually melted to form depressions which filled with water. These lakes are discernible by deep basins with irregular bathymetry and are often geographically isolated from streams (Wetzel 2001; Tiner 2003; Wolfson 2009). Many of the shallower kettle lakes eventually filled with plant debris and sediments during periods of stabilizing drainage patterns and falling lake levels, becoming depressional wetlands (Tiner 2003).

Some lakes and wetlands in Michigan have other origins. Oxbow lakes, for example, form when alluvial deposits cutoff river meander bends. Additionally, karst lakes are known to form where caverns in highly soluble carbonate bedrock have collapsed to form water-filled depressions, a phenomenon common to the eastern Upper Peninsula and northern Lower Peninsula (Wolfson 2009). The abundance of surface water in Michigan makes it an ideal region for land cover-water quality studies, however much of these natural resources have been greatly altered since Euro-American settlement in the mid-1800s.

### 3.1.4 Euro-American Settlement

Michigan's native landscape, as interpreted from the General Land Office (GLO) surveys of 1816-1856 (Figure 3.3), consisted of approximately 29% mixed forest, 27% deciduous forest, 15% evergreen forest, 23% wetlands, 3% grasslands, 2% lakes and streams, and ~ 1% barren land (Comer et al. 1995). The impetus of the surveys was to facilitate the sale of available land to the general public by defining the location and description of parcels in a systematic manner (Thomas 2009). Initial observations from deputy surveyors often reported the land as "poor and brushy" or "swampy and tedious to survey" (Thomas 2009). Regardless of this inhospitable impression, pioneer farmers were attracted to Southern Michigan for the fertile prairie soils and potentially arable land below the hardwood forests (Lewis 2009).



**Figure 3.3:** Map of pre-settlement land cover in Michigan as interpreted from the General Land Office (GLO) public land surveys between 1816-1856 (Comer et al. 1995).

By 1860, the landscape was drastically altered by Euro-American colonization that culminated in widespread deforestation, wetland drainage, and agricultural expansion. For example, industrial lumbering of the northern Lower Peninsula and the Upper Peninsula's expansive coniferous forests began servicing the rapid expansion of cities such as Chicago by 1840 (Lewis 2009). Farther south, much of the original oak-openings/prairie lands in southeastern Lower Michigan were converted to oak forests through fire suppression practices (Albert 1995; Comer et al. 1995). Statewide, an estimated 50% of the native wetlands were ultimately drained and converted to agricultural land (Albert 1995; Comer et al. 1995; Lewis 2009). An understanding of the natural variability of pre-settlement vegetation, landforms, soils and hydrology was a critical requirement for selecting lakes in this study, in order to capture the spatial variability of these natural features that have since been anthropogenically altered, to varying degrees.

#### 3.2 Study Lake Selection and Characteristics

In this thesis research, lacustrine sediment core data and water chemistry measurements were compiled from two broader sources: the EPA's 2007 National Lake Assessment (NLA) and an associated project managed by Dr. Patricia Soranno of Michigan State University (MSU) and Paul Garrison of the Wisconsin Department of Natural Resources (WDNR) in 2007. Diatom assemblages from bottom-core sediments of 48 lakes were used to reconstruct phosphorus concentrations, theoretically representative of undisturbed lake conditions. Verification of the 'pre-settlement age' of the bottom ~2 cm of core sediments (maximum depth of core bottoms ranged from 23 to 69 cm from the sediment-water interface) was essential to this research; therefore initial lake selection from the NLA and Soranno/Garrison datasets was based on access to these sediment samples and by a pollen method employed to make this determination, which is detailed in Chapter 4 of this thesis.

#### 3.2.1 Lake Datasets

The EPA's 2007 NLA sampled sediment diatoms and physiochemical parameters of 1,028 lakes throughout the contiguous United States using a randomized, probability-based site selection method (USEPA 2007a, 2009; Stevenson et al. 2013). This statistical sampling approach was leveraged to randomly select study lakes from the USGS National Hydrography Dataset (NHD) if they met a set of predetermined criteria. The main factors in selection were size, depth, and equal geographic representation of lakes throughout the United States. More

specifically, freshwater lakes or reservoirs were required to be > 4 ha in size, >1m deep, and spatially distributed across the lower 48 states (USEPA 2007a, 2009). Accessibility and safety were also taken into consideration. The final compilation of target lakes included seepage lakes, drainage lakes, and even constructed impoundments (USEPA 2009).

Fifty-four Michigan lakes were sampled as part of the 2007 NLA, however diatominferred phosphorus analyses of sediment core-bottoms were not performed for twelve of these lakes. For the remaining forty-two lakes, a set of criteria was used by the EPA to establish confidence in core-bottoms representing pre-disturbance conditions (USEPA 2010). In general, long cores and/or natural lakes with presumably low sedimentation rates were awarded a high reliability ranking (USEPA 2010). Based on these criteria, the lab results from six lakes were excluded from this thesis study because the sediment cores yielded uncertain data or the lake was a constructed impoundment. Lastly, two of the lakes were duplicated in the Soranno/Garrison dataset and therefore not included in my initial dataset that was subjected to pollen analysis (Appendix A). Therefore, a total of thirty-four lakes from the EPA's 2007 NLA were considered for potential use in this study.

The lacustrine sediment samples and water chemistry data collected in 2007 by Mr. Paul Garrison and Dr. Patricia Soranno came from 31 inland lakes located throughout Michigan. Sample sites were originally chosen to represent a range of catchment and lake morphometry values and hydrologic settings. Similar to the NLA dataset, lakes are each at least 1 m deep, a minimum of 4 ha in area, and they represent both seepage and drainage hydrological settings (P. Soranno pers. comm. 2012).

Of the 31 lakes in the Soranno/Garrison dataset, two of the lake sediment samples were not available to test for relative age and were therefore excluded from this study (Appendix A). The remaining 29 lakes from the Soranno/Garrison dataset were analyzed to verify presettlement conditions using *Ambrosia* pollen as a relative dating method (described in Chapter 4). Based on the palynological dating method used to verify pre-settlement representation of core-bottoms, diatom-inferred phosphorus and hydrogeomorphic data from a total of 48 study lakes (compiled from the 2007 NLA and Soranno/Garrison data sources) were used in the final dataset that is the basis of this thesis research (Table 3.1).

Lake Name	County	Latitude	Longitude	North/South	Max	Surface
	U		0	of FTZ*	Lake	Area (ha)
					Depth	
					(m)	
Coldwater	Branch	41.83	-84.98	South	28.05	647.77
Bartholomew	Branch	41.88	-84.93	South	17.07	35.30
Clark	Jackson	42.12	-84.32	South	15.50	232.86
Campbell	Kalamazoo	42.32	-85.47	South	11.10	59.59
Eagle	Kalamazoo	42.33	-85.32	South	9.45	29.88
Pleasant	Jackson	42.39	-84.35	South	15.85	114.14
Warner	Barry	42.47	-85.53	South	11.20	16.02
Chemung	Livingston	42.58	-83.85	South	21.10	120.09
Mill	Oakland	42.74	-83.31	South	9.00	11.22
Campau	Kent	42.83	-85.45	South	15.24	38.29
Murray	Kent	43.03	-85.38	South	19.51	140.62
Stoner	Kent	43.15	-85.48	South	1.50	30.02
Pine	Kent	43.22	-85.47	South	6.80	20.59
Whitefish	Montcalm	43.33	-85.54	South	16.46	206.77
Blue	Mecosta	43.62	-85.28	North	15.00	88.65
Big Star	Lake	43.83	-85.95	North	7.62	365.26
Eight Point	Clare	43.84	-85.07	North	7.70	159.99
Big	Osceola	43.87	-85.20	North	25.91	86.15
Cadillac	Wexford	44.24	-85.43	North	8.23	481.65
Mitchell	Wexford	44.24	-85.47	North	6.71	1089.16
Bear	Manistee	44.43	-86.16	North	7.20	768.20
Au Sable	Ogemaw	44.43	-83.92	North	15.00	105.58

**Table 3.1:** Study area information for the 48 lakes in the final dataset.

# Table 3.1 (cont'd)

Big Blue	Kalkaska	44.81	-84.90	North	27.40	44.24
Big Twin	Kalkaska	44.82	-84.96	North	24.38	85.20
Starvation	Kalkaska	44.85	-84.95	North	14.33	79.18
West Twin	Montmorency	44.88	-84.35	North	8.53	535.13
Hanley	Antrim	45.08	-85.26	North	8.23	37.44
Deer	Charlevoix	45.16	-84.98	North	6.71	197.84
Blomgren	Dickinson	45.87	-87.80	North	1.80	31.73
McDonald	Delta	46.04	-86.81	North	2.20	11.82
Ottawa	Iron	46.08	-88.77	North	27.43	217.42
Duck	Gogebic	46.20	-89.22	North	7.60	261.42
Imp	Gogebic	46.22	-89.08	North	26.21	38.82
Lotto	Marquette	46.26	-88.09	North	5.00	43.39
Little	Marquette	46.27	-87.36	North	15.24	187.16
Witch	Marquette	46.28	-88.01	North	28.96	87.57
Dewey	Marquette	46.30	-87.78	North	2.50	20.65
Bobcat	Gogebic	46.36	-89.67	North	5.49	38.37
St. Kathryn	Iron	46.38	-88.72	North	9.10	65.05
Howe	Alger	46.43	-87.08	North	1.00	36.23
Fence	Baraga	46.48	-88.19	North	16.00	93.91
Teal	Marquette	46.51	-87.62	North	9.75	202.60
Gogebic	Gogebic	46.47	-89.58	North	7.00	5280.38
Keewaydin	Baraga	46.60	-88.12	North	7.00	52.59
Craig	Baraga	46.62	-88.18	North	7.62	57.18
Hall	Houghton	46.99	-88.76	North	14.00	6.23
Mud	Houghton	47.13	-88.32	North	3.20	69.28
Bailey	Keweenaw	47.46	-88.10	North	1.00	77.70

**\*FTZ = Forest Tension Zone** 

# 3.2.2 Lake and Native Vegetation Catchment Characteristics

Of the 48 lakes in the dataset, 14 are located south of the forest tension zone's center mark and 34 lakes are located north of this ecotone (Figure 3.4). Comparisons of the presettlement land cover in both the northern and southern local lake catchments are shown in Figures 3.5 and 3.6. Unsurprisingly, deciduous forests comprise the majority (55%) of the presettlement land cover in local lake catchments south of the forest tension zone while mixed coniferous-deciduous forests comprise the majority (68%) of the pre-settlement land cover in local lake catchments north of the forest tension zone. The percentage of wetlands in the local catchments of the northern and southern lakes is approximately equal.



**Figure 3.4:** Map of study lakes and the forest tension zone (after Hupy and Yansa 2009a). Fourteen lakes are located south of the forest tension zone's center mark; thirty four lakes are located north of the forest tension zone's center mark.



Figure 3.5. Land cover in local catchment of lakes south of the forest tension zone.



Figure 3.6. Land cover in local catchment of lakes north of the forest tension zone.

### 4. METHODS

#### 4.1 Lacustrine Core Sample Collection

As discussed in Chapter 3, the lake sediment samples analyzed in this thesis came from two sources, however, the water data collection methods and analysis techniques performed on these two sets of samples are comparable and of high quality. In both cases, the deepest part of each lake was determined using bathymetric maps or a hand-held sonar unit and sampling locations were recorded with a GPS device. The methods used to core the sediments of the sampled lakes are also similar.

Sediment cores from the Soranno/Garrison dataset were extracted from each lake in August and September 2007 using a gravity corer with a barrel that fits a 6.8-cm diameter plastic tube that can be replaced to take multiple cores. The length of sediment cores varied between the lakes sampled, ranging from 28 to 69 cm below the sediment-water interface. That same summer, EPA personnel collected sediment cores from lakes included in the NLA using a modified K-B gravity corer, a piece of equipment specially designed for collecting the topmost lake sediments (Dixit et al. 1999; USEPA 2007a). The core lengths recovered by the NLA project varied from 23 to 64.5 cm between sites in Michigan based on substrate permeability (Dixit et al. 1999; USEPA 2007a).

Of the sediment cores taken from 63 lakes in Michigan by WDNR research scientists and EPA personnel, only the topmost 1-2 cm section ("top," capturing the sediment-water interface) and the bottommost 1-3 cm ("bottom") section of each sediment core were kept and analyzed the rest of the core was discarded. The "top" of each sediment core, thus, is interpreted as modern time (the last 2-3 years) and the "bottom" represents an earlier timeframe, the age of which is undetermined until the sediments are dated using absolute or relative techniques. This "top-

bottom" approach to sediment sampling (Figure 4.1) can be used in paleolimnological studies as an alternative to stratigraphic core analyses when the goal is to compare current and past water quality parameters (Cumming et al. 1992; Dixit et al. 1999; Smol 2002; USEPA 2010). Although a centimeter-by-centimeter analysis of the entire sediment core can provide more detailed information for temporal trends in lake water conditions, these studies are often costly and time-consuming, and therefore unrealistic for large-scale studies with many lakes (Smol 2002). Rather, sediment samples from two discrete points in time, before (bottom) and after (top) major human impact, provide enough information to determine the degree of change, as exemplified by the results obtained in this thesis.



**Figure 4.1:** A "top-bottom" approach to core sampling (left) yields two sediment samples, representative of pre-settlement and modern conditions. The numerous sediment intervals from "stratigraphic" sampling (right) provide a continuous analog of lake conditions (modified from USEPA 2010).

In addition to the sediment core collection, both the Soranno/Garrison and the EPA projects collected water column chemistry measurements at the deepest part of each lake to be used in the development of calibration data sets. Specifically, the physical data obtained

included water depth, temperature, and transparency, which was measured with a Secchi disk. The collected chemical/biological parameters included total phosphorus, total nitrogen, alkalinity, specific conductance, chlorophyll, and pH.

# **4.2 Diatom-Inferred Phosphorus Modeling**

# 4.2.1 Soranno/Garrison Methods

The top and bottom intervals of sediment cores collected from 31 lakes in the Soranno/Garrison dataset were kept chilled until delivery to a Wisconsin Department of Natural Resources (WDNR) laboratory in Monona, Wisconsin, at which point diatom analyses were conducted under the direction of Paul Garrison. Samples were placed into tall beakers and cleaned with strong oxidizing agents (i.e., hydrogen peroxide and potassium dichromate) to remove organic matter (van der Werff 1956; Garrison and LaLiberte 2010). The cleaned samples were dried and mounted onto glass slides to be counted under a microscope with an oil immersion objective (1400X) until a total of 400 diatom frustules were identified and enumerated (Garrison and LaLiberte 2010).

Water column chemistry and modern diatom assemblages in the uppermost 1-2 cm of the lacustrine cores from 31 lakes in Michigan and 152 lakes in Wisconsin were collated to create a surface-sediment calibration set. Relationships between modern diatom assemblages and limnological variables in the calibration lakes were examined using a canonical correspondence analysis (CCA) in the statistical program CANOCO (ter Braak and Šmilauer 1998). This multivariate statistical technique is appropriate when diatom species have a unimodal response curve with respect to environmental gradients (ter Braak 1986; Birks 1998). The tested limnological variables included lake surface area, maximum depth, mean depth, alkalinity,

summer mean phosphorus, summer mean nitrogen, summer mean Secchi depth, and water color (Garrison pers. comm. 2012).

#### 4.2.2 EPA Methods

The top and bottom sections of the EPA sediment cores were stored on ice until delivery to an algal ecology laboratory located in the Department of Zoology at Michigan State University in East Lansing, Michigan, where diatom analyses were performed by Dr. Jan Stevenson and his assistants. They prepared samples for analysis following a standard digestion procedure, and transferred the prepared material to glass microscope slides (USEPA 2007b). A minimum of 500-600 valves per sample were identified and enumerated by diatom taxonomists under research quality microscopes at 1000X magnification. The relative abundances of all taxa were calculated in order to reconstruct changes in the study lakes (USEPA 2007b; Stevenson et al. 2013).

A calibration set was developed by Stevenson and colleagues using modern diatom assemblages from the uppermost core sediments and the chemical variables measured in the water column of each lake (USEPA 2010; Stevenson 2013). The calibration set was examined with a detrended correspondence analysis (DCA) to determine if a linear-based or an unimodalbased ordination was appropriate for the environmental gradient based on the observed diatom species data (Smol 2002; USEPA 2010). Unimodal statistical methods were selected because the length of the environmental gradient was > 2.5 standard deviation units (SD) (USEPA 2010). A gradient length less than 2.5 SD would have indicated a relatively narrow range of variation in the diatom assemblage for which a linear, principal components analysis (PCA) approach would have been appropriate.

# 4.3 Relative "Dating" of Sediment Using Pollen

# 4.3.1 Background

Stratigraphic levels of a sediment core are commonly dated using radioisotopes <sup>210</sup>Pb and <sup>137</sup>Cs in site-specific reconstructions of limnological changes (e.g., Garrison and Wakeman 2000; Umbanhowar et al. 2003; Moser et al. 2010). In fact, producing <sup>210</sup>Pb and <sup>137</sup>Cs curves and fitting them to the known dates of events (i.e., banning of leaded gasoline beginning in 1973 and atomic bomb detonations in the 1950s and 1960s) is the only method able to determine an "absolute date" of historic-age sediments. Radiocarbon dating is not reliable, because the youngest age for this method is ~ 200 cal yr BP, due to the longer half-life of this isotope (Björk. and Wohlfarth 2001). However, analyzing sediment profiles for <sup>210</sup>Pb and <sup>137</sup>Cs in regional studies involving many lakes is not feasible as these are very costly dating procedures (Umbanhowar et al. 2003). Moreover, this technique requires the analysis of <sup>210</sup>Pb and other isotopes from many stratigraphic levels per core to assign ages for these levels, which is impossible in this study given that only top and bottom samples were retained.

Although sediment samples in this study were not dated using radioisotopes, I employed a pollen-based, relative dating method identical to those used elsewhere in paleoenvironmental studies. Fossil pollen preserved in lake sediments are useful proxy indicators of dates relative to known historical events that altered land cover composition (e.g., Euro-American settlement) in temperate watersheds of North America (McAndrews 1988; Blais et al. 1995; Dixit et al. 1999). Events such as land clearance and agricultural expansion, in particular, are correlated with a distinct "rise" in *Ambrosia* (ragweed) pollen (Finkelstein et al. 2005; Jaques et al. 2008). Most studies that use this technique compare continuous, stratigraphic levels of sediment core pollen assemblages to find the interval of abrupt *Ambrosia* increase, that has shown to correspond to <sup>210</sup>Pb dates of 150±50 yr BP (Reeder and Eisner 1994). Such studies from the Midwestern U.S. have documented *Ambrosia* percentages to increase from around 5% (prior to wide-spread disturbances) to around 18% at the time of settlement (Grimm 1983). This relative dating is sufficient in comparative limnological studies where only top and bottom sediment core intervals are examined. However, relative dating methods cannot replace "absolute" dating techniques that provide calibrated ages for studies that require high resolution, detailed reconstructions.

Therefore, I analyzed pollen grains preserved in the bottom 1-3 cm interval of each lake core to verify that it has conservatively low (1-3%) values of *Ambrosia* pollen, which would be expected for sediments dating before Euro-American land clearance and agriculture (Grimm 1983). Grimm (1983) had a higher cutoff value (5%), however, his site was farther west at the forest-prairie boundary in Minnesota, a region with naturally higher *Ambrosia* percentages. Therefore, I hypothesized that if any bottom-core sediment from the sampled lakes contained >3% *Ambrosia* pollen values, then that sample did not "date" to pre-settlement times, but rather came from sediments deposited after Euro-American settlement and would be excluded from the statistical analysis of diatom-inferred phosphorus data.

# 4.3.2 Laboratory Methods

Pollen analysis of 63 lake core-bottoms (34 from the EPA dataset and 29 from the Soranno/Garrison dataset) was conducted at Michigan State University's Palynology Laboratory to determine the abundance of *Ambrosia* pollen. A multi-step sample preparation process was used that concentrated pollen grains through chemical and physical removal of extraneous matter (Faegri and Iverson 1975). Centrifuging in swinging buckets at 2900 rpm concentrated the samples after each step.

First, a consistent volume (approximately  $1 \text{ cm}^3$ ) of each sediment sample was treated with 10% hydrochloric acid (HCl) to dissolve calcium carbonates. This step was repeated as necessary for reactive, marl sediments until carbonates were effervescently removed. Next, sodium pyrophosphate (Na<sub>4</sub>P<sub>2</sub>O<sub>7</sub>) was added and the test tubes were placed in a hot bath to suspend clay particles and prevent coagulation during centrifuging. Samples were then sieved through 125 µm metal screens to remove coarse inorganic and organic particles from the fine particles that include pollen grains. In the next step, samples were treated with 10% KOH and placed in a hot bath to remove humic acids. After the KOH was decanted, the samples were made acidic through a 10% HCl rinse and chemically treated with 48% hydrofluoric acid (HF) in a hot water bath to dissolve siliceous matter. The HF treatment was followed immediately by another 10% HCl rinse to remove the dissolved silicates (i.e., colloidal silicates and silicoflourides). Samples were then dehydrated with glacial acetic acid ( $C_2H_4O_2$ ) before adding an acetolysis mixture of concentrated sulfuric acid (H<sub>2</sub>SO<sub>4</sub>) and acetic anhydrite ((CH<sub>3</sub>CO)<sub>2</sub>O). During acetolysis, samples were boiled for 1.5 - 2 minutes to clean pollen exines of cellulose and fine organic residue. This reaction was immediately neutralized with glacial acetic acid. Finally, the samples were dehydrated using ethyl alcohol (ETOH) and tertiary-butanol ((CH<sub>3</sub>)<sub>3</sub>COH), and then transferred to a 1-dram vial along with non-volatile silicon oil.

The concentrated pollen specimens were mounted on glass slides with additional silicone oil and examined using a LEICA DMLB light microscope at 400x magnification. One hundred

pollen grains were enumerated per slide and classified into four categories: *Ambrosia*, deciduous taxa, coniferous taxa, and maize (corn). *Ambrosia* was distinguishable from other tricolporate (C<sub>3</sub>P<sub>3</sub>) pollen types by its short spines and small, oblate grains (Moore and Webb 1978). Abundance was expressed as a percentage of the total pollen count per sample.

# 4.4 Spatial Data Processing

Geographic Information Systems (GIS) is an integrated platform for spatial analysis and modeling that provides a framework for the visualization and management of geographic data (Maguire et al. 2005). The emergence of GIS within the last few decades has revolutionized natural resource management and monitoring of land use/land cover changes across spatial and temporal scales, in part because it is well developed for simultaneously depicting discrete and continuous fields (Turner et al. 2001; Gergel and Turner 2002; Maguire et al. 2005; Johnson and Host 2010). For example, it is now possible to detect and quantitatively analyze vegetation changes within a delineated area or measure the amount of runoff within urban boundaries using a digital mapping program. Furthermore, high quality geospatial data, such as digital elevation models (DEMs), surface hydrology networks, and mosaicked digital raster graphs (DRGs), are now widely available from government agencies, free of charge.

The native land cover, local lake catchment, and inland lake data used in this analysis were acquired from the Michigan Center for Geographic Information (CGI) and the MSU Fisheries and Wildlife Department. Spatial analyses of land cover data were performed in ArcGIS 10 software using basic analytical tools (ESRI 2011). All spatial data were defined and projected into a common coordinate system, North\_American\_1983 Hotine Oblique Mercator.

### 4.4.1 Acquisition and Description of Data

A flow-chart of the data acquired and analysis performed in GIS is shown in Figure 4.2. First, I imported a vector shapefile of the Michigan's native landscape circa mid-1800s into ArcMap. This file contained pre-settlement vegetation details such as the species composition of forests and the location of wetlands in Michigan, as documented in the original General Land Office (GLO) survey notes of 1816-1856 (Comer et al. 1995; Albert et al. 2008). The Michigan Natural Features Inventory (MNFI) earlier had translated these field notes alongside expert interpretation of soils, climate, geology, and landforms into a digital map of over 74,000 polygons with attribute data such as the location, area, and type of native land cover in Michigan (Comer et al. 1995; Kost et al. 2007; Albert et al. 2008). Polygon land cover attributes were classified into 31 unique categories based on the dominant vegetative community or the type of barren land. For simplification purposes, I reclassified these categories into seven groupings to reflect Anderson's Level I and II framework of a national land use and land cover classification system (Anderson et al. 1976).



**Figure 4.2:** Flow chart of GIS methods. A dotted line around a box indicates a task performed, whereas a solid line identifies a coverage or product.

Next, I imported a vector shapefile of Michigan inland lake boundaries and selected the 48 lakes represented in my dataset from the attribute table. Geographic coordinates were used as supplementary identifiers because the abundance of lakes in Michigan has generated many repeat names. The selected lakes were then exported using 'Data Export' to create a new shapefile containing only the study lakes. Three study lakes were not included in the original shapefile were digitized based on existing watershed delineations and DRG contour lines and merged with the new study lakes shapefile. I then edited the lake polygon vertices, if necessary, to reflect the hydrologic boundaries in the native land cover layer; this was to account for perimeter changes that may have occurred since pre-settlement.

### 4.4.2 Data Analysis

Pre-settlement land cover data surrounding each study lake was quantified in the context of a 100-m buffer and the local lake catchment to be used in a spatial comparison of the influence of land cover on lake phosphorus at different scales. First, I used buffer analysis to delineate a shoreline corridor at a fixed-width of 100 m around each lake polygon boundary. I then clipped the land cover data with the 100 m buffers. The attribute data were extracted from the clipped land cover data and percentages of land cover classes were calculated by dividing the area of each cover type by the total land cover area in the 100 m buffer (excluding the lake itself).

Next, I imported a shapefile containing local lake catchment boundaries for inland lakes in Michigan. A lake catchment encompasses the area on the landscape that drains to the focal lake, including land that drains into inflowing streams. These catchment boundaries were delineated in GIS by creating or readjusting nodes of existing line features compiled from the

USGS DRG, MDNR 24k watershed boundary lines, and the Institute for Fisheries Research (IFR) 100k lake catchment boundary lines. Similar to the 100 m buffer extent, I clipped the land cover with the local catchment delineations for each lake. Finally, I imported these data into an Excel spreadsheet and calculated the proportions of total land cover for each study lake.

### **4.5 Statistical Data Analysis**

Quantifying the relationship between spatial landscape patterns and key biophysical drivers is often of primary interest in ecological research (Wiens 2002). Multiple regression analysis is a multivariate statistical technique commonly applied in these investigations of complex ecological system dynamics because it allows for the determination of a single dependent response variable against several explanatory variables (Clark and Hosking 1986; Devito et al. 2000; Webster et al. 2008). Furthermore, multiscale analyses based on multiple regressions can test the explanatory power of a set of variables at different scales, thus allowing ecological responses to be examined in regards to habitat configuration and spatial landscape patterns (Turner et al. 2001). The goal of regression modelling is to find the explanatory values that minimize the difference between predicted and observed values of the dependent variable. The mathematical equation for a linear regression equation takes the following form:

 $y = X\beta + e$ 

Where:

y is *n* observations of a dependent variable; X is a *n* by *k* matrix of observations on predictor variables  $\beta$  is the standardized regression coefficient for X; and e is the observed error term.

Developing a regression model with low error terms that explains landscape contributions to lake phosphorus can be challenging because of regional variability in interactions among biological, geochemical, and climatic variables. Regardless, models are important tools in landscape limnology, as it is necessary to simplify these interactions to define problems more precisely and concepts more clearly (Turner et al. 2001). Therefore, simple linear and multiple regressions were used in this study to explore the relationship between diatom-inferred phosphorus concentrations and pre-settlement landscape variables.

My research used a quasi-experimental (non-randomized) design to examine these relationships. Unlike true-experimental designs where the minimum sample size required is controllable, these designs can be used when it is not logistically feasible to conduct a randomized, full manipulation of the dataset parameters (Harris et al. 2004). Therefore, a power analysis, which normally determines the size of the sample needed, was not performed because the data used in this research was predetermined and collected by others.

# 4.5.1 Summary of Potential Predictor Variables

Exploratory data analysis of potential predictor (independent) variables in conjunction with the dependent variable (pre-settlement, diatom-inferred total phosphorus concentrations) is a way to maximize the amount of information extracted from a dataset, formulate new hypotheses, and identify the most important predictor variables (Behrens and Yu 2003). Automated selection procedures, a method of exploratory analysis that relies on various algorithms to iteratively identify predictor variables, can be useful when the researcher has no basis for formulating a planned analysis (Marona et al. 2006; Hession and Moore 2011). However, this technique has many pitfalls, including multicollinearity and confounding between

predictor variables, which can yield inaccurate results (Hession and Moore 2011). To avoid confounding, I specified hypotheses-driven independent variables relevant to the dependent variable, based on the existing breadth of knowledge, before running my analysis. This method allowed the correlation among independent variables to be evaluated and described before developing the regression model. Thus, the first step to predicting patterns in past phosphorus concentrations was to identify an initial set of potential predictor variables, *a priori*. The following land cover, hydrogeomorphic (HGM) and geographic variables were evaluated prior to the final model selection procedure (Table 4.1).

**Table 4.1:** Potential landscape parameters hypothesized to explain variations in pre-settlement phosphorus concentrations ( $\mu$ g/L).

Potential Predictor Variables			
% Native Land Cover			
Deciduous forest*			
Coniferous forest*			
Mixed forest*			
Total forest*			
Wetland*			
Grassland*			
HGM Features			
Wetland/lake ratio*			
Maximum lake depth			
Lake area			
Lake perimeter			
Lake area/depth ratio			
Catchment/lake ratio			
Geographic Variables:			
Latitude			
Longitude			
Forest Tension Zone			
*tested at 100 m buffer and local			

catchment scale

Considerable attention has been given to statistical modeling of land cover effects on water quality, although it is often done in light of anthropogenic drivers that affect phosphorus export rather than natural landscape factors. Many studies that examine nutrient export suggest sediment accumulation and phosphorus loading monotonically change relative to the amount of agricultural land in a watershed (Dillon and Kirchner 1975; Gémesi et al. 2011). However, some studies do give merit to natural controllers of phosphorus variation. For example, Allen et al. (1997) demonstrated that nutrient yields and sedimentation rates decreased proportionally to forested land cover in modeled watersheds. Therefore, I hypothesize that lakes with greater proportions of aggregated forest cover (deciduous, coniferous, and mixed) will have lower presettlement phosphorus concentrations.

Additionally, there is much debate as to whether wetlands act as nutrient sinks or sources (Richardson 1985). Much of the published research within the last two decades has largely focused on the use of constructed wetlands as a nutrient sink in the wastewater treatment process. However, although wetlands can remove and store large amounts of nutrients, they can also release nitrogen and phosphorus to the adjacent aquatic ecosystems. Once a nutrient enters a wetland, it is stored, altered, or discharged, depending on the complex biotic and abiotic mechanisms involved. My hypothesis is that in the absence of anthropogenic phosphorus contributions (i.e., agricultural and urban runoff), wetland ecosystems provide the essential nutrients needed to sustain primary productivity in lakes.

In developing the regression model, I hypothesized that phosphorus concentrations had a negative linear relationship with lake depth, based on existing research involving this abiotic variable. For example, in a series of lake management models, Håkanson (1996) demonstrated a significant positive relationship between lake depth and Secchi depth, an indicator of

limnological conditions. This was because sediments in deeper lakes (with more water beneath the wave base) were less likely to be resuspended in the water column. The resuspension of sediments also causes a resuspension of benthic phosphorus, adsorbed to the soil particles (Wetzel 2001). For this reason, total phosphorus concentrations often correspond with water transparency, as measured by Secchi depth (Carlson and Simpson 1996; Kleeberg et al. 2007).

Precipitation and temperature can affect the rate of weathering of phosphorus-containing rocks, as well as the erosion and accumulation of sediments (Chuman et al. 2013). Therefore, three parameters reflective of climate were included as potential predictor variables of phosphorus variability: latitude, longitude, and relative location to the forest tension zone (FTZ) in Michigan. Relative location to the FTZ, a categorical variable, was included in addition to latitude and latitude to capture the approximate divergence in the number of degree growing days between northern and southern Michigan (Andresen et al. 2012). Lakes in areas south of the FTZ, were coded '0,' whereas lakes north of the FTZ were coded '1'.

### 4.5.2 Initial Variable Selection

The *a priori* prediction that these aforementioned independent variables would explain some of the variability in diatom-inferred phosphorus concentrations among the study lakes was tested using bivariate, exploratory data analyses in the statistical computing package, *R* (R Core Team 2013). Simple linear correlation coefficients (*r*) measured the degree and direction of associations between the predictor variables and the diatom-inferred phosphorus concentrations. Values of *r* can range from -1 (perfect negative correlation) to +1 (perfect positive correlation), with increasing significance as the value approaches  $\pm 1$  (Clark and Hosking 1986).

The calculation of correlation coefficients was repeated for each potential predictor variable to obtain a set of correlations, output in the form of a correlation matrix (Appendix B). The correlation matrix was used to assess collinearity, the degree of independence and similarity in the variability among the independent variables (Hunsaker and Levine 1995). More importantly, the potential predictor variables were correlated against the dependent variable, historic diatom-inferred phosphorus. The results of these correlation analyses were used to determine which variables explained the most variance in pre-settlement phosphorus as an aid to model selection.

# 4.5.3 Model Selection

Linear regressions based on ordinary least squares (OLS) were performed in R using presettlement diatom-inferred phosphorus concentrations as the dependent variable (y) and the predictor variables identified by the selection process in Section 4.5.2. An OLS equation seeks to predict the value of a dependent variable (y) based on the value of one or more independent variables (x) by minimizing the sum of the squared differences (residual errors) between the observed and predicted values of the dependent variable (Overmars et al. 2003). Seven linear models were run for the predictor variables identified with bivariate, correlation testing to determine which set of theoretically relevant variables provided the most satisfactory model.

These standard linear regression models were based on the assumption that observations were statistically independent of one another (Overmars et al. 2003). However, if this data assumption is incorrect, and spatial autocorrelation does exist within the dependent variable or the residual errors, the regression equation would likely yield biased results (Anselin 1998; Hession and Moore 2011). Therefore, I evaluated spatial dependency with Moran's I, a

conventional measure of autocorrelation that compared the value of a pre-settlement, diatominferred phosphorus concentration at a given location to the value of diatom-inferred phosphorus at all other lake locations (Anselin 1998). Similar to Pearson's Product Correlation Coefficient, the significance level of Moran's I increases as the value approaches -1 or +1. I used this measure to identify the number of nearest neighbors and weights matrix that yielded the most spatial autocorrelation. Using the number of nearest neighbors and the weights matrix identified in the Moran's I step, I ran the model in the statistical platform, GeoDa to administer further diagnostic tests for spatial autocorrelation. Both methods used to evaluate spatial autocorrelation revealed that spatial regression was not necessary (further detail provided in Chapter Five), and an OLS regression models was appropriate for explaining the variation in my dataset. These OLS regression models imply spatial independence amongst pre-settlement lake phosphorus concentrations.

Finally, the candidate OLS models that incorporated historic lake phosphorus data and landscape parameters were evaluated by minimizing Akaike's information criterion (AIC) in *R*. This method of model selection, which uses maximum likelihood measures of the fit of a model (Burnham and Anderson 2004), can be defined as:

### AIC = 2k-2ln(L)

Where:

k is the number of parameters in the model;

and L is the maximum likelihood.

This statistical model selection technique replaces the mean squared error of prediction as a measure of model "badness" (Akaike 1981). In the model selection process, lower AIC values indicate a better model. The final model, best fit to the data variability, was an aspatial OLS regression with the relatively lowest AIC values of all models tested.

# **5. RESULTS AND INTERPRETATIONS**

This chapter presents the results derived from the basic spatial and statistical analyses of this study. First, phosphorus concentrations representative of pre-settlement conditions were inferred from transfer functions applied to diatom assemblages in the bottom interval of lacustrine sediment cores. Relative dating of sediments using *Ambrosia*-type pollen verified that the concentrations are reflective of undisturbed lake conditions, prior to Euro-American settlement. Quantitative measures of pre-settlement land cover, as interpreted from GLO survey data, were calculated and analyzed at two scales to be used as predictor variables in statistical modeling. Lastly, the relationship between potential predictor variables and diatom-inferred phosphorus concentrations in lakes prior to Euro-American land clearance and settlement was examined using bivariate and multivariate regression analyses.

### 5.1 Results of Diatom Ordinations and Analyses

# 5.1.1 Results of Soranno/Garrison Dataset Analysis

Results of the ordination analyses conducted by the WDNR on the Soranno/Garrison calibration dataset revealed that distributions of diatom taxa in surface sediments were most correlated with total phosphorus and alkalinity (Garrison pers. comm. 2012). The CCA ordination diagram (Figure 5.1) produced in CANOCO (version 4.1) by the WDNR presents a graphical summary of the patterns of correlation between diatom species variation and water chemistry variables from the calibration set. The vector (arrow) length represents the proportional strength of correlation between the observed environmental variables and diatom assemblages. This information was used to reconstruct diatom-inferred phosphorus concentrations from the bottom core assemblages. Error estimates in inferences were derived by the WDNR using the bootstrap error estimation approach in the statistical computing program
WACALIB version 3.3 (Line et al. 1994). The root mean squared error of prediction (RMSEp) for summer mean phosphorus is log 0.29  $\mu$ g/L (Garrison pers. comm. 2012).



**Figure 5.1:** Canonical correspondence analysis (CCA) ordination diagram of speciesenvironment relationship in the 187 lake calibration dataset generated by the WDNR with the forward-selection option and Monte Carlo permutation tests in CANOCO (version 4.1) (ter Braak and Šmilauer 1998; Garrison pers. comm, 2012).

A summary of the observed and inferred phosphorus concentrations from the top (modern) and bottom (pre-settlement) sediment core samples of the 31 lakes (Soranno/Garrison dataset, prior to relative dating) are shown in Table 5.1. The mean of reconstructed phosphorus concentrations of the 31 lake core-bottom samples was 11.61  $\mu$ g/L. These bottom-core samples are representative of oligotrophic lake conditions (reconstructed phosphorus ranged from 5.52  $\mu$ g/L to 17.92  $\mu$ g/L). Reconstructed diatom-inferred phosphorus increased by a mean of by 0.47  $\mu$ g/L from the core bottom to the core top.

**Table 5.1:** Summary statistics of reconstructed phosphorus (inferred from diatom assemblage) and observed phosphorus (measured in water column) from the Soranno/Garrison lake dataset (n=31). Diatom-inferred total phosphorus concentrations from the bottom (DI-TP Bottom) and top (DI-TP Top) core intervals are in units  $\mu$ g/L.

				<b>Δ DI-TP Top</b>	Δ  Inferred TP
	<b>DI-TP Bottom</b>	DI-TP Top	<b>Observed TP</b>	& Bottom	& Observed TP
Mean	11.61	12.08	13.26	0.47	4.83
Min	5.52	3.67	2.02	-7.63	0.28
Max	17.92	44.39	32.88	26.47	19.21
Median	10.94	11.50	11.02	-0.76	3.03
Range	12.40	40.72	30.86	34.10	18.93

## 5.1.2 Results of EPA Dataset Analysis

The results of the EPA's CCA ordination (and associated Monte Carlo permutation test) of the 42 sampled lakes (undated samples) suggested statistically significant relationships (p = 0.001) between the surface sediment diatom species of the national calibration dataset and total phosphorus ( $\mu$ g/L), total nitrogen ( $\mu$ g/L), conductivity ( $\mu$ S/cm at 25° C), and pH (USEPA 2010a). The total phosphorus transfer function developed by EPA operatives using weighted averaging partial-least-squares (WA-PLS) models provided a robust reconstructive relationship.

Evaluations of transfer function strength, characterized by squared correlation coefficient ( $r^2$  =0.74), involved a comparison of diatom-inferred total phosphorus from the surface sediments to total phosphorus measured from the water column. Moreover, jackknife (leave-one-out) cross-validation was used for more realistic error estimation ( $r^2$  jacknife=0.67; RMSEp=0.36 log (µg/L)) (USEPA 2010).

Unlike the results of the 31 Soranno/Garrison dataset, mean diatom-inferred phosphorus concentrations from the 42 EPA lakes decreased from the bottom (27.34  $\mu$ g/L) to the top (18.49  $\mu$ g/L) of the cores (Table 5.2). At first glance, these findings could be misinterpreted to mean that natural levels of phosphorus in Michigan lakes were higher prior to Euro-American settlement. In actuality, these relatively high phosphorus concentrations results could indicate that the bottom-core depths are not representative of natural, pre-settlement conditions. In fact, the range of diatom-inferred phosphorus concentrations from the undated sediment core-bottoms of the EPA lakes (2.29  $\mu$ g/L to 286.03 $\mu$ g/L) is considerably greater than the range from the Soranno/Garrison core-bottoms (5.52  $\mu$ g/L to 17.92  $\mu$ g/L) (Table 5.2). This variability in the EPA data more likely implies that bottom cores were not taken from a great enough depth, and so the diatom assemblages in the sediments are not reflective of pre-settlement conditions, but are of historic times when anthropogenic impacts were greater. Fortunately, the relative dating technique based on *Ambrosia* pollen that was used in this study, and discussed in Section 5.2, was able to substantiate this hypothesis.

**Table 5.2:** Summary statistics of reconstructed phosphorus (inferred from diatom assemblage) and observed phosphorus (measured in water column) from the EPA lake dataset (n=42). Diatom-inferred total phosphorus concentrations from the bottom (DI-TP Bottom) and top (DI-TP Top) core intervals are in units  $\mu$ g/L.

				Δ DI-TP Top	∆  Inferred TP
	<b>DI-TP Bottom</b>	DI-TP Top	<b>Observed TP</b>	& Bottom	& Observed TP
Mean	27.34	18.49	14.36	-8.85	4.13
Min	2.29	2.88	3.00	-274.97	0.03
Max	286.03	118.22	65.00	35.53	86.22
Median	13.26	10.65	8.50	-1.32	4.83
Range	283.74	115.34	62.00	310.50	86.19

### 5.1.3 General Discussion of Diatom-Inferred Data

Water chemistry reconstructions from calibration datasets require sufficient knowledge about the factors controlling relative dominance of a diatom species. However, despite strong correlations between measured environmental variables and surface sediment assemblages, calibration datasets may not include all the significant limnological factors driving patterns in species composition (Fritz et al. 1999). It is also possible to introduce error into diatom inferences from potential within-lake variability of sediment diatom assemblages or taxonomic inconsistencies between the top and bottom core sediments (Hall and Smol 1999). Comparison of diatom-inferred epilimnetic total phosphorus and water column total phosphorus are therefore necessary to determine the predictive ability of the model despite these obstacles that can hinder accurate inferences of pre-settlement total phosphorus.

# **5.2 Pollen Analysis Results**

I analyzed all available and suitable bottom-core samples from the Soranno/Garrison and EPA datasets for *Ambrosia* pollen, which, when in high abundance, marks the approximate

timeframe of Euro-American settlement in the Midwest. Of the 85 bottom-cores samples (54 EPA and 31 Soranno/Garrison), 22 were excluded from the pollen analysis for various reasons including: lab data for sediment sample did not exist, the sediment samples were unavailable for relative dating (i.e., discarded or misplaced), low confidence levels in the accuracy of the diatom-inferred phosphorus data reflecting pre-settlement conditions (this index was unique to the EPA samples), or the sample was duplicated in both Soranno/Garrison and EPA datasets (Appendix A). The results of my palynological investigations of the 63 Soranno/Garrison and EPA bottom-core samples are shown in Table 5.3 and Table 5.4.

Ambrosia was found in 75% of the samples. The amount of Ambrosia pollen per sediment sample ranged from 0 to 24%, with a mean of approximately 3%. In addition to the 22 excluded lakes, I eliminated 15 lakes from the final dataset of study lakes because the bottomcore sediments contained  $\geq$  4% of *Ambrosia*-type pollen, which is interpreted as above the natural (pre-settlement) value of ragweed in the forests of Michigan. Based on this criterion, I found 48 (out of 63) lake core-bottoms to be representative of pre-settlement conditions. In all samples, total arboreal deciduous and coniferous pollen percentages exceeded that of *Ambrosia*. Lake samples north of the FTZ were dominated by pollen grains of a mix of deciduousconiferous species, whereas the pollen spectra in samples south of the FTZ in Michigan were comprised mainly of deciduous species. Although attempts were made to count maize pollen as further evidence of Euro-American settlement and agricultural expansion, none were found, but this is not unexpected as corn pollen is poorly dispersed and is rarely recovered in pollen studies of lakes (McAndrews and Turton 2010).

Lake Name	County	<b>Bottom Core</b>	Sample Depth (cm)	Ambrosia	Deciduous	Coniferous
	-	DI-TP (µg/L)				
Bartholomew	Branch	14.63	44-46	1	86	13
Bear	Manistee	9.84	48-50	2	45	53
Big Blue	Kalkaska	11.30	30-31	0	56	44
Big Lake	Osceola	10.78	43-45	0	52	48
Big Star	Lake	10.04	67-69	0	36	64
Bobcat	Gogebic	14.52	48-50	1	51	48
Cadillac	Wexford	10.25	33-35	1	31	68
Campau	Kent	10.53	32-34	2	70	28
Coldwater	Branch	10.83	36-38	2	91	7
Craig	Baraga	9.89	54-56	1	28	71
Deer	Charlevoix	12.06	46-48	1	48	51
Duck	Gogebic	10.40	50-52	1	44	55
Eagle	Kalamazoo	12.55	31-33	2	84	14
Hanley	Antrim	14.03	37-40	1	69	30
Imp	Gogebic	5.52	49-51	0	28	72
Mitchell	Wexford	15.52	40-42	0	37	63
Little	Marquette	10.14	50-52	0	65	35
Murray	Kent	12.90	46-48	1	72	27
Ottawa	Iron	10.02	45-47	0	47	53
Pleasant	Jackson	11.11	28-30	1	10	89
Starvation	Kalkaska	7.92	39-41	0	46	54
Teal	Marquette	13.00	50-52	2	29	69
Twin	Kalkaska	9.81	46-48	0	44	56
West Twin	Montmorency	7.80	46-48	0	9	91
Whitefish	Montcalm	10.17	32-34	0	67	33
Witch	Marquette	12.55	55-57	0	64	36
Reeds	Kent	17.92	38-40	10	58	32
Portage	Manistee	13.06	40-42	10	52	38
Townline	Montcalm	10.94	56-58	11	46	43

**Table 5.3:** Results of the pollen analyses for 29 bottom-core samples from the Soranno/Garrison dataset. The italicized rows located at the bottom of the table are representative of non-pre-settlement conditions.

Lake Name	County	DI-TP (µg/L)	Sample Depth (cm)	Ambrosia	Deciduous	Coniferous
Au Sable	Ogemaw	2.92	41.5-42.5	0	63	37
Bailey	Keweenaw	11.98	28-30	0	35	65
Blomgren	Dickinson	41.18	27-32	2	37	61
Blue	Mecosta	6.32	44-45	0	41	59
Campbell	Kalamazoo	2.29	53.5-64.5	2	85	13
Chemung	Livingston	8.34	37.3-38.3	2	81	17
Clark	Jackson	9.46	36-39	0	91	9
Dewey	Marquette	5.99	59-61	0	15	85
Eight Point	Clare	6.89	54.5-55.5	0	36	63
Fence	Baraga	11.64	56-61	3	36	61
Gogebic	Gogebic	14.71	26-28	1	50	49
Hall	Houghton	12.63	34-35	2	56	42
Howe	Alger	14.79	56-58	0	65	35
Kathryn	Iron	19.54	56-57	0	37	63
Keewaydin	Baraga	9.83	37-39	0	38	62
Lotto	Marquette	10.56	40-42	0	47	53
McDonald	Delta	15.87	58-61	1	52	47
Mill	Oakland	9.03	41.5-42.5	0	76	24
Mud	Houghton	12.72	23-25	2	36	62
Pine	Kent	4.81	38.5-39.5	1	53	46
Stoner	Kent	16.28	52-53	1	73	26
Warner	Barry	32.23	54.5-55.5	0	84	16
Saddle	Van Buren	16.04	49-50	8	69	23
Big	Baraga	14.03	31-33	4	31	65
Bogie	Oakland	9.07	54-55	14	74	12
Crooked	Emmet	5.11	33.5-34.5	9	56	35
Martin	Newaygo	18.01	31.5-32.5	24	41	35

**Table 5.4:** Results of the pollen analyses for 34 bottom-core samples from the EPA dataset. The italicized rows located at the bottom of the table are representative of non-pre-settlement conditions.

$1 a \mu \mu \cup J = 1 ( \cup \mu \cup \mu \cup \mu)$	Tab	le 5.4	(cont'd)
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Muskegon	Muskegon	70.25	46-47	15	78	7
Pere-	Mason	65.33	26.5-27.5	8	70	22
Marquette						
Round	Van Buren	8.35	37.5-38.5	11	78	11
Tallman	Mason	12.69	42-43	8	25	67
Tims	Jackson	7.08	49-51.5	17	76	7
Upper Scott	Allegan	21.40	34.5-35.5	14	48	38
Wyckoff	Oceana	50.33	36.5-37.5	13	61	26

#### 5.2.1 Correlation of Ambrosia with Depth

The abundance of *Ambrosia* pollen from my analysis was statistically examined for trends that could provide insight for future paleolimnological investigations. First, I tested the correlation between sediment core-bottom depths and *Ambrosia* pollen counts to determine if a high percentage of *Ambrosia* in the pollen spectra generally corresponded to relatively short lake cores. This would provide some explanation as to the why 35% of sediment cores from the EPA dataset and 10% of the sediment cores from the Soranno/Garrison dataset were not representative of pre-settlement conditions.

However, the results of simple linear correlation test (Figure 5.2) indicated a low association between the two variables (r = -0.120). Regardless, the negative correlation indicates a slight tendency for *Ambrosia* counts to decrease as the core depth increases. This suggests that the deeper cores are generally more successful in representing lake sediments that correspond to a time period before anthropogenic landscape disturbances. Although the direction of this relationship was expected, I anticipated a stronger correlation between bottomcore depth and the percentage of *Ambrosia* pollen. It is possible that variations in sedimentation rates, due to regional or localized differences in erosion or lake productivity, is confounding what might otherwise be a simple and linear relationship.



**Figure 5.2:** Correlation plot between the *Ambrosia* counts and the depth (cm) of the bottom-core interval (r = -0.120).

### 5.2.2 Correlation of Ambrosia with Total Phosphorus

I also conducted a bivariate correlation test to determine statistical relationship between *Ambrosia* pollen counts and pre-settlement total phosphorus concentrations from the bottom-core samples. The relationship between *Ambrosia* and phosphorus concentrations (Figure 5.3) was stronger (r = 0.3635) than the relationship with sediment core depth (r = -0.120). Additionally, the relationship was positive, which means that sediments with relatively higher percentages *Ambrosia* pollen had relatively higher total phosphorus concentrations. This result was expected, as phosphorus concentrations are known to be higher in lakes of disturbed watersheds (such as those initiated by Euro-American settlement) (e.g., Rast and Lee 1978; Omernik et al. 1981).



**Figure 5.3:** A scatter plot of the bivariate correlation testing results between diatom-inferred total phosphorus (DI-TP) in the bottom sediment cores and the total *Ambrosia* pollen count from lakes in the Soranno/Garrison and EPA datasets (r = 0.3635). An 'X' denotes sediment intervals that were interpreted to be representative of pre-settlement conditions. Conversely, an 'O' denotes post-settlement sediment intervals because *Ambrosia*-type pollen comprises > 3% of the total pollen spectra.

#### 5.2.3 Phosphorus Reconstructions from Study Lakes

The summary statistics of the final dataset of pre-settlement sediment samples from 48 lakes (Table 5.5) are noticeably different from the original EPA data. This is because 34% of the EPA phosphorus reconstructions were eliminated from the final dataset due to > 3% *Ambrosia* pollen in the bottom-core samples. The mean of the final dataset is 15.54  $\mu$ g/L less than the mean of the EPA samples alone. However, the mean of the 31 Soranno/Garrison data (11.61  $\mu$ g/L) more closely resembles the final dataset.

**Table 5.5:** Summary statistics of reconstructed phosphorus (inferred from diatom assemblage) and observed phosphorus (measured in water column) from the "pre-settlement" lake dataset (n=48). Diatom-inferred total phosphorus concentrations from the bottom (DI-TP Bottom) and top (DI-TP Top) core intervals are in units  $\mu$ g/L.

				Δ DI-TP Top	Δ  Inferred TP &
	<b>DI-TP Bottom</b>	DI-TP Top	<b>Observed TP</b>	& Bottom	<b>Observed TP</b>
Mean	11.84	10.64	11.24	-1.20	4.93
Min	2.29	2.88	2.02	-25.04	0.03
Max	41.18	29.65	37.00	24.84	20.86
Median	10.67	9.95	9.96	-1.02	2.96
Range	38.88	26.77	34.98	49.88	20.83

#### **5.3 GIS Analysis Results**

The reclassification of native land cover from the MNFI shapefile resulted in the following seven categories: "Deciduous Forest," "Evergreen Forest," "Mixed Forest," "Wetland," "Surface Water," "Grassland," and "Barren" (Tables 5.6). Based on the total area of the 48 lake catchments in this study, approximately 155,411 square kilometers of native land cover in Michigan's Upper and Lower Peninsulas were analyzed. The GIS analysis from this

Original Land Cover Description	Land Cover Reclassification
Aspen-Birch Forest	Deciduous Forest
Beech-Sugar Maple Forest	
Black Oak Barren	
Mixed Oak Forest	
Oak-Hickory Forest	
Sugar Maple-Basswood Forest	
Sugar Maple-Yellow Birch Forest	
Hemlock-White Pine Forest	Evergreen Forest
Jack Pine-Red Pine Forest	
Pine Barrens	
Spruce-Fir-Cedar Forest	
White Pine-Red Pine Forest	
Beech-Sugar Maple-Hemlock Forest	Mixed Forest
Hemlock-Yellow Birch Forest	
Mixed Pine-Oak Forest	
Oak/Pine Barrens	
Sugar Maple-Hemlock Forest	
White Pine-Mixed Hardwood Forest	
White Pine-White Oak Forest	
Black Ash Swamp	Wetland
Cedar Swamp	
Mixed Conifer Swamp	
Mixed Hardwood Swamp	
Muskeg/Bog	
Shrub Swamp/Emergent Marsh	
Wet Prairie	
Lake/River	Surface Water
Grassland	Grassland
Mixed Oak Savanna	
Exposed Bedrock	Barren
Sand Dune	

**Table 5.6:** Reclassification of pre-settlement land cover categories, as captured by the GLO surveys (Anderson 1976).

research indicated that at the time of the 19<sup>th</sup> century surveys, forests were the dominant land cover at both the 100 m buffer and local catchment (lakeshed) spatial extent (Table 5.7). In local lake catchments, the aggregation of forest classes comprised approximately 82% of the native

land cover composition, compared to approximately 70% of the total land cover within the 100 m buffer. Conversely, wetland cover is at a greater percentage (nearly 29%) in the 100 m buffer than in the local catchment (about 17%). Furthermore, wetlands constituted > 50% of the 100 m buffer of 13 of the study lakes, but only 1 of the local catchments were composed of >50% wetlands. The greater percentage of native wetland cover within the100 m buffer of lakes is not surprising, as wetlands are known to develop in shallow water or areas associated with high water tables within near proximity to lakes, rivers, or streams (Tiner 2003; Mitsch and Gosselink 2007).

Land Cover	% in 100 m Buffer	% in Local Catchment
Deciduous Forest	19.2%	12.1%
Mixed Forest	36.4%	63.6%
<b>Coniferous Forest</b>	15.0%	6.1%
Wetland	28.7%	17.2%
Grassland	0.8%	1.0%
Barren	0%	>1%

**Table 5.7:** Summary of reclassified native land covers in the 100 m buffer and local catchment area.

Although GIS has revolutionized the way ecologists monitor landscape changes across spatial and temporal scales, positional data interpreted from survey notes should be approached with caution (Turner et al. 2001; Gergel and Turner 2002). For example, the GLO surveyors typically recorded landscape observations at limited locations such as along township and section lines. Moreover, the experience and overall thoroughness of deputy surveyors is at times reflected in the precision of the GLO mapping project (Thomas 2009). Potential inconsistencies between USPLS sections of the GLO native land cover map may be caused by other reasons than surveyor error. For example, mapping of wetlands varied with the season of the field work, or whether it was a wet or dry year (Thomas 2009). As a result, the positional accuracy of land cover delineations from in the MNFI digital map may be within an estimated 80-186 m of their actual position (MNFI 2000; Thomas 2009). Unfortunately, coarse resolution of digital maps adds an inevitable degree of uncertainty to any landscape mapping project, particularly when remotely sensed data or aerial imagery is unavailable. Regardless of these caveats, the presettlement digital dataset has been used by scientists and researchers investigating the impact of landscape-scale changes in vegetation patterns (e.g., Hupy and Yansa 2009b).

### 5.4 Statistical Data Analysis Results and Discussion

I used an *a priori* model approach to potential predictor variable selection as an alternative approach to automated model selection procedures (e.g., forward, backward, and stepwise regressions) to avoid inaccuracies introduced by multicollinearity and confounding between the predictor variables. Automated selection procedures that rely on various algorithms for decision making can yield unreasonable results when there are missing values in the potential covariates (Burnham and Anderson 1998; Hession and Moore 2012). Rather, the inherent limitations of automated procedures were avoided through careful selection of hypothesis-driven predictor variables. Only the predictor variables with a significant correlation to historic phosphorus concentrations were selected and evaluated in an OLS regression model.

### 5.4.1 Bivariate Regression Results and Discussion

After ranking the % land cover variables in the 100 m buffer and local catchment according to their correlation coefficients (r), I found that the relationship between prehistoric % wetland in the 100 m buffer demonstrated the strongest correlation (r = 0.269; Table 5.8) with pre-settlement phosphorus concentrations. I then derived a variable that was the combination of land cover and a hydrogeomorphic feature (lake area) in an attempt to reduce multicollinearity

that inevitably occurs when using land cover percentages as variables. I used wetland area within the 100m buffer and local catchment because wetland percentage showed the highest correlation with phosphorus out of all the native land cover variables. The derived variables were the ratio of wetland area to lake area, at two spatial scales. These variables describe the amount of wetlands relative to the size of the lake, which I hypothesized would be more accurate in capturing the influence of land cover on phosphorus variability. I defined wetland/lake ratio as the total wetland area (ha) (in 100 m buffer and local catchment) divided by the total lake area (ha).

Interestingly, the correlation between wetland/lake ratio in the 100 m buffer and presettlement phosphorus concentrations (r = 0.565) was stronger than the correlation of phosphorus with % wetlands (r = 0.269). Furthermore, bivariate analysis between phosphorus and the wetland/lake ratio in the local catchment revealed an even weaker correlation (r = 0.230). This suggests that land cover within near proximity (100 m) to the lake shore may be able to better account for the variability in pre-settlement phosphorus concentrations than land cover in the entire lake catchment.

Highly correlated variables in regression equations can cause mathematical corruption of a model (Van Sickle 2003). As shown in the correlation matrix produced from bivariate correlation analysis (Appendix B), the forest tension zone (FTZ) variable is highly correlated with both latitude and longitude (r=0.826 and -0.531, respectively). Therefore, to reduce collinearity in the models and to filter out weak correlations with phosphorus, I excluded the FTZ variable (r = 0.004) from the model selection procedure. In addition to multicollinearity introduced from redundant variables, land cover percentages are not mutually independent of one another because by definition, they must total 100%. For example, a high wetland percentage in a 100 m buffer inevitably means that land cannot be forested, and as a result, the percentage of

forest in a 100 m buffer is low. For this reason, I chose only one land cover metric (at both spatial scales), wetland/lake ratio, based on correlation coefficient values, to include in the OLS regression analysis (Table 5.8).

Potential Predictor Variable	Spatial Scale (if applicable)		
% Native Land Cover	100 m Buffer	Local Catchment	
Deciduous forest	0.03	0.066	
Coniferous forest	-0.146	-0.185	
Mixed forest	-0.088	0.01	
Total forest	-0.198	-0.132	
Wetland	0.269	0.131	
Grassland	-0.03	0.055	
HGM Features			
Wetland/lake ratio	0.565	0.230	
Maximum lake depth		-0.248	
Lake area		0.042	
Lake perimeter		-0.051	
Lake area/depth ratio		0.074	
Catchment/lake ratio		0.039	
Geographic Variables:			
Latitude		0.101	
Longitude		-0.21	
Forest Tension Zone		0.004	

**Table 5.8:** Pearson's Correlation Coefficients (*r*) of the potential predictor variables with pre-settlement phosphorus concentrations.

Scatter plots of these bivariate analyses between diatom-inferred phosphorus and the continuous predictor variables visually demonstrated strength and direction of relationships (Appendix C). The simple linear regression line on each plot is a graphical representation of the correlation coefficients (*r*) as shown in Table 5.8. However, the results from these bivariate correlation analyses alone are not sufficient to meaningfully declare a level of significance to the relationship between the predictors and the dependent variable (Van Sickle 2003). Therefore, multiple linear regressions were used to further explore trends in pre-settlement phosphorus

variability. The five predictor variables used individually, and then combined, in the model selection process were latitude, longitude, maximum depth, wetland/lake ratio in the local catchment, and wetland/lake ratio in a 100 m buffer.

# 5.4.2 Model Selection Results

Individual predictor variables and combinations of predictor variables (selected based on methods described in Section 5.4.1) were evaluated for fit in seven OLS regression models based on AIC and adjusted  $R^2$  values (Table 5.9). The adjusted  $R^2$  values are an indicator of how well

Model ID	Model Name	AIC	Adjusted R <sup>2</sup>	Notes
1	lat	317.6	-0.0113	Latitude, in decimal degrees
2	long	315.93	0.02317	Longitude, in decimal degrees
3	max_d	315.04	0.04127	Maximum depth as defined in original data collection
4	wetland_ratio_LC	315.19	0.03812	Area of total wetlands in the local catchment divided by the area of lake
5	wetland_ratio_100	299.64	0.3044	Area of total wetlands in the 100 m buffer divided by the area of lake
6	wetland_ratio + max_d	301.24	0.2948	Wetland:lake ratio combined with maximum lake depth
7	wetland_ratio + max_d + wetland_ratio*max_d	298.7	0.3438	Wetland:lake ratio combined with maximum lake depth and an interaction term of wetland:lake ratio x maximum lake depth

**Table 5.9:** Results of model-fitness testing from seven OLS regressions.

the OLS model fits the data; values close to 1.0 indicates that the model accounts for nearly all of the variance in the independent variables. AIC values in this study were meaningful and

comparable between models because the dependent variable was not log or square roottransformed.

Model #7 qualifies as a good-fitting, parsimonious regression model with a (relatively) large  $R^2$  and only three predictor variables which keeps the interpretation simple. Models #1, 2, 3, and 4 yielded poor results, with adjusted  $R^2$  values < 0.1. Models #5 and 6 were similar in adjusted  $R^2$  values to #7, however the AIC values were higher, indicating less fit with the data. Therefore, I chose Model #7 as the final OLS model best fit to explain the variability in presettlement phosphorus concentrations because it had the highest adjusted  $R^2$  and the lowest AIC. This model can explain 34.4% of the variability in pre-settlement phosphorus.

Although it is tempting to interpret this model as a final model, OLS does not take into account spatial autocorrelation. If spatial autocorrelation does exist, but is not accounted for, the estimated coefficients ( $\beta$ ) and the standard errors (SE) of this model would be biased. It is therefore necessary to test for spatial autocorrelation within the best-fit OLS model, Model #7, and consequently reject or accept a spatial regression model. The tests for spatial dependence, conducted in GeoDa using Moran's I and three nearest neighbor matrices (2, 4, and 9), showed the nearest neighbor configuration of nn=2 to have the strongest correlation with reconstructed phosphorus concentrations in my dataset (Appendix D). However, even at nn=2, the Moran's I value was low. Furthermore, diagnostic testing of Model #7 residuals confirmed that spatial autocorrelation was not present in the dataset (Table 5.10). This conclusion was reached because the p-values of the tested spatial regression models were not significant.

<b>DIAGNOSTICS FOR SPATIAL DEPENDENCE (nn=2)</b>					
TEST	MI/DF	Test-statistic value	p-value		
Moran's I (error)	-0.223	-1.5792292	0.1142835		
Lagrange Multiplier (lag)	1	1.7645536	0.1840576		
Robust LM (lag)	1	0.3261957	0.5679078		
Lagrange Multiplier (error)	1	2.3995222	0.1213723		
Robust LM (error)	1	0.9611643	0.3268937		
Lagrange Multiplier (SARMA)	2	2.7257179	0.2559281		

Table 5.10: Results of diagnostic testing for spatial autocorrelation, conducted in GeoDa.

### 5.3.5 Final OLS Model Interpretation

Since spatial autocorrelation was not present in my data, I was able to accept OLS Model #7 as the best fit model. In this model, the intercept, wetland/lake ratio, and the interaction term of wetland/lake ratio\*max lake depth were statistically significant (Table 5.11). Although maximum lake depth did now show statistical significance, it was influential in the relationship between wetlands and lake phosphorus concentrations. As shown in Table 5.11, the interaction term actually has a negative estimated coefficient ( $\beta$ ), which implies that as maximum depth decreases (the lake becomes shallower), the influence of wetland within the 100 m buffer has a greater effect on pre-settlement lake phosphorus concentrations

Independent Variables	Estimated Coefficient (β)	Std. Error (SE)	t-value	p-value
Intercept	9.41262	1.67794	5.61	1.26E-06
Wetland_ratio100	12.2749	2.64739	4.637	3.17E-05
Max_d	0.05978	0.11485	0.52	0.6053
Wet_ratio100*max_d	-0.7294	0.34933	-2.088	0.0426

Table 5.11: Final OLS model (#7).

## 6. DISCUSSION

It would be easy to argue that all biotic and abiotic characteristics of a watershedhydrology, geology, topography, climate, land cover- contribute to phosphorus dynamics and primary productivity in aquatic ecosystems. However, overfitting from the addition of too many parameters can cause overly complex statistical models that seldom produce meaningful results. It is therefore important to consider AIC values, which impose a penalty to increasing the number of estimated parameters in a regression model. Therefore, simplified models that quantify the strength of correlation between phosphorus and spatial patterns in land cover and/or abiotic variables (e.g., morphometry and latitude) can be used to provide insight into complex dynamics in landscape limnology (Soranno et al. 1996).

### **6.1 Discussion of Model Results**

Past nutrient-related research has focused primarily on agriculture as a single predictive variable that causes a synchronous shift in phosphorus loading to aquatic ecosystems (e.g. Osborne and Wiley 1988; Shapley et al. 1994; Sims et al. 1998). However, my research has demonstrated that natural factors contributed to pre-settlement phosphorus variability in lakes in Michigan. This suggests that anthropogenic factors alone may not adequately explain modern spatial heterogeneity in lake phosphorus concentrations.

The results of my regression models support the distance decay theory, central to geographical studies, which implies that near things are more related than distant things (Montello and Sutton 2006). As such, it was not surprising to find that wetlands within a 100 m buffer of lakes had a more significant relationship to reconstructed phosphorus concentrations than wetlands in the local catchment. This metric of proximity likely represents a degree of hydrologic connectivity that facilitates an exchange of materials between terrestrial and aquatic

networks. Although the proximity of land covers to surface waters is not as frequently investigated as land cover composition (primarily because these spatially explicit measures are often more difficult to quantify), my results indicate this parameter is important to understanding phosphorus variability (Osborne and Wiley 1988; Soranno et al. 1996, Gémesi et al. 2011).

Although wetlands within the 100 m buffer showed higher correlation with diatominferred phosphorus than any other land cover metric, it is necessary to recognize the effect of collinearity between the native land cover variables. For example, a greater wetland/lake ratio inevitably corresponds to a lower forest/lake ratio, although this metric was not derived. From this standpoint, it could be argued that not only does the increase of wetlands yield higher phosphorus concentrations, but the corresponding decrease in forests results in greater observed phosphorus concentrations. Other studies have demonstrated forested areas to be important in the physical retention of fine sediments from upslope erosion (e.g., Omernik et al. 1981; Filippelli and Souch 1999; Reynolds and Davies 2001). Therefore, it is necessary to acknowledge that not only is the increase of wetlands influencing the phosphorus concentrations, but also the decrease in forest cover.

However, land cover and proximity alone did not produce the most successful model for explaining pre-settlement phosphorus variability. The model best fit to describe this variability also included maximum lake depth and an interaction term (wetland/lake ratio in the 100 m buffer X maximum lake depth). Based on these findings, I was able answer the following research question: "Are land cover metrics combined with hydrogeomorphic features and/or geographic variables better predictors of water quality than land cover alone?" My results demonstrated that the strength of the relationship between phosphorus and land cover was greatly improved when combined with hydrogeomorphic features (lake area and lake depth).

While wetland cover alone was able to explain some of the variability in pre-settlement phosphorus, the strength of this relationship increased (as determined by a significant,  $\geq$ 3, decrease in AIC values) when lake depth was incorporated in the model. I speculate that this results from the combination of phosphorus additions from adjacent wetlands and resuspended lake sediments. Lake sediments typically contain greater concentrations of phosphorus than the overlying water column and, in shallow lakes, wind and wave disturbances can resuspend these sediments. In deep waters where sediments are more likely left undisturbed, phosphorus becomes buried and is essentially "removed" from the system. The exception to this is if deep lakes develop anoxic conditions in their hypolimnion and consequentially release phosphorus into the overlying waters. However, internal phosphorus loading such as this is more often the result of cultural eutrophication and soil erosion associated with shoreland development (Garrison and Wakeman 2000).

Maximum depth, was thus an important variable in this study. Although other studies have incorporated mean lake depth, this metric was not available for all 48 lakes in my dataset. I was, however, able to test mean depth for 29 of the lakes (from the Soranno/Garrison dataset) against pre-settlement phosphorus. The results of the correlation were weaker (r = -0.099) than the results from the maximum depth bivariate correlation testing (r = -0.227) of those same 29 lakes.

## **6.2 Nutrient Policy Implications**

Under the jurisdiction of the CWA, states are required to establish a set of water quality standards that define watershed goals and pollution limits for surface waters within their administrative authorities (Clean Water Act 1972). Designated uses, antidegradation policies,

and water quality criteria are the three major, interrelated components of these water quality standards that must be reported to and approved by the EPA. However, efforts to establish numeric nutrient criteria (a subset of water quality criteria) specific to phosphorus have been stymied by substantial aquatic ecosystem variability, and only eight states or territories have adopted site-specific standards for some or all of their lakes (Obreza et al. 2011; USEPA 2013b).

Ideally, nutrient criteria would be site-specific for every lake based natural background levels inferred from paleolimnological reconstructions. However with nearly 70,000 lakes > 4 ha in the continental United States, this is an unrealistic goal. It is thus necessary to develop a classification scheme to condense the number of different ecosystems types into more manageable facets (Martin et al. 2011; Herlihy et al. 2013). Numeric nutrient criteria have been derived to reflect regional patterns in lake conditions, minimally-disturbed reference lakes, diatom-inferred phosphorus reconstructions, lake morphometry, and/or designated usage of the water body (e.g., recreational or ecological) (WDNR 2007; Heiskary and Wilson 2008; Soranno et al. 2011).

In 2001, the Michigan Department of Environmental Quality (MDEQ) began developing numeric phosphorus criteria based on the relationship between total phosphorus and biological responses (namely, fish and algae), however the Michigan Legislature withdrew the MDEQ's rule-making authority in 2006 (MDEQ 2013). As of now, Michigan has only a set *narrative* nutrient criterion (a statement of water quality goals) and a total phosphorus limit of 1  $\mu$ g/L for effluents from point source discharges (MDEQ 2013). Methods for quantifying nutrient criteria for Michigan lakes and streams are currently being developed by a team of scientists at Michigan State University (MDEQ 2013).

Given the substantial range of regional geographic patterns and diversity of lake types in Michigan, a single, statewide nutrient criterion for total phosphorus concentrations is not appropriate for adoption (Soranno et al. 2008). Although aquatic ecosystems should not be treated as if they are all the same, it is impractical to devise individual management strategies for each system. Rather, the methods described in this study (namely, the comparison of diatominferred phosphorus to lake morphometry and pre-settlement land cover) can be applied to the nutrient criteria development process. Unlike the commonly used approach of defining reference site conditions, the paleolimnological approach described in this thesis is guaranteed to quantify nutrient conditions in lakes as they were before significant human disturbances began (Herlihy et al. 2013). However, this method can only be applied to natural lakes and is less applicable in arid regions (e.g., the southwestern United States) that are dominated by constructed impoundments. Additionally, given the underlying uncertainties introduced through inferring phosphorus levels from fossil diatom assemblages and interpreting GLO survey notes, using minimally-disturbed reference sites is certainly a defensible approach to establishing nutrient criteria (Herlihy et al. 2013).

One of the significant findings of this study was the demonstrated importance of dating the lake-core sediments of paleolimnological reconstructions of pre-settlement phosphorus concentrations. For example, I determined that 15 of 63 sediment samples analyzed were not truly representative of undisturbed conditions;12 of these samples were collected as part of the EPA's NLA in 2007. This finding alone is meaningful enough to refute the conclusions drawn in a recent, controversial publication by Bachmann et al. (2013) that used data from the same dataset. Therefore, their deduction that phosphorus concentrations in lakes were 14% higher

during pre-settlement conditions, is not entirely based on data reconstructed from undisturbed lake conditions.

#### **6.3 Lake Management Implications**

Most inland lakes in Michigan have highly sought-after real estate along their shore lines. However, the majority of these residential developments have a constructed sea-wall that abruptly marks the boundary between terrestrial and aquatic ecosystems, thus altering the natural cycle of allochthonous materials such as sediments and nutrients. Meanwhile, lake associations and foundations spend thousands of dollars annually to stock sport fish in support of local recreational fishing (Goodman 1991). It could simply be that the rate at which these fish are harvested is greater than the rate at which they are produced. However, it is also possible that these anthropogenically altered lakes are not as well-suited to productive fisheries as they might have been if left undisturbed. This, I hypothesize, is due in part to the substantial decrease in wetlands around inland lake shores.

My research demonstrated that wetlands in the 100 m buffer (Model #5) were significantly related (p< 0.005) to reconstructed phosphorus concentrations, and could explain approximately 30% of the variability in the study lakes. When this variable was combined with maximum lake depth (Model #7), the explanatory strength increased to 34%. These results revealed that wetlands in an undisturbed environment were capable of influencing pre-settlement lake phosphorus concentrations to a degree. Although most studies of phosphorus are in regards to cultural eutrophication processes and over-enrichment of freshwaters, this element is fundamental to normal food web functioning when in manageable concentrations. Thus, I speculate that the re-emergence of hydrologically-connected wetlands could improve fish

productivity by restoring a natural nutrient balance that stimulates primary productivity in oligotrophic lakes to a reasonable level. This, of course, would greatly benefit lakes associations that invest tremendous amounts of money annually to stock fish in order to appease the recreational fishing industry interests.

The point here is not to say that lake managers should enhance waters with phosphorus as some have suggested (Stockner et al. 2000), because biomanipulation almost always has unforeseen negative repercussions. It is also important to recognize that excess phosphorus from cultural eutrophication is undeniably detrimental to surface waters for a myriad of interrelated reasons (e.g., nuisance algae blooms, dissolved oxygen depletion, and toxin-producing cyanobacteria). Rather, I argue that the restoration of native land covers within a 100 m buffer of lakes could yield satisfactory results for a diverse group of stakeholders. Poor shore habitat was identified by the EPA's NLA as a leading stressor affecting the biological health of lakes. (USEPA 2010). While it would be unreasonable and infringing to suggest an outlaw of all shoreline development, the construction of wetlands at public or association/foundation-owned parks would be beneficial and economical in the long-run. It is also sensible to encourage new residential developments to adopt natural lakeshores comprised of native wetland vegetation.

### **6.4 Future Research Directions**

Several shortcomings exist with the final regression model in this thesis. First, it is possible that important independent variables may have been excluded from the model. For example, I excluded soil texture and drainage variables as potential predictor variables to reduce the introduction of multicollinearity into the model selection process. Many studies have made reference to the tendency of vegetation patterns to covary with edaphic patterns in soil texture

and drainage class (e.g., Yahner 2000; Harmon 2009). For example, coarse-textured sandy soils with poor water-retaining capacity provide optimal growing medium for coniferous tree communities such as pine barrens because these species require less water than their deciduous counterparts (Yahner 2000; Kost et al. 2007). In contrast, poorly drained soils with extended periods of saturation are optimal locations for hydric vegetation growth, characteristic of wetland ecosystems (Mitsch and Gosselink 2007). Regardless of the confounding potential of these edaphic variables, the relationships here are worth exploring in GIS. Using soil patterns, rather than native land cover patterns, could potentially improve the model results. These results of such an analysis could be more applicable to establishing nutrient criteria because unlike native vegetation, soil texture has remained relatively unchanged by Euro-American settlement.

Furthermore, when I expanded the examination of wetlands near the lake perimeters, versus the entire lake catchment, I found a spatial trend in patterns of wooded vs non-wooded wetlands. I noticed that 74% of the wetlands within the local catchment were wooded (i.e., black ash swamp, cedar swamp, mixed conifer swamp, and mixed hardwood swamp), and the remaining 26% were non-wooded wetlands (bog, emergent marsh, and wet prairie). Within a 100 m buffer, the percentage of non-wooded wetlands was slightly higher (33%), when compared to the entire local lake catchment. However, at both spatial scales, the total area of wooded wetlands is twice that of non-wooded wetlands. This may suggest that there is some underlying mechanism driving phosphorus loading from these land cover types, which warrants further investigations.

Future studies should consider expanding upon hydrogeomorphic features in terms of interactions with land cover. Of these hydrogeomorphic features, water residence time could be an important predictor of pre-settlement phosphorus variability. Although the data was not

available for this study, modern studies of phosphorus loading could easily incorporate this variable with knowledge of lake volume and watershed hydrology. While it is unlikely that residence time alone could explain the remaining ~60% variability in phosphorus concentrations, the ultimate model results would likely increase in fit if such a variable was considered in conjunction with wetland cover.

Finally, a significant proportion of the variation in diatom-inferred phosphorus was not explained by the model, illustrating that the landscape factors controlling water chemistry are complex and remain poorly understood. The use of a larger and more diverse lake dataset would likely increase the power of the explanatory data analysis. Although there is some debate as to the minimum number of observations required to generate significant results, large sample sizes are generally more reliable. Nonetheless, the findings of this study can improve our understanding of natural mechanisms influencing lake phosphorous.

## 7. CONCLUSION

The many inland lakes of Michigan are ecologically and economically valuable resources that can increase property values, and provide recreational opportunities, wildlife habitat, and drinking water sources. Anthropogenic impacts on lakes have increased during recent decades due to intensification of agriculture practices and water consumption. Excess nutrients in surface runoff from agricultural and urban are now recognized as one of the leading causes of water body impairments in the United States (Interlandi et al. 2003; USEPA 2010). It is therefore in our best interest to protect these precious water reserves, in part through the establishment of numeric nutrient criteria that define phosphorus pollution limits for freshwaters.

Lake sediment cores containing diatom fossil assemblages can be interpreted to determine natural, pre-settlement nutrient concentrations in a lake before anthropogenic perturbation (Battarbee 2000; Wetzel 2001). Determining lake conditions prior to Euro-American settlement can facilitate in defining practical restoration goals, as it would be unrealistic to expect a reduction in phosphorus beyond natural, ambient concentrations. Furthermore, the evaluation of inferred phosphorus concentrations with lake morphometry and land cover data can be used to establish nutrient criteria as an alternative to minimally-disturbed reference conditions.

The primary objective of this research was to understand the interaction between in-lake phosphorus concentrations and land cover/HGM variables, as it existed before human disturbances. I compiled data from two pre-existing datasets containing reconstructed phosphorus concentrations from lake sediment cores collected in 2007. A pollen-based relative dating technique was used to verify that the bottom-core sediments were representative of pre-

settlement lake conditions. The phosphorus reconstructions for 48 lakes in Michigan were supplemented by pre-settlement land cover metrics at two spatial scales: a 100 m buffer and local catchment. Finally, hydrogeomorphic features and geographic variables were incorporated in the statistical modeling to test their relative importance to predicting phosphorus variability.

The findings of this study indicated that, when considering land cover percentages alone, native wetlands were able to best explain some pre-settlement phosphorus variability in my dataset. However, wetlands in a 100 m buffer, when combined with lake area (wetland/lake ratio), was the predictor variable with the strongest correlation with total phosphorus. Furthermore, results of OLS Model #7 suggested that wetlands are influenced by lake depth such that as depth decreases, the effect of wetlands on lake phosphorus increases. Although this model applied pre-settlement data, these interactions likely still occur and should be carefully considered when establishing realistic nutrient criteria.

APPENDICES

# APPENDIX A

Lake Name	Data Source	Rational
Belleville	EPA	No lab data for core
Besser	EPA	No lab data for core
Bridge	EPA	No lab data for core
Clark Co.	EPA	No lab data for core
Donnell	EPA	No lab data for core
Ford	EPA	No lab data for core
Forestville	EPA	No lab data for core
Pine (2)	EPA	No lab data for core
Silver	EPA	No lab data for core
Stony Creek	EPA	No lab data for core
Sullivan	EPA	No lab data for core
West	EPA	No lab data for core
Brighton	EPA	Man-made lake
Hi-Land	EPA	Man-made lake
Palmer	EPA	Man-made lake
Loon	EPA	Uncertainty in representation of pre-settlement conditions
Squaw	EPA	Uncertainty in representation of pre-settlement conditions
Thornapple	EPA	Uncertainty in representation of pre-settlement conditions
Deer	EPA	Duplicated in Soranno/Garrison dataset
Ottawa	EPA	Duplicated in Soranno/Garrison dataset
Big	EPA	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Bogie	EPA	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Crooked	EPA	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Martin	EPA	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Muskegon	EPA	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Pere Marquette	EPA	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Round	EPA	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Saddle	EPA	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Tallman	EPA	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Tims	EPA	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Upper Scott	EPA	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Wyckoff	EPA	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Big Chub	Soranno/Garrison	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Hess	Soranno/Garrison	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Portage	Soranno/Garrison	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Reeds	Soranno/Garrison	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions
Townline	Soranno/Garrison	> 3% <i>Ambrosia</i> , therefore not pre-settlement conditions

**Table A.1:** List of lakes excluded from final dataset.

# **APPENDIX B**

	DI-TP	FTZ	Latitude	Longitude	Max_D	Lake Area	Perimeter	Lake area/depth
DI-TP	1.000	0.004	0.101	-0.210	-0.248	0.042	-0.051	0.074
FTZ		1.000	0.826	-0.531	-0.171	0.121	0.142	0.160
Latitude			1.000	-0.816	-0.282	0.106	0.080	0.169
Longitude				1.000	0.212	-0.213	-0.174	-0.254
Max_D					1.000	-0.087	-0.029	-0.183
Lake Area						1.000	0.940	0.989
Perimeter							1.000	0.908
Lake area/depth								1.000

**Table B.1:** Correlation matrix of all potential predictor variables.

	Catchment/Lake Area	% Wet_LC	% Dec_LC	% Mix_LC	% Ever_LC	% For_LC	% Grass_LC
DI-TP	0.039	0.131	0.066	0.010	-0.185	-0.132	0.055
FTZ	0.067	0.143	-0.678	0.479	0.280	0.181	-0.470
Latitude	0.114	0.238	-0.448	0.256	0.228	0.048	-0.377
Longitude	-0.115	-0.239	0.167	-0.124	-0.041	-0.014	0.319
Max_D	0.292	-0.190	0.107	0.141	-0.210	0.212	-0.113
Lake Area	-0.057	0.057	-0.154	0.150	-0.018	0.011	-0.086
Perimeter	-0.069	0.137	-0.126	0.061	0.068	-0.006	-0.159
Lake area/depth	-0.063	0.062	-0.178	0.153	0.004	0.004	-0.080
Catchment/Lake							
Area	1.000	-0.022	-0.048	0.097	-0.066	0.027	-0.014
% Wet_LC		1.000	-0.027	-0.227	-0.136	-0.782	0.050
% Dec_LC			1.000	-0.696	-0.297	-0.062	0.140
% Mix_LC				1.000	-0.342	0.380	-0.321
% Dec_LC					1.000	0.165	-0.133
% For_LC						1.000	-0.660
% Grass_LC							1.000

Table B.1 (cont'd)
Table B.1 (cont'd)

	% Wet_100	% Dec_100	% Mix_100	% Ever_100	% For_100	% Grass_100	WR100	WRLC
DI-TP	0.269	0.030	-0.088	-0.146	-0.198	-0.030	0.565	0.242
FTZ	-0.140	-0.507	0.394	0.307	0.215	-0.374	-0.067	0.031
Latitude	-0.086	-0.279	0.222	0.169	0.125	-0.279	0.023	0.123
Longitude	0.062	0.118	-0.182	-0.034	-0.136	0.281	-0.004	-0.092
Max_D	-0.227	0.084	0.245	-0.168	0.268	-0.130	-0.313	-0.238
Lake Area	-0.117	-0.153	0.188	0.037	0.109	-0.064	-0.179	-0.076
Perimeter	-0.148	-0.145	0.087	0.172	0.095	-0.116	-0.267	-0.091
Lake area/depth	-0.113	-0.157	0.182	0.043	0.102	-0.056	-0.149	-0.051
Catchment/Lake A	0.052	0.036	-0.053	-0.009	-0.037	-0.044	0.031	0.042
% Wet_LC	0.449	-0.034	-0.294	-0.074	-0.471	0.076	0.428	0.654
% Dec_LC	0.050	0.807	-0.586	-0.338	-0.169	0.062	-0.004	0.007
% Mix_LC	-0.101	-0.654	0.896	-0.242	0.265	-0.255	-0.114	-0.143
% Dec_LC	-0.244	-0.054	-0.317	0.878	0.238	-0.105	-0.163	-0.132
% For_LC	-0.506	0.025	0.399	0.124	0.636	-0.622	-0.521	-0.516
% Grass_LC	0.278	0.004	-0.272	-0.144	-0.459	0.907	0.322	0.038
% Wet_100	1.000	-0.208	-0.310	-0.273	-0.832	0.260	0.702	0.511
% Dec_100		1.000	-0.569	-0.173	0.187	-0.092	-0.132	-0.027
% Mix_100			1.000	-0.273	0.463	-0.215	-0.290	-0.248
% Ever_100				1.000	0.266	-0.115	-0.227	-0.140
% For_100					1.000	-0.463	-0.691	-0.458
% Grass_100						1.000	0.291	0.005
WR100							1.000	0.526
WRLC								1.000

## **APPENDIX C**

**Figure C.1:** Histogram of the dependent variable, showing the distribution of presettlement phosphorus concentrations for the 48 lakes.





Latitude (decimal degrees)

Latitude (decimal degrees)

**Figure C.2:** Univariate and bivariate testing results for latitude. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



Longitude (decimal degrees)

Longitude (decimal degrees)

**Figure C.3:** Univariate and bivariate testing results for longitude. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.4:** Univariate and bivariate testing results for native deciduous forest (%) in the 100 m buffer. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.5:** Univariate and bivariate testing results for native deciduous forest (%) in the local catchment. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.6:** Univariate and bivariate testing results for native evergreen forest (%) in the 100 m buffer. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.7:** Univariate and bivariate testing results for native evergreen forest (%) in the local catchment. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.8:** Univariate and bivariate testing results for native mixed forest (%) in the 100 m buffer. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.9:** Univariate and bivariate testing results for native mixed forest (%) in the local catchment. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.10:** Univariate and bivariate testing results for native total forest (%) in the 100 m buffer. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.11:** Univariate and bivariate testing results for native total forest (%) in the local catchment. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.12:** Univariate and bivariate testing results for native wetland cover (%) in the 100 m buffer. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.13:** Univariate and bivariate testing results for native wetland cover (%) in the local catchment. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.14:** Univariate and bivariate testing results for native grassland cover (%) in the 100 m buffer. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.15:** Univariate and bivariate testing results for native grassland cover (%) in the local catchment. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.16:** Univariate and bivariate testing results for wetland/lake ratio in the 100 m buffer. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.17:** Univariate and bivariate testing results for wetland/lake ratio in the local catchment. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.18:** Univariate and bivariate testing results for maximum lake depth (m). The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.19:** Univariate and bivariate testing results for lake area (ha). The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.20:** Univariate and bivariate testing results for lake/catchment ratio. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).



**Figure C.21:** Univariate and bivariate testing results for lake perimeter. The histogram (left) shows the distribution of the variable and the scatter plot (right) shows the strength of the relationship with pre-settlement phosphorus (n=48).

## **APPENDIX D**



**Figure D.1:** Moran's I results; nearest neighbor= 2



**Figure D.2:** Moran's I results; nearest neighbor= 4



**Figure D.3:** Moran's I results; nearest neighbor= 9

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